Addressing Nitrate in California's Drinking Water

TECHNICAL REPORT 6:

Drinking Water Treatment for Nitrate

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

California Nitrate Project,
Implementation of Senate Bill X2 1

Center for Watershed Sciences
University of California, Davis
http://groundwaternitrate.ucdavis.edu

Prepared for the California State Water Resources Control Board

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Prepared by:

Vivian B. Jensen and Jeannie L. Darby[1] of UC Davis

Chad Seidel and Craig Gorman of Jacobs Engineering Group, Inc.[2]

Center for Watershed Sciences
University of California, Davis
California Nitrate Project, Implementation of Senate Bill X2 1
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[1] Corresponding author: jdarby@ucdavis.edu

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Disclaimer: This document is intended as a guide of nitrate treatment technologies and should be used as an informational tool only. The contents of this document are solely the responsibility of the authors and do not necessarily represent the official views of supporting agencies. The selection and design of the most appropriate nitrate treatment alternative for a particular water system depend on a variety of factors and require the expertise of experienced professional engineers. Discussion of proprietary technologies is intended to provide information about treatment alternatives and does not imply endorsement.

For further inquiries, please contact

Thomas Harter, Ph.D.

ThHarter@ucdavis.edu

125 Veihmeyer Hall

University of California

Davis, CA 95616-8628

Phone: 530-752-2709

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Name	**Affiliation**
Michelle De Haan, Chair	Water Works Engineers
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Jess Brown	Carollo Engineers
Susan Brownstein	California Department of Public Health
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Acronyms and Abbreviations

AF	Acre Feet
AFY	Acre Feet per Year
AWWA	American Water Works Association
BMP	Best Management Practice
BD	Biological Denitrification
BV	Bed Volume
CD	Chemical Denitrification
CIP	Clean-In-Place
DBP	Disinfection Byproduct
DL	Detection Limit
DO	Dissolved Oxygen
EBCT	Empty Bed Contact Time
ED	Electrodialysis
EDR	Electrodialysis Reversal
FBR	Fluidized Bed Reactor
FXB	Fixed Bed Reactor
GAC	Granular Activated Carbon
GHG	Greenhouse Gas
GPM	Gallons per Minute
GWUDI	Groundwater Under Direct Influence (of Surface Water)
HERO™	High Efficiency Reverse Osmosis
HLR	Hydraulic Loading Rate
ISEP®	Ion Exchange Separation System
IX	Ion Exchange
LSI	Langelier Saturation Index
MBfR	Membrane Biofilm Reactor
MBR	Membrane Bioreactor
MCL	Maximum Contaminant Level
MGD	Million Gallons per Day
MIEX®	Magnetic Ion Exchange
NDMA	N-nitrosodimethylamine
O&M	Operations and Maintenance
PICME	Permits Inspection Compliance Monitoring and Enforcement
POE	Point-of-Entry
POU	Point-of-Use
PRB	Permeable Reactive Barrier
RO	Reverse Osmosis
SBA IX	Strong Base Anion Exchange
SBR	Sequencing Batch Reactor

SDI	Silt Density Index
SDWA	Safe Drinking Water Act
SED	Selective Electrodialysis
SMI	Sulfur Modified Iron
TDS	Total Dissolved Solids
TSS	Total Suspended Solids
ULPRO	Ultra-Low Pressure Reverse Osmosis
USBR	Up-flow Sludge Blanket Reactor
U.S. EPA	United States Environmental Protection Agency
VSEP	Vibratory Shear Enhanced Process
WAC	Weak Acid Cation Exchange
WBA IX	Weak Base Anion Exchange
WQM	Water Quality Monitoring
ZVI	Zero Valent Iron

Unit Conversions

Metric to US		**US to Metric**	
Mass		*Mass*	
1 gram (g)	0.04 ounces (oz)	1 ounce	28.35 grams
1 kilogram (kg)	2.2 pounds (lb)	1 pound	0.45 kilograms
1 megagram (Mg) (1 tonne)	1.1 short tons	1 short ton (2000 lb)	0.91 megagrams
1 gigagram (Gg) (1000 tonnes)	1102 short tons	1000 short tons	0.91 gigagrams
Distance		*Distance*	
1 centimeter (cm)	0.39 inches (in)	1 inch	2.54 centimeters
1 meter (m)	3.3 feet (ft)	1 foot	0.30 meters
1 meter (m)	1.09 yards (yd)	1 yard	0.91 meters
1 kilometer (km)	0.62 miles (mi)	1 mile	1.61 kilometers
Area		*Area*	
1 square meter (m^2)	10.8 square feet (ft^2)	1 square foot	0.093 square meters
1 square kilometer (km^2)	0.39 square miles (mi^2)	1 square mile	2.59 square kilometers
1 hectare (ha)	2.8 acres (ac)	1 acre	0.40 hectares
Volume		*Volume*	
1 liter (L)	0.26 gallons (gal)	1 gallon	3.79 liters
1 cubic meter (m^3) (1000 L)	35 cubic feet (ft^3)	1 cubic foot	0.03 cubic meters
1 cubic kilometer (km^3)	0.81 million acre-feet (MAF, million ac-ft)	1 million acre-feet	1.23 cubic kilometers
Farm Products		*Farm Products*	
1 kilogram per hectare (kg/ha)	0.89 pounds per acre (lb/ac)	1 pound per acre	1.12 kilograms per hectare
1 tonne per hectare	0.45 short tons per acre	1 short ton per acre	2.24 tonnes per hectare
Flow Rate		*Flow Rate*	
1 cubic meter per day (m^3/day)	0.296 acre-feet per year (ac-ft/yr)	1 acre-foot per year	3.38 cubic meters per day
1 million cubic meters per day (million m^3/day)	264 mega gallons per day (mgd)	1 mega gallon per day (694 gal/min)	0.0038 million cubic meters/day

Nitrate Units

*Unless otherwise noted, nitrate concentration is reported as milligrams/liter as nitrate (mg/L as NO_3^-).

To convert from:

Nitrate-N (NO_3-N)	→	Nitrate (NO_3^-)	multiply by 4.43
Nitrate (NO_3^-)	→	Nitrate-N (NO_3-N)	multiply by 0.226

Summary

Objective

The purpose of the first three sections of this document is to provide a detailed guide to the current state of nitrate treatment alternatives that can be used as a reference tool for the drinking water community. The remainder of this document focuses on nitrate treatment of drinking water in California and specifically in the Tulare Lake Basin and the Salinas Valley.

Background

Nitrate contamination of potable water sources is becoming one of the most important water quality concerns in California and across the United States. The maximum contaminant level (MCL) for nitrate is 45 mg/L as nitrate (NO_3^-), which is approximately equivalent to 10 mg/L as nitrogen (N). The major health concern of nitrate exposure through drinking water is the risk of methemoglobinemia, or "blue baby syndrome," especially in infants and pregnant women. Due to the nature of the infant digestive system, nitrate is reduced to nitrite which can render hemoglobin unable to carry oxygen (SWRCB 2010). Nitrate is naturally occurring at low levels in most waters, but it is particularly prevalent in groundwater that has been impacted by certain agricultural, commercial, or industrial activities. Of specific concern are crop fertilization activities and discharges from animal operations, wastewater treatment facilities, and septic systems. Small rural communities are particularly impacted by nitrate (Pacific Institute 2011). Nitrate presents unique water treatment challenges. The United States Environmental Protection Agency (U.S. EPA) lists only anion exchange (IX), reverse osmosis (RO), and electrodialysis reversal (EDR) as accepted potable water treatment methods for nitrate removal (U.S. EPA 2010). Due to the production of high-strength brine residuals, sustainable application of these three technologies is often limited by a lack of local residual disposal options and the challenge of increasing salt loads. The lack of affordable and feasible nitrate treatment alternatives can force impacted utilities to remove nitrate-contaminated sources from their available water supply. In many instances, this action can severely compromise a water utility's ability to provide an adequate supply of safe and affordable potable water.

The need for additional nitrate treatment technologies has driven the drinking water community to begin developing alternative options to effectively remove nitrate while limiting cost and brine production challenges. Promising treatment options include weak base anion (WBA) exchange and improvements in strong base anion (SBA) exchange such as low brine residual technologies; biological treatment using fluidized bed, fixed bed, and membrane biofilm (MBfR) reactors; and chemical reduction using media such as zero valent iron (ZVI) and sulfur modified iron (SMI). A summary of the options to address nitrate contamination of drinking water is presented in Figure S.1. In this diagram treatment options are classified in terms of their ability to either remove nitrate to a residual waste stream or transform nitrate to other nitrogen species through reduction.

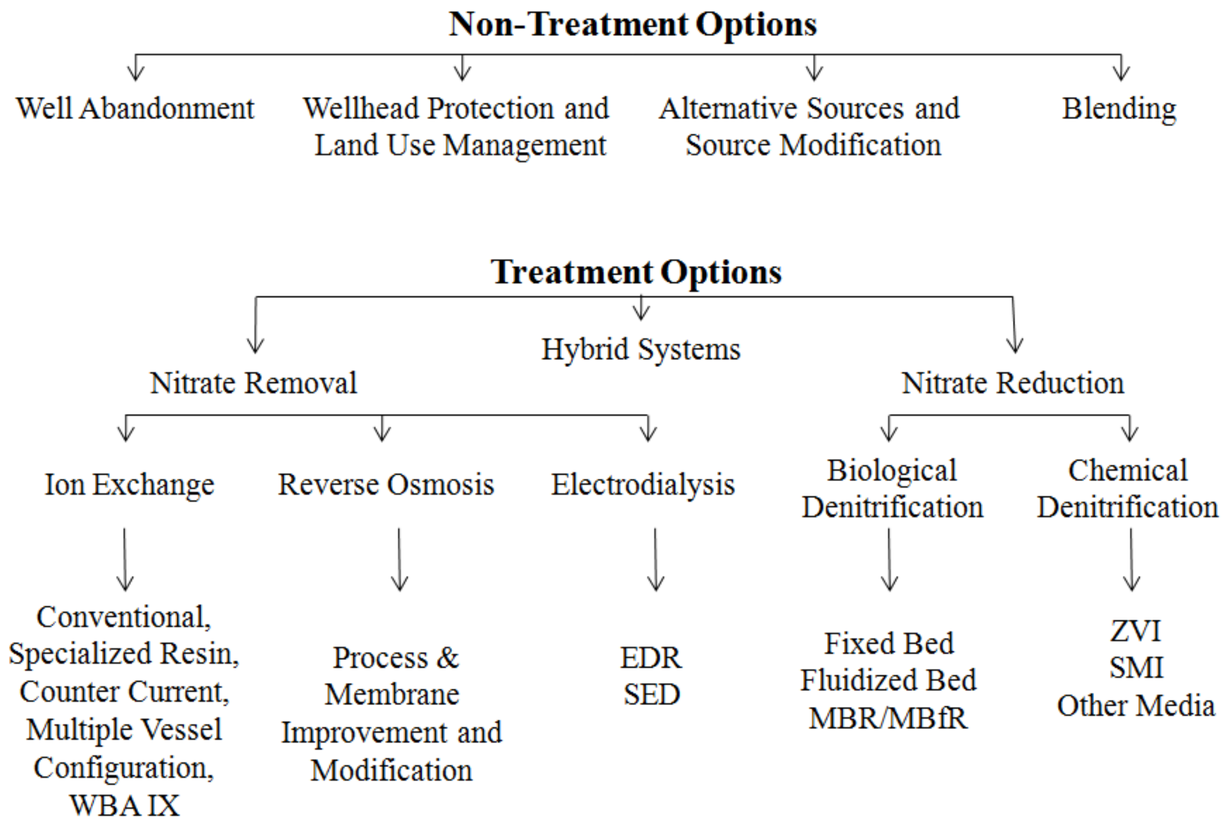

Figure S.1. Summary of nitrate management options.[4]

Approach

This report includes a comprehensive literature review and case studies of specific systems across the range of nitrate treatment alternatives. The literature review is intended to provide background information about current and emerging potable water treatment alternatives to address nitrate contamination. In addition to peer-reviewed literature, information found in the "grey papers" of conference proceedings has been included to assure capture of the most recent technology developments. For each of the major treatment technologies, subsections of the literature review detail the following:

- Design considerations including water quality, system layout, and site considerations; residuals management and disposal; and maintenance, monitoring, and operational complexity,
- cost considerations,
- selected research, and
- a summary of advantages and disadvantages.

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[4] For the purposes of this discussion, blending has been categorized as a "non-treatment" option; however, in practice, blending is sometimes referred to as "treatment." Treatment options throughout this report refer to treatment technologies available for the removal or reduction of nitrate in drinking water. Blending can sometimes be used to cost-effectively address the nitrate problem through dilution, but has been categorized separately from treatment options for simplicity.

Information is summarized in tables whenever appropriate, including a summary table of selected research studies for each of the major treatment technologies (Appendix).

A survey was conducted to collect detailed information about the application of nitrate treatment. A subset of utilities, currently treating for nitrate and/or in design for future treatment, was included in the survey. The survey was developed to gather information with respect to the benefits and limitations of the various nitrate treatment technologies and was conducted via phone and in-person when applicable. The list of utilities included in the survey was developed with the intention of covering a range of utilities with respect to geographic location, treatment type, population size and residual handling techniques (Table S.1). Detailed case studies have been compiled for each of the treatment technologies where full-scale facilities have been in operation or are moving ahead with design. This survey was conducted through collaboration with Jacobs Engineering in the completion of the associated assessment of nitrate treatment alternatives for the American Water Works Association (AWWA) and is complemented by a parallel survey of nitrate treatment systems in California. Details from the initial survey are included as examples following a discussion of each of the treatment technologies. Details of the complementary survey of California systems are included in the second half of this report.

Table S.1. Utilities included in the case studies.

Case #	Treatment Type	Location	Capacity (gpm)	Avg. Influent Nitrate mg/L as NO_3^- (mg/L as N)
	Ion Exchange			
1	Conventional ion exchange with blending	California	400	31 – 53 (7 – 12)
2	Conventional ion exchange with blending	California	400	~45 (~10)
3	Counter Current Ion Exchange (MIEX®)	Indian Hills, CO	50	53 – 71 (12 – 16)
4	Multiple vessel ion exchange	California	500 – 900	35 – 89 (8 – 20)
5	Multiple vessel ion exchange	Chino, CA	5000	40 – 200 (9 – 45)
	Reverse Osmosis			
6	Reverse osmosis and blending	Bakersfield, CA	120	75 – 84 (17 – 19)
7	Reverse osmosis, exploring biological reduction	Brighton, CO	4600	49 – 89 (11 – 20)
8	Reverse osmosis and blending	Arlington Desalter, Riverside, CA	4583	44 – 89 (10 – 20)
	Combined Reverse Osmosis and Ion Exchange			
9	Reverse osmosis, ion exchange and blending	Chino Desalter I, Chino, CA	4940 (RO), 3400 (IX)	148 – 303 (33 – 68)
10	Reverse osmosis, ion exchange and blending	Chino Desalter II, Mira Loma, CA	4167 (RO), 2778 (IX)	70 – 224 (16 – 51)
	Electrodialysis/Electrodialysis Reversal/Selective Electrodialysis			
11	Electrodialysis Reversal (EDR)	Spain	3,260 (each, 2 systems)	~80 (~18)
12	Selective Electrodialysis (SED)	Israel	310	84 – 89 (19 – 20)
	Biological Denitrification			
13	Implementing fluidized bed biological reduction	Rialto, CA	2000 – 4000	17 – 19 (~4 – 5)
14	Implementing fixed bed biological reduction	Riverside, CA	1670	44 – 89 (10 – 20)

Findings

Non-Treatment Options

The focus of this assessment is the current state of nitrate treatment alternatives. However, in practice, non-treatment options are generally considered first as they can often be more sustainable and less costly. Non-treatment options include wellhead protection, land use management, well inactivation, source modification, development of alternative sources (including consolidation/connection to a nearby system), and blending. Blending was found to be the most common method to address nitrate contamination. When a low nitrate water supply source is available, dilution of high nitrate sources to produce water with nitrate levels below the MCL is typically more cost-effective than installing treatment.

Treatment Options

Nitrate treatment technologies were categorized into five major types. Ion exchange (IX), reverse osmosis (RO), and electrodialysis/electrodialysis reversal (ED/EDR) remove nitrate to a concentrated waste stream, while biological denitrification (BD) and chemical denitrification (CD) transform nitrate to other nitrogen species through reduction. Common concerns in the application of the removal technologies include waste management costs and treatment interference from other water quality parameters (e.g., hardness and sulfate). Pretreatment is often required to avoid fouling or scaling of the resin for IX and the membranes for RO and ED/EDR. Due to the destruction of nitrate, both biological and chemical denitrification have the potential for more sustainable treatment without brine residuals, but also have limitations to consider. Full-scale application of these nitrate treatment options is currently limited.

The selection of the most appropriate treatment option depends on various key factors specific to the needs and priorities of individual water systems. A brief comparison of fundamental design considerations, and advantages and disadvantages of these treatment options is listed in Table S.2. It is important to note that the contents of Table S.2 are not intended to provide a comprehensive set of criteria for treatment options. Other important criteria in determining the best treatment option, which are site specific and cannot be broadly generalized, include capital and operations and maintenance (O&M) costs, system size (capacity), and system footprint. Overall, there is no single treatment option that can be considered the best method for nitrate removal across all water quality characteristics and for all systems.

Table S.2. Potable water treatment options for nitrate management (adapted from WA DOH 2005).

	Ion Exchange	Reverse Osmosis	Electrodialysis	Biological Denitrification	Chemical Denitrification
Full-scale Systems	Yes	Yes	Yes	Yes	No
Treatment Type	Removal to waste stream	Removal to waste stream	Removal to waste stream	Biological reduction	Chemical reduction
Common Water Quality Design Considerations	Sulfate, iron, manganese, total suspended solids (TSS), metals (e.g., arsenic), hardness, organic matter	Turbidity, iron, manganese, SDI, particle size, TSS, hardness, organic matter, metals (e.g., arsenic)	Turbidity, iron, manganese, TSS, hydrogen sulfide, hardness, metals (e.g., arsenic)	Temperature and pH, anoxic conditions	Temperature and pH
Pretreatment Needs	Pre-filter, address hardness	Pre-filter, address hardness	Pre-filter, address hardness	pH adjustment, nutrient and substrate addition, need for anoxic conditions	pH adjustment
Post-treatment Needs	pH adjustment	pH adjustment Remineralization	pH adjustment Remineralization	Filtration, disinfection, possible substrate adsorption	pH adjustment, iron removal, potential ammonia control
Waste/Residuals Management	Waste brine	Concentrate	Concentrate	Sludge/biosolids	Waste media, Iron sludge
Start-up Time	Minutes	Minutes	Minutes	Initial plant startup: Days to weeks After reaching steady state: Minutes	Minutes
Water Recovery	Conventional (97%) Low brine (Up to 99.9%)	Up to 85%	Up to 95%	Nearly 100%	Not demonstrated full-scale
Advantages	Nitrate selective resins, common application, multiple contaminant removal	Multiple contaminant removal, desalination (TDS removal)	Multiple contaminant removal, higher water recovery (less waste), desalination, unaffected by silica	No waste brine or concentrate, nitrate reduction rather than transfer to a waste stream, high water recovery, and potential for multiple contaminant removal	No waste brine or concentrate, nitrate reduction rather than transfer to a waste stream, and potential for multiple contaminant removal
Disadvantages	Potential for nitrate peaking, high chemical use (salt), brine waste disposal, potential for disinfection byproduct (DBP) formation (e.g., NDMA)	Membrane fouling and scaling, lower water recovery, operational complexity, energy demands, waste disposal	Energy demands, operational complexity, waste disposal	Substrate addition, potentially more complex, high monitoring needs, possible sensitivity to environmental conditions, risk of nitrite formation (potential incomplete denitrification), post-treatment to address turbidity standards and 4-log virus removal (state dependent)	Inconsistency of nitrate reduction, risk of nitrite formation (potential incomplete denitrification), reduction to ammonia, lack of full-scale systems, pH and temperature dependence, possible need for iron removal

Ion Exchange (IX)

The most commonly used nitrate treatment method is IX. Anion exchange for nitrate removal is similar to a water softener, with nitrate ions removed rather than hardness ions. Nitrate is removed from the treatment stream by displacing chloride on an anion exchange resin. Subsequently, regeneration of the resin is necessary to remove the nitrate from the resin. Regeneration is accomplished by using a highly concentrated salt solution resulting in the displacement of nitrate by chloride. The result is a concentrated waste brine solution high in nitrate that requires disposal. The most significant drawback of this treatment option is the cost for disposal of waste brine, especially for inland communities. The brine volume is largely dependent on the raw water quality and the configuration of the system.

Key factors in the consideration of IX include the pretreatment requirements to avoid resin fouling, the potential need for nitrate selective resin, the frequency of resin replacement, the possible post-treatment requirements to address corrosion or other product water quality concerns (e.g., the potential for NDMA formation), and the management of waste brine. If waste brine disposal options are not limiting, IX can be the best option for low to moderate nitrate contamination and removal of multiple contaminants (including arsenic, perchlorate, and chromium). Application of IX may not be feasible for extremely high nitrate levels due to salt use and waste volume. Current research on brine treatment alternatives may lead to the development of technologies capable of effectively addressing the disposal concern; however, the costs for full-scale implementation of this are unknown at this point.

Modifications to conventional IX have emerged in recent years offering low brine alternatives with improved efficiency. The efficiency of IX systems is dependent on the raw water characteristics. It is important to note that there can be cases where conventional IX systems yield greater water efficiency than a modified system that is implemented at a location with lesser water quality.

Another promising alternative to consider for the future is weak base ion exchange (WBA IX). This emerging technology is more operationally complex than conventional IX, but may offer the advantage of recycling waste as fertilizer.

Reverse Osmosis (RO)

As the second most common nitrate treatment alternative, RO can be feasible for both municipal and Point-of-Use applications and can be used simultaneously for desalination and removal of nitrate and many co-contaminants. Following pretreatment to prevent membrane fouling and scaling, water is forced through a semi-permeable membrane under pressure such that the water passes through, while contaminants are impeded by the membrane.

Key factors in the consideration of RO are the pretreatment requirements, the trade-off between water recovery and power consumption, the management of waste concentrate, and the typically higher costs relative to IX. One deciding factor favoring the selection of RO over IX for nitrate removal would be the need to address salinity.

Recent advancements in membrane technology and optimization of pre- and post-treatment have led to increases in the efficiency of RO treatment systems. For example, the use of Ultra-Low Pressure Reverse Osmosis (ULPRO) membranes enables lower power consumption.

Electrodialysis (ED, EDR, SED)

The use of ED in potable water treatment has increased in recent years, offering the potential for lower residual volumes through improved water recovery, the ability to selectively remove nitrate ions, and the minimization of chemical and energy requirements. ED works by passing an electric current through a series of anion and cation exchange membranes that trap nitrate and other ions in a concentrated waste stream. To minimize fouling and thus the need for chemical addition, the polarity of the system can be reversed with electrodialysis reversal (EDR). By reversing the polarity (and the solution flow direction) several times per hour, ions move in the opposite direction through the membranes, minimizing buildup.

Key factors in the consideration of EDR are the pretreatment requirements, the operational complexity of the system, the limited number of system manufacturers, the management of waste concentrate, and the lack of full-scale installations for nitrate removal from potable water in the United States. Like RO, EDR is commonly used for desalination and can be an alternative for nitrate treatment of high TDS waters. In contrast to conventional RO, EDR is unaffected by silica. EDR costs are similar to RO and evidence suggests that EDR can be the preferable option as the Silt Density Index (SDI) increases. For very small particle sizes, robust pretreatment can be necessary for RO. It is important to note that the EDR process does not directly filter the treatment stream through the membranes; contaminants are transferred out of the treatment stream and trapped by the membranes. This generally minimizes membrane fouling, decreasing pretreatment requirements in comparison to RO.

Biological Denitrification (BD)

Biological denitrification in potable water treatment is more common in Europe with recent full-scale systems in France, Germany, Austria, Poland, Italy, and Great Britain. To date, full-scale drinking water applications in the United States are limited to a single plant in Coyle, OK (no longer online). However, two full-scale systems are anticipated in California in the next couple of years. Biological denitrification relies on bacteria to transform nitrate to nitrogen gas through reduction. Substrate and nutrient addition is necessary and post-treatment can be more intensive than for the removal processes. Biological denitrification offers the ability to address multiple contaminants and the avoidance of costly waste brine disposal.

Key factors in the consideration of biological denitrification are the chemical requirements, the need for anoxic conditions, the level of operator training, the robustness of the system, and the post-treatment requirements. State regulations are expected to vary and, until more experience with the application of biological denitrification for potable water treatment is obtained in the United States, pilot and demonstration requirements may be intensive. Typically, biological treatment is thought to have a larger footprint; however, with the latest design configurations, the system footprint may be comparable to that of RO or EDR systems.

With reduction of nitrate to nitrogen gas, the lack of a problematic brine waste stream is a clear advantage of biological treatment over the removal processes. Biological treatment has the potential to provide a sustainable nitrate treatment option for the long term. More will be known with the completion of the anticipated full-scale systems in California; cost estimation suggests that biological treatment can be economically competitive with IX.

Chemical Denitrification (CD)

Chemical denitrification uses metals to transform nitrate to other nitrogen species. As an emerging technology, no full-scale chemical denitrification systems have been installed in the United States for nitrate treatment of potable water, and application for nitrate treatment has been strictly limited to bench- and pilot-scale studies. A significant body of research has explored the use of zero valent iron (ZVI) in denitrification. Several patented granular media options have also been developed including sulfur modified iron (SMI) media, granular clay media, and powdered metal media.

Key factors in the consideration of chemical denitrification are the reliability and consistency of nitrate reduction, the lack of full-scale installations, the type of media, and the dependence on temperature and pH. Chemical denitrification has the potential to become a feasible full-scale nitrate treatment alternative, with the advantage of reducing nitrate to other nitrogen species and avoiding the need to dispose of a concentrated waste stream. However, currently this option is an emerging technology in need of additional pilot- and full-scale testing. Due to the potential benefits, further research and optimization of chemical denitrification systems will likely make this a competitive option in the future, especially for multiple contaminants (e.g., arsenic and chromium).

Conclusions

- Current full-scale nitrate treatment installations in the United States consist predominantly of IX and RO. While EDR is a feasible option for nitrate removal from potable water, the application of EDR is generally limited to waters that have high TDS or silica. The use of biological denitrification to address nitrate contamination of drinking water is more common in Europe than in the U.S. However, this option is emerging in the U.S. and two full-scale systems are expected in a few years. Chemical denitrification may become a feasible nitrate treatment option in the future; however, the lack of current full-scale implementation suggests the need for further research, development and testing.

- Brine reuse and treatment are vital to the continued reliance on IX for nitrate treatment of potable water. The low brine technologies offer a minimal waste approach and current research and development of brine treatment alternatives seem to be lighting the path toward future progress.

- In regions with declining water quality and insufficient water quantity, the need to address multiple contaminants will increase in the future, suggesting the future dominance of technologies capable of multiple contaminant removal. In this context, for any individual water source or system, the most appropriate technology will vary with the contaminants requiring mitigation. Although complex, analysis of the optimal treatment option for pairs and groups of contaminants will assist in the treatment design and selection. In such scenarios, the best treatment option for nitrate may not be the most viable overall.

- Currently and into the future, selection of the optimal and most cost-effective potable water treatment options will depend not only on the specific water quality of a given water source, but also on the priorities of a given water system. If land is limited, the typical configuration required for biological treatment may not be feasible. If brine waste disposal options are costly or limited, implementation of denitrification treatment or development of brine recycling and treatment may be the most suitable option.

- When deciding on nitrate treatment, the characteristics of the water system must be taken into account as well. With consideration of economies of scale, many rural small water systems cannot afford to install treatment. Even with financial assistance to cover capital costs, the long term viability of a treatment system can be undermined by O&M costs that are simply not sustainable. For such systems, treatment can become more affordable through consolidation of multiple small water systems into larger combined water systems that can afford treatment as a conglomerate. With a continued decline in water quality, non-treatment options alone, like blending groundwater sources or drilling a new well, may become insufficient measures for a water system to provide an adequate supply of safe and affordable potable water. Especially in rural small communities, perhaps the most promising approach will be consolidation of multiple nearby water systems and the installation of a single centralized treatment plant. Alternatively, separate small treatment facilities can be consolidated under a single agency. For additional discussion on the comparison of alternative water supply options and associated costs see Technical Report 7 (Honeycutt et al. 2012).

- While current cost considerations are commonly the driving force in selecting nitrate treatment, it is essential to consider the long term implications of current industry decisions. For example, it may be cost-effective for a particular system to utilize conventional IX currently, but future water quality changes (e.g., increasing nitrate levels, co-contamination, high salt loading), discharge regulations, or disposal fees may lead to an unmanageable increase in costs. Environmental sustainability in drinking water treatment is being addressed with brine treatment alternatives and denitrification options. It is important to approach the future of drinking water treatment with the mindset that environmental sustainability and economic sustainability are tightly interwoven.

- Centralized treatment may not be feasible for widespread rural communities; another approach to consider is centralized management (e.g., design, purchasing, and maintenance) to minimize costs.

- Point-of-Use (POU) and Point-of-Entry (POE) treatment equipment is an important option to consider, especially for the provision of safe drinking water from private wells. Unless connecting to a nearby public water system becomes an option, users relying on domestic wells have two main alternatives: drilling a new well to attain safe drinking water or installing a POU or POE device for the treatment of contaminated water. The use of POU and POE treatment equipment by small public water systems is currently only a temporary option in California and reliance on these devices for the long-term would require regulation changes. While POU and POE treatment equipment has been shown to effectively address nitrate and other contaminants, it is important to properly maintain these devices to ensure the supply of consistently safe drinking water.

- Within the drinking water community, the options typically considered to address nitrate contamination are IX and RO. Alternative technologies are available or emerging (EDR, BD, CD) because, under some circumstances, they offer advantages over IX and RO. New technologies will continue to be investigated and developed because no single option is ideal for all situations. There is not a nitrate treatment option currently available that can affordably address all possible scenarios. The following diagram is a rough guide for treatment technology selection based on water quality concerns and possible priorities for a given water source or system (Table S.3). This diagram includes generalizations and is not intended to be definitive. In the selection of nitrate treatment technologies the unique needs of an individual water system must be assessed by professional engineers to optimize treatment selection and design.

Table S.3. Comparison of major treatment types.[1]

Concerns	IX	RO	EDR	BD	CD
High Nitrate Removal	🟨	🟨	🟨	🟩	
High TDS Removal	🟥	🟩	🟩	🟥	🟥
Arsenic Removal	🟩	🟩	🟩	🟨	🟩
Radium and Uranium Removal	🟩	🟩	🟩		
Chromium Removal	🟩	🟩	🟩	🟩	🟩
Perchlorate Removal	🟩	🟩	🟩	🟩	

Priorities	IX	RO	EDR	BD	CD
High Hardness Not a Major Concern	🟥	🟥	🟨	🟩	
Reliability	🟩	🟩	🟩	🟨	🟥
Training/ Ease of operation	🟩	🟩	🟩	🟨	
Minimize Capital Cost	🟩	🟨	🟨	🟨	
Minimize Ongoing O&M Cost	🟩	🟨	🟨	🟩	
Minimize Footprint	🟩	🟩	🟩	🟨	
Industry Experience	🟩	🟩	🟨	🟥	🟥
Ease of Waste Management	🟥	🟥	🟨	🟩	🟩

🟩	🟨	🟥	
Good	→	Poor	Unknown (blank)

[1] Ion Exchange (IX), Reverse Osmosis (RO), Electrodialysis Reversal (EDR), Biological Denitrification (BD), Chemical Denitrification (CD). This table offers a generalized comparison and is not intended to be definitive. There are notable exceptions to the above classifications.

1 Introduction

Nitrate contamination of potable water sources is becoming one of the most important water quality concerns in California and across the United States. The maximum contaminant level (MCL), 45 mg/L as nitrate (NO_3^-) (10 mg/L as nitrogen (N)), is currently being approached or exceeded in potable water supply sources at locations throughout the United States (Nolan et al. 2002; Chen et al. 2009; U.S. EPA N.D.). A major source of nitrate contamination is fertilizer. Application of fertilizer in excess of the amount taken up by crops leads to leaching into the groundwater. Leakage from livestock feedlots and waste storage also contributes to the nitrate problem (LLNL 2002). Additional sources include wastewater treatment discharge, faulty septic systems, and various industrial applications. Due to the typical sources, nitrate contamination is more common in rural agricultural areas. The major health concern of nitrate exposure through drinking water is the risk of methemoglobinemia, especially in infants and pregnant women. Due to the nature of the infant digestive system, nitrate is reduced to nitrite which can render hemoglobin unable to carry oxygen (SWRCB 2010).

1.1 Management Options for Nitrate in Potable Water

To meet the nitrate MCL in the provision of potable water, both non-treatment and treatment options are considered. Source management with non-treatment can sometimes provide less costly solutions through wellhead protection, land use management, well abandonment, source modification, development of alternative sources (including consolidation or connection to a nearby system), or blending.[5] The feasibility of non-treatment options can be limited by various factors including location, budget, source availability, and variability of water quality (i.e., fluctuations in nitrate levels), resulting in the need for treatment to remove or reduce nitrate.

Current treatment methods include ion exchange (IX), reverse osmosis (RO), electrodialysis/ electrodialysis reversal (ED/EDR), biological denitrification (BD), and chemical denitrification (CD). These nitrate management options are examined in detail to assess research findings, capital and O&M costs, typical limitations, and the latest improvements. Design and cost considerations will be addressed with the development of guidelines for determining the most appropriate treatment option based on source water quality and other water system characteristics.

Point-of-Entry (POE) and Point-of-Use (POU) treatment equipment should also be considered as part of a comprehensive examination of nitrate treatment. The Safe Drinking Water Act (SDWA) [Section 1412(b)(4)(E)(ii)] (U.S. EPA 1998) identifies both POE and POU treatment units as options for compliance technologies for small systems; California regulations governing the use of POU and POE devices for water system compliance currently restrict their use to a temporary basis and only for systems having particular characteristics (Section 3.7).

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[5] For the purposes of this discussion, blending has been categorized as a “non-treatment” option; however, in practice, blending is sometimes referred to as “treatment.” Treatment options throughout this report refer to treatment technologies available for the removal or reduction of nitrate in drinking water. Blending can sometimes be used to cost-effectively address the nitrate problem through dilution, but has been categorized separately from treatment options for simplicity.

Lastly, hybrid systems are explored. The combination of multiple treatment technologies, including several developing brine treatment alternatives, can maximize the advantages of each option. The goal of this investigation is to provide an overview of management strategies and treatment options, highlighting the most recent advances and elucidating costs and common problems in application.

2 Non-Treatment Options for Nitrate Contaminated Potable Water

2.1 Well Abandonment, Inactivation, and Destruction

With adequate capacity from other sources, the simplest option for management of nitrate contaminated potable water sources is well abandonment and proper destruction. However, the lack of sufficient alternative water supplies often rules out well abandonment as an option. Based on a recent survey conducted by the American Water Works Association (AWWA), 30.4% (17/56) of survey participants with wells impacted by nitrate selected well abandonment as the implemented option for addressing nitrate contamination (Weir & Roberson 2010; Weir & Roberson 2011). It is important to determine the local requirements for safely removing a well from service. For proper abandonment, local requirements can include covering, sealing, and plugging of the well to prevent contamination and to avoid hazardous conditions. Inactivation or abandonment of a well differs from well destruction. Through inactivation or temporary abandonment, the well can be brought back online in the future (e.g., when treatment is installed). In contrast, well destruction involves the filling of a well, making it no longer viable. The costs for proper well abandonment and destruction can be substantial and vary with well depth, diameter, location, and local standards for well destruction. Analysis of public supply well abandonment and destruction in the Tulare Lake Basin and Salinas Valley, as well as across California, is included below in the Section 5.1 Well Abandonment, Destruction, and Inactivation. Additional information on private well abandonment and destruction is provided in Technical Report 2, Section 9 (Viers et al. 2012).

2.2 Wellhead Protection and Land Use Management

While limiting current nitrate contamination of groundwater will not immediately remove the need for treatment, over time, load reduction will minimize source water nitrate levels. Agricultural practices, management of dairies, control of wastewater treatment plant discharge, and monitoring and remediation of septic tank discharges can be improved to minimize nitrogen loading (for a full discussion of source load reduction see Technical Report 3, Dzurella et al. 2012). For example, a project addressing well head protection and land use management performed by the University of Waterloo (Rudolph 2010) successfully decreased groundwater nitrate levels within a two year travel time from 17 to 7 mg/L total stored nitrogen. Reduced nitrogen loading was accomplished by purchasing agricultural land and implementing Best Management Practices (BMPs).

2.3 Development of Alternative Sources and Source Modification

With adequate information about the nitrate distribution and movement in the subsurface, a new well can potentially be developed to access higher quality source water. Due to the anthropogenic nature of the contamination, nitrate concentration typically decreases with depth (Nolan et al. 2002). If suitable water quality exists, drilling a deeper well can remove the need for nitrate treatment. However, the quality improvements must be balanced by a potential decrease in source capacity. Due to drilling and pumping requirements, capital and operational costs increase with the depth to uncontaminated water. When considering the installation of a deeper well to avoid nitrate contamination, it is important to be aware of the risk of encountering other water quality concerns at greater depths (e.g., arsenic) (see Section 4.2 Water Quality - Co-contaminants).

Connecting to a nearby water system that is not impacted by nitrate or to a larger system that can afford nitrate treatment is often the best option for smaller systems. For example, since 1995, the City of Modesto, CA, has been in charge of providing compliant water to the residents of Grayson, using an ion exchange plant for nitrate removal (Scott 2010). Similarly, consolidation of multiple nearby small systems can decrease the cost of treatment per customer to more reasonable levels. Additional alternative source options include purchasing water rights, trucking in potable water, or temporarily relying on bottled water. Reliance on hauled and/or bottled water is only an interim solution for use in emergencies or while an effective compliance option can be implemented. Technical Report 7 provides a comprehensive discussion of alternative water supply options and associated costs (Honeycutt et al. 2012).

Modification of impacted source wells can allow for withdrawal of water with lower nitrate levels by limiting screened intervals to regions of better water quality. Down hole remediation requires characterization of the water quality profile to determine the screening depth range of the higher water quality. Specialized monitoring equipment and techniques are available that can be used without removing pumps (BESST Inc. 2008). With water profile characterization, existing wells can be selectively screened using a packer/plug to limit withdrawal from unwanted regions (Figure 1). The effective application of such well modification techniques is dependent on the subsurface characteristics in the vicinity of the well. Vertical migration of nitrate through the surrounding porous media can lead to increasing nitrate levels in the water withdrawn from the well.

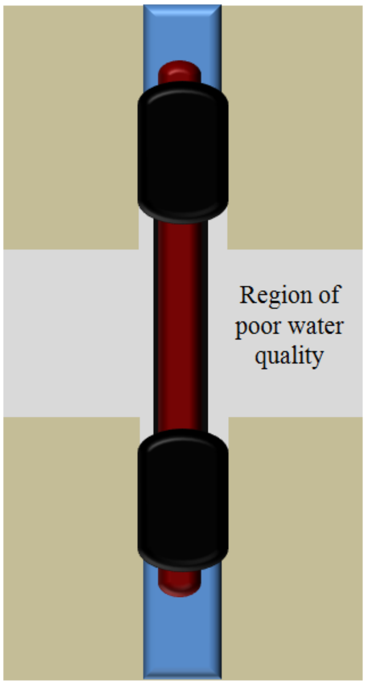

Figure 1. Selective well screening using a packer/plug.

The City of Ceres, CA, is in the process of drilling new wells, in part to avoid the need for nitrate treatment; well modification has also been implemented to avoid water with high nitrate levels (Cannella 2009).

2.4 Blending

The dilution of a nitrate impacted source with an alternate low nitrate source can be a cost-effective option to produce compliant water; this is known as blending and can be applied independently or with treatment.[6] Blending is a common practice for the production of compliant water, but relies on the availability of a low nitrate source and the consistency of nitrate levels to avoid exceedances. High nitrate groundwater can also be blended with surface water when a surface water source is available; however, surface water treatment requirements would increase costs. One drawback of implementing blending to address nitrate contamination is that reliance on blending can limit operational flexibility. If the source used for dilution were compromised, then production would need to be stopped from both wells. Water can also be trucked in for blending purposes when a low nitrate source is unavailable locally; however, hauling water for blending purposes is a temporary solution. Based on the recent AWWA survey, 51.8% (29/56) of respondents with nitrate impacted sources selected blending as the option to address nitrate contamination (Weir & Roberson 2010; Weir & Roberson 2011). Likewise,

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[6] For the purposes of this discussion, blending has been categorized as a “non-treatment” option; however, in practice, blending is sometimes referred to as “treatment.” Treatment options throughout this report refer to treatment technologies available for the removal or reduction of nitrate in drinking water. Blending can sometimes be used to cost-effectively address the nitrate problem through dilution, but has been categorized separately from treatment options for simplicity.

nitrate contamination of drinking water in Germany is often addressed by blending, avoiding the costs of treatment (Dördelmann 2009). When feasible, blending is a simple alternative to treatment that avoids disposal concerns and the certification requirements of treatment (WA DOH 2005). However, disadvantages include the capital investment for accessing an alternative source and monitoring requirements to ensure consistent supply of compliant water (WA DOH 2005). Analysis of water systems utilizing blending to address nitrate contamination in the Tulare Lake Basin and Salinas Valley, as well as across California, is included below in the Section 5.2 Survey of Blending and Treating Systems.

3 Treatment Options for Nitrate Contaminated Potable Water

IX, RO, and ED/EDR transfer nitrate ions from water to a concentrated waste stream that requires disposal. The United States Environmental Protection Agency (U.S. EPA) lists these three processes as accepted potable water treatment methods for nitrate removal (U.S. EPA 2010). In contrast, through biological and chemical denitrification, nitrate is converted to reduced nitrogen species, rather than displaced to a concentrated waste stream that requires disposal.

A survey of nitrate treatment systems was conducted to assess the current state of nitrate treatment. The list of surveyed utilities was developed with the intention of covering a range of utilities with respect to geographic location, treatment type, population size, and residual handling techniques (Table 1). Detailed case studies have been compiled for each of the treatment technologies where full-scale facilities have been in operation or are moving ahead with design. This survey was conducted through collaboration with Jacobs Engineering in the completion of the associated assessment of nitrate treatment alternatives for AWWA and is complemented by a parallel survey of nitrate treatment systems in California. Details from the initial survey are included as examples following a discussion of each of the treatment technologies. Details of the complementary survey of California systems are included in the second half of this report.

A brief comparison of fundamental design considerations, and advantages and disadvantages of the treatment options examined herein is listed in Table 2. It is important to note that the contents of Table 2 are not intended to provide a comprehensive set of criteria for treatment options. Other important criteria in determining the best treatment option, which are site specific and cannot be broadly generalized, include capital and O&M costs, system size (capacity), and system footprint.

IX is the most commonly used nitrate treatment method, with full-scale systems in use throughout the United States. Full-scale application of biological denitrification in potable water treatment is mainly limited to Europe and chemical denitrification methods have been investigated only at the pilot-scale. Others have provided thorough reviews of available nitrate treatment technologies (Kapoor & Viraraghavan 1997; Soares 2000; Shrimali & Singh 2001); however, a recent comprehensive review of the state of nitrate treatment is absent from the literature.

Table 1. Utilities included in the case studies.

Case #	Treatment Type	Location	Capacity (gpm)	Avg. Influent Nitrate mg/L as NO_3^- (mg/L as N)
	Ion Exchange			
1	Conventional ion exchange with blending	California	400	31 – 53 (7 – 12)
2	Conventional ion exchange with blending	California	400	~45 (~10)
3	Counter Current Ion Exchange (MIEX®)	Indian Hills, CO	50	53 – 71 (12 – 16)
4	Multiple vessel ion exchange	California	500 – 900	35 – 89 (8 – 20)
5	Multiple vessel ion exchange	Chino, CA	5000	40 – 200 (9 – 45)
	Reverse Osmosis			
6	Reverse osmosis and blending	Bakersfield, CA	120	75 – 84 (17 – 19)
7	Reverse osmosis, exploring biological reduction	Brighton, CO	4600	49 – 89 (11 – 20)
8	Reverse osmosis and blending	Arlington Desalter, Riverside, CA	4583	44 – 89 (10 – 20)
	Combined Reverse Osmosis and Ion Exchange			
9	Reverse osmosis, ion exchange and blending	Chino Desalter I, Chino, CA	4940 (RO), 3400 (IX)	148 – 303 (33 – 68)
10	Reverse osmosis, ion exchange and blending	Chino Desalter II, Mira Loma, CA	4167 (RO), 2778 (IX)	70 – 224 (16 – 51)
	Electrodialysis/Electrodialysis Reversal/Selective Electrodialysis			
11	Electrodialysis Reversal (EDR)	Spain	3,260 (each, 2 systems)	~80 (~18)
12	Selective Electrodialysis (SED)	Israel	310	84 – 89 (19 – 20)
	Biological Denitrification			
13	Implementing fluidized bed biological reduction	Rialto, CA	2000 – 4000	17 – 19 (~4 – 5)
14	Implementing fixed bed biological reduction	Riverside, CA	1670	44 – 89 (10 – 20)

Table 2. Potable water treatment options for nitrate management (adapted from WA DOH 2005).

	Ion Exchange	Reverse Osmosis	Electrodialysis	Biological Denitrification	Chemical Denitrification
Full-scale Systems	Yes	Yes	Yes	Yes	No
Treatment Type	Removal to waste stream	Removal to waste stream	Removal to waste stream	Biological reduction	Chemical reduction
Common Water Quality Design Considerations	Sulfate, iron, manganese, total suspended solids (TSS), metals (e.g., arsenic), hardness, organic matter	Turbidity, iron, manganese, SDI, particle size, TSS, hardness, organic matter, metals (e.g., arsenic)	Turbidity, iron, manganese, TSS, hydrogen sulfide, hardness, metals (e.g., arsenic)	Temperature and pH, anoxic conditions	Temperature and pH
Pretreatment Needs	Pre-filter, address hardness	Pre-filter, address hardness	Pre-filter, address hardness	pH adjustment, nutrient and substrate addition, need for anoxic conditions	pH adjustment
Post-treatment Needs	pH adjustment	pH adjustment Remineralization	pH adjustment Remineralization	Filtration, disinfection, possible substrate adsorption	pH adjustment, iron removal, potential ammonia control
Waste/Residuals Management	Waste brine	Concentrate	Concentrate	Sludge/biosolids	Waste media, Iron sludge
Start-up Time	Minutes	Minutes	Minutes	Initial plant startup: Days to weeks After reaching steady state: Minutes	Minutes
Water Recovery	Conventional (97%) Low brine (Up to 99.9%)	Up to 85%	Up to 95%	Nearly 100%	Not demonstrated full-scale
Advantages	Nitrate selective resins, common application, multiple contaminant removal	Multiple contaminant removal, desalination (TDS removal)	Multiple contaminant removal, higher water recovery (less waste), desalination, unaffected by silica	No waste brine or concentrate, nitrate reduction rather than transfer to a waste stream, high water recovery, and potential for multiple contaminant removal	No waste brine or concentrate, nitrate reduction rather than transfer to a waste stream, and potential for multiple contaminant removal
Disadvantages	Potential for nitrate peaking, high chemical use (salt), brine waste disposal, potential for disinfection byproduct (DBP) formation (e.g., NDMA)	Membrane fouling and scaling, lower water recovery, operational complexity, energy demands, waste disposal	Energy demands, operational complexity, waste disposal	Substrate addition, potentially more complex, high monitoring needs, possible sensitivity to environmental conditions, risk of nitrite formation (potential incomplete denitrification), post-treatment to address turbidity standards and 4-log virus removal (state dependent)	Inconsistency of nitrate reduction, risk of nitrite formation (potential incomplete denitrification), reduction to ammonia, lack of full-scale systems, pH and temperature dependence, possible need for iron removal

3.1 Ion Exchange (IX)

As the most commonly used method for the removal of nitrate in potable water treatment, IX has been widely researched, with numerous full-scale installations in operation. With the potential for multiple contaminant removal, IX can also be used to address other water quality concerns including arsenic, perchlorate, selenium, chromium (total and chromium-6), and uranium (AWWA 1990; Boodoo 2004). Selected IX installations used for nitrate treatment in the United States are listed in Table 3.

Table 3. Selection of full-scale ion exchange installations for nitrate removal.

Locations	Year Installed	Influent nitrate (mg/L as NO_3^-)	Capacity (MGD)	Reference
Ellsworth, MN	1994	-	0.047	MN Dept. of Ag. (N.D.)
Clear Lake, MN	1995	-	0.047	MN Dept. of Ag. (N.D.)
Adrian, MN	1998	-	0.129	MN Dept. of Ag. (N.D.)
Edgerton, MN	2002	-	0.137	MN Dept. of Ag. (N.D.)
McCook, NE	2006	Up to 125	6.8	Contaminant Removal News (2007)
McFarland, CA, Well 2	1983	60	1	Guter (1995), See also Guter (1982)
McFarland, CA, Well 4	1987	64	1	Guter (1995), See also Guter (1982)
La Crescenta, CA	1987	70 – 100	2.7	Guter (1995)
Grover City, CA, 3 wells	-	80 – 130	2.3	Guter (1995)
Des Moines, IA	1992	Up to 55	10	Des Moines Water Works, Rash (1992)
Glendale, AZ: full-scale pilot	2010	Spiked up to 177	10	Meyer et al. (2010), See also Clifford et al. (1987)
Indian Hills, CO	2009	53 – 71	0.072 (design)	See Case Study

Due to its common application, the investigation of IX for the removal of nitrate is prevalent in the literature (Yoon et al. 2001; Chabani et al. 2006; Samatya et al. 2006; Clifford 2007; Meyer et al. 2010; Clifford et al. 2010). Kapoor & Viraraghavan (1997) provide an extensive review of IX research up to 1997. Modifications of conventional IX have led to the emergence of more efficient IX processes including multiple vessel configurations, counter current configurations, the use of specialized resins, improved hydraulics, and weak base anion exchange (WBA IX).

3.1.1 Conventional Ion Exchange

Conventional IX utilizes a strong base anion (SBA) exchange resin. In accordance with Figure 2, raw water passes through pretreatment to remove suspended solids and to address other constituents capable of fouling the resin. The nitrate laden treatment stream then enters the ion exchange vessel. Upon contacting the resin, nitrate displaces chloride at surface sites, removing nitrate from the water.

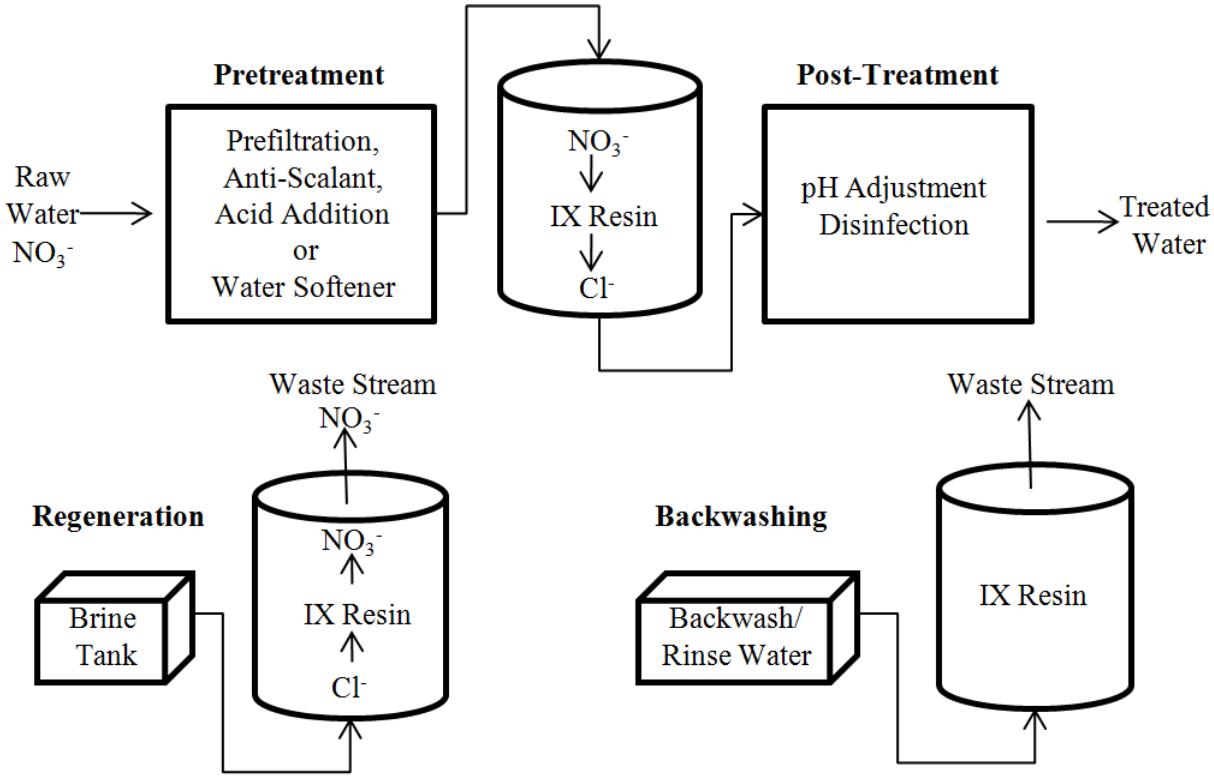

Figure 2. Conventional ion exchange schematic.

This technique is similar to a water softener, which replaces the divalent cations of hard water (Mg^{2+} and Ca^{2+}) with a monovalent cation (Na^+). Eqn. 1 depicts the transfer of ions, with R as the resin surface site. Leaving nitrate behind, treated water exits the ion exchange vessel and passes on to post-treatment for stabilization and disinfection.

$$R\text{-}Cl + NO_3^- \rightarrow R\text{-}NO_3 + Cl^- \qquad \text{(Eqn. 1)}$$

To prevent nitrate breakthrough, regeneration is necessary when the resin is exhausted of chloride ions (chloride has been displaced at the majority of surface sites). The media is backwashed with a high salt solution (0.5 – 3 M, Clifford 2007) to reverse the process, resulting in a brine waste stream high in nitrate and other concentrated ions (Eqn. 2).

$$R\text{-}NO_3 + Cl^- \rightarrow R\text{-}Cl + NO_3^- \qquad \text{(Eqn. 2)}$$

The relative affinity of common anions for conventional anion exchange resin is $SO_4^{2-} > NO_3^- > Cl^- > HCO_3^-$ (Bae et al. 2002; Clifford et al. 2010). If generic resins are not regenerated soon enough, sulfate displacement of nitrate in the resin can lead to nitrate release from the resin to the treatment stream (Eqn. 3). This is known as nitrate dumping, nitrate peaking, or chromatographic peaking and is further discussed below.

$$2R\text{-} NO_3 + SO_4^{-2} \rightarrow R_2\text{-}SO_4 + 2NO_3^- \qquad \text{(Eqn. 3)}$$

Due to the stronger affinity of the sulfate ion for generic anion exchange resins, nitrate selective resins have been developed for which the order of affinity is $NO_3^- > SO_4^{2-} > Cl^- > HCO_3^-$ (Guter 1982; Guter 1995). Important factors in resin selection are the exchange capacity[7] and selectivity coefficient[8] of the resin and the rate of ion transfer (kinetics[9]).

Detailed case studies of conventional IX plants are included in Section 3.1.7 Ion Exchange - Case Studies.

3.1.2 Ion Exchange - Design Considerations

Various tools are available to assist with IX system design including Dow's CADIX (Computer Assisted Design for Ion Exchange) (Dow 2010b) and Lenntech's Ion Exchange calculator (Lenntech 2009b.) Table 4 summarizes key design considerations in the application of conventional IX to nitrate removal from potable water.

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[7] Exchange capacity: The exchange capacity is a measure of how many ions the resin can capture per unit volume.

[8] Selectivity coefficient: The selectivity coefficient of a resin refers to the relative affinity of resin surface sites for a particular ion, in this case nitrate.

[9] Kinetics: The term kinetics refers to the rate of a reaction. The rate that nitrate displaces chloride on the resin is important for efficient treatment and can be affected by competing ions.

Table 4. Summary of design considerations for conventional IX.

Resin Selection	• Generic SBA resins for maximum exchange capacity (for low sulfate) o Less expensive than nitrate selective resins o Less frequent regeneration due to higher capacity (in the absence of co-contaminants) o Nitrate dumping potential • Nitrate selective resins to avoid nitrate dumping (for high sulfate) o More expensive than generic resins o Longer bed life o More nitrate removed per unit of waste brine
Pretreatment	• Filtration to remove iron, manganese, TSS, and organic matter to prevent resin fouling • Water softening (anti-scalant, acid, or water softener) to prevent scaling • Dechlorination to prevent resin oxidation[10]
Post-Treatment	• Chloride : alkalinity ratio and dezincification[11] • Chloride : sulfate ratio and galvanic corrosion[12] • Potential pH adjustment and restoration of buffering capacity to avoid corrosion
Chemical Usage	• pH adjustment (caustic soda or soda ash) • Regenerant brine, salt consumption
O&M	• Frequency of regeneration depends on water quality and resin type • Fresh brine preparation and waste disposal • Resin loss and replacement: 3 – 8 year lifetime (WA DOH 2005; Dow 2010c) • Continuous or frequent monitoring of nitrate levels • Backwashing to dislodge solids
System Components	• Fixed bed versus continuous regeneration • Key system configuration parameters are system flow rate, bed swelling, bed depth, backwash flow rate, and rinse requirements o Vessels in parallel or in-series o Co-current or counter-current regeneration
Waste Management and Disposal	• Significant cost of waste brine disposal is of greatest concern for inland systems • Close proximity to coastal waters is beneficial for brine disposal • Management options can include sewer or septic system, drying beds, trucking off-site, coastal pipeline, deep well injection, and advanced treatment • Disposal options can be limited by waste brine water quality (e.g., volume, salinity, metals, and radionuclides) • Optimization of recycling and treatment of waste brine is desirable
Limitations	• Need to manage resin fouling o Hardness, iron, manganese, suspended solids, organic matter, and chlorine • Competing ions (especially sulfate) • Disposal of waste brine • Possible role of resin residuals in DBP formation

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[10] The resin can be degraded by oxidation; the functional amine groups on the resin surface are susceptible to oxidation which can lead to diminished capacity (Dow 2010d).

[11] As nitrate and other anions displace chloride on the resin, chloride is released to the product water, leading to the potential for taste issues and dezincification (Kapoor & Viraraghavan 1997). Dezincification refers to the ability of product water to dissolve zinc from brass and is dependent on the ratio of chloride to alkalinity (> 0.5 can be problematic). By restoring alkalinity, the dezincification potential can be minimized.

[12] Galvanic corrosion can result in the release of lead from brass and galvanized solder-copper connections and is associated with a high ratio of chloride to sulfate (> 0.58 can be problematic) (Edwards et al. 1999; Edwards & Triantafyllidou 2007).

Water Quality

Raw water quality is a key factor in the efficiency of an IX system, impacting resin selection, pretreatment and post-treatment needs, regeneration efficiency, chemical usage, and waste disposal. Important water quality parameters include alkalinity, hardness, iron, manganese, and potential competing ions (predominantly sulfate).

Selection of the appropriate resin for a given system depends directly on source water quality. In the presence of high levels of co-contaminants, nitrate selective resins may be necessary rather than generic resins. Both strong base anion exchange (SBA) resin and weak base anion exchange (WBA) resin can be suitable for nitrate removal from potable water. The latter will be addressed separately. The two standard types of SBA resins deviate in their functional groups. Anion exchange is dependent on the trimethylamine groups of the SBA Type I resin and the dimethylethanolamine groups of the SBA Type II resin (Helfferich 1995).

Nitrate selective resin was invented in the early 1980’s by Gerald Guter (Guter 1982, see related patent: Guter 1983). Clifford & Weber (1978 and 1983) contributed to the development and characterization of these resins with their research on resin selectivity (Clifford et al. 2010). Nitrate selective resins rely on different functional groups than Type I and Type II SBA resins. Nitrate selectivity is attributed to the increased hydrophobicity and site spacing of exchange sites due to the triethyl, tripropyl, and tributyl functional groups of nitrate selective resin (Clifford & Weber 1978; Guter 1982; Clifford et al. 2010). The use of nitrate selective resin avoids the problem of nitrate peaking (i.e., nitrate dumping or chromatographic peaking), typically caused by the greater affinity of generic resins for sulfate.

It is important to note the distinction between nitrate peaking and nitrate breakthrough. As nitrate displaces chloride on the resin, the nitrate capacity of the resin is gradually exhausted leading to increasing effluent nitrate levels until influent levels are reached. This is known as breakthrough and can occur regardless of resin type. Nitrate peaking can also occur upon exhaustion of the resin capacity for nitrate. However, with nitrate peaking, the nitrate on the resin is displaced by sulfate, thereby increasing the effluent nitrate concentration to levels above that of the raw water (Clifford et al. 2010). The peak in nitrate concentration is due to the combination of the influent nitrate ions and the nitrate ions that are coming off of the resin via sulfate displacement.

Under low sulfate conditions, the use of generic SBA resins is preferred, due to their larger exchange capacity. As the ratio of sulfate to nitrate increases, the use of nitrate selective resins avoids possible nitrate peaking, minimizing the risk of MCL exceedance and the need for more frequent regeneration. However, with the highest nitrate selectivity, regeneration efficiency can decrease, increasing chemical use. With a stronger affinity for nitrate, the removal of nitrate from the resin in regeneration is more difficult because nitrate is more strongly bound (Dow 2010c). Nitrate selective resins are more costly than the generic option (Water Quality Products 2003), but under appropriate conditions the use of regenerant can be minimized and bed life can be increased significantly with their use. Different system configurations have been implemented as an alternative to the use of nitrate selective resins to address the problem of co-contaminants and the risk of nitrate peaking (Clifford et al. 2010), for more information, see System Layout and Site Considerations, below. In the consideration of IX for multiple contaminant removal, the selection of the most appropriate resin depends in part on the type and concentration of co-contaminants. Modeling and column or pilot testing is important to determine appropriate design parameters and to design treatment based on a full life cycle analysis of costs.

Upstream of the IX vessels, pretreatment of the source water may be necessary to avoid resin fouling (Water Quality Products 2003; WA DOH 2005). Potential constituents of concern include organic matter, turbidity, total suspended solids (TSS), sand and metals, primarily iron and manganese (Kapoor & Viraraghavan 1997; Water Quality Products 2003; WA DOH 2005). Pre-filtration is typically used to remove these constituents; however, additional pretreatment methods may be used. For example, coagulation/flocculation and filtration may be necessary for surface waters. Pretreatment may be needed for waters with a total concentration of metals (e.g., iron and manganese) above 0.1 mg/L (Ten States Standards 2007). Hard, alkaline water can lead to resin scaling, due to calcium and magnesium accumulation; pH adjustment or water softening can be used to prevent resin scaling (Water Quality Products 2003). Created by Wolfgang Holl, the carbon dioxide regeneration of ion exchange (CARIX) process combines anion and cation exchange (Holl 1995). The CARIX process enables simultaneous removal of cations (for hardness reduction) and anions through the combination of a weak acid cation exchanger and a strong base anion exchanger.

Resin exposure to disinfectants (chlorine and chloramines) should be avoided to prevent resin oxidation and the possible release of disinfection byproducts, specifically nitrosamines (Kemper et al. 2009). Disinfection byproducts (DBPs) are potentially cancer-causing compounds that can be formed through a reaction of disinfection chemicals like chlorine and chloramines with organic matter. Due to the amine functional groups of anion exchange resins, "these resins may serve as precursors for nitrogenous disinfection byproducts, such as nitrosamines, nitramines, and halonitromethanes" (Kemper et al. 2009, p. 466). Recent research suggests that the risk of DBP formation is higher with the use of new resin; however, precursors can be a problem with downstream chloramine use or with upstream disinfection (see Kemper et al. 2009 in Table A.1 of the Appendix). Magnetic ion exchange resin is an exception as its primary purpose is to remove organic carbon and limit DBP formation (Boyer & Singer 2006) (see the MIEX® process in Section 3.1.7 Ion Exchange - Case Studies).

IX can reduce alkalinity and pH due to removal of bicarbonate. To prevent corrosion in downstream pipes, the product water pH and buffering capacity may need to be increased.[13] Soda ash can be added to the regenerant brine to load a portion of the resin sites with bicarbonate rather than chloride. This can restore some alkalinity to the water as bicarbonate is released from the resin when displaced by nitrate and other anions in the treatment stream (Water Quality Products 2003). To minimize the need for caustic addition in post-treatment, an upstream atmospheric degasifier for carbon dioxide removal can be added during pretreatment (Dow 2010).

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[13] It is important to note the relationship between alkalinity and pH. Alkalinity is a measure of buffering capacity or the resistance to changes in pH. Demineralized water or water with a low buffering capacity will be susceptible to more dramatic pH changes and is considered unstable. The pH of acidic product water should be adjusted and the buffering capacity of demineralized product water should be restored to avoid corrosion downstream and the potential for lead and copper challenges.

As nitrate and other anions displace chloride on the resin, chloride is released to the product water, leading to the potential for taste issues, dezincification, and galvanic corrosion. Dezincification refers to the ability of product water to dissolve zinc from brass and is dependent on the ratio of chloride to alkalinity (as $CaCO_3$)(> 0.5 can be problematic, according to Kapoor & Viraraghavan (1997)). By restoring alkalinity, the dezincification potential can be minimized. Galvanic corrosion can result in the release of lead from brass and galvanized solder-copper connections and is associated with a high ratio of chloride to sulfate (> 0.58 can be problematic) (Edwards et al. 1999; Edwards & Triantafyllidou 2007). It is important to consider the potential downstream impact of subtle water quality changes caused by treatment.

System Layout and Site Considerations

The IX system can be operated using a fixed bed or as a continuous system. Details of modifications to conventional fixed bed systems are provided below. Key parameters in vessel sizing and system configuration are system flow rate, bed swelling, bed depth, backwash flow rate, and rinse requirements (Dow 2010). Regeneration can be designed in a co-current or counter-current configuration. Vessels can be operated in parallel or in-series for redundancy, to maximize removal efficiency per regeneration cycle, to address nitrate peaking and to consistently produce water with limited variation in water quality parameters (Clifford et al. 2010).

Residuals Management and Disposal

Management of waste brine can be costly. Options include discharge to a sewer or septic system, waste volume reduction using drying beds, trucking to an off-site approved disposal location, ocean discharge through a coastal pipeline, deep well injection, and advanced treatment. Water quality characteristics of the waste brine (e.g., volume, salinity, metals, and radionuclides) can affect the feasibility and costs of disposal options. Proximity and access to coastal waters can be a significant factor in determining the burden of brine disposal. Generally, disposal is of greatest concern to inland communities. Although other removal technologies (RO and ED) also require concentrate disposal, this is an issue of particular concern with IX. Because IX requires the addition of salt for resin regeneration, the waste stream consists of not only the nitrate and other ions that have been removed from the water, but also the spent brine solution used in regeneration. The high cost of nitrate laden brine disposal has led to research into optimization of recycling and treatment of this waste stream. Partial regeneration and reuse of treated brine for multiple regeneration cycles can minimize the volume of waste while maximizing regeneration efficiency (Clifford 2007; Clifford et al. 2010). Application of IX systems coupled with biological, chemical, or catalytic denitrification enables removal of nitrate from waste brine, with reduction to nitrogen gas. The electrochemical destruction of nitrate in waste brine is also being explored. Several combined configurations of interest are discussed below in Section 3.6 Brine Treatment Alternatives and Hybrid Treatment Systems.

Maintenance, Monitoring, and Operational Complexity

Regeneration frequency will depend on pretreatment measures, water quality, and the type of resin used (WA DOH 2005). The typical amount of brine waste compared to water produced can range from ~3.0% for conventional systems (Clifford 2007) to ~0.5% for low brine systems (Calgon Carbon Corporation 2007, discussed below). A constant supply of brine must be available for resin regeneration and waste brine requires appropriate storage and disposal. Resin loss can be controlled by adjusting the backwash flow rate and adding screens (Keller 2000). Resin life will also depend on water quality and pretreatment measures; resin replacement may be required after 3 – 8 years (WA DOH 2005; Dow 2010c). To ensure the production of compliant water, continuous or frequent monitoring of nitrate levels is necessary. In addition to resin regeneration, backwashing is used to dislodge solids and “resin fines” (Dow 2010). In comparison with alternative treatment options, IX requires limited O&M with high feasibility of automation and low operational complexity.

3.1.3 Ion Exchange - Cost Considerations

For optimal operation of an IX system, the fundamental objective is to maximize regeneration efficiency, while meeting necessary potable water guidelines. Factors affecting system cost include facility size (flow rate), source water quality (including nitrate concentration), environmental factors (temperature), and target effluent nitrate concentration. Disposal of waste brine is commonly a significant portion of O&M costs.

Capital costs for IX include land, housing, piping, storage tanks, O&M equipment, vessels, resin, preliminary testing (pilot studies), permits, and training. O&M costs include resin replacement (due to loss or degradation), resin disposal, brine disposal or treatment, chemical use (salt, anti-scalant, pH adjustment), repair and maintenance, power, and labor.

Published cost information from existing IX installations is listed in Table 5. Costs have been adjusted to 2010 dollars, unless indicated otherwise. Costs can be difficult to assess due to inconsistencies in how cost information is reported. Comparison of IX costs is not always valid due to differences in influent water quality parameters, system size, waste management options, and system configuration. Published costs do not always include comparable information. It would be inappropriate to compare the O&M costs of a facility that excludes disposal costs with others that include this information. The listed cost information is provided as an approximate range of costs for specific facilities. Costs for implementing IX may be very different from those listed here. A thorough cost analysis of design parameters for specific locations would be required for accurate cost estimation. The information gathered through the questionnaire includes detailed costs associated with the unique case studies included in this analysis. A detailed discussion of treatment costs is included below in Section 6 Treatment Cost Analysis.

Table 5. Selected published costs* of ion exchange systems for nitrate removal.

System Flow**	< 0.5 MGD	0.5 – 5 MGD	5+ MGD
Annualized Capital Cost ($/1000 gal)	0.37 – 1.21 [1]	0.28 – 0.94 [2, 3]	0.28 – 0.61 [3, 4, 5]
O&M Cost ($/1000 gal)	0.60 – 4.65 [1]	0.46 – 1.25 [2, 3]	0.37 – 0.87 [3, 4, 5]
Total Annualized Cost ($/1000 gal)	0.97 – 5.71 [1]	0.74 – 2.19 [2, 3]	0.65 – 1.44 [3, 4, 5]

*Costs have been adjusted to 2010 dollars with 7% interest over 20 years, unless indicated otherwise (below).
**When available, costs are based on actual system flow rather than design capacity.
[1] Minnesota Department of Agriculture (N.D.), not adjusted to 2010 dollars, 20 year amortization without interest. [2] Guter (1995). [3] Conlon et al. (1995). [4] Meyer et al. (2010). [5] Drewry (2010).

3.1.4 Ion Exchange - Selected Research

A large body of research has focused on IX. Table A.1 of the Appendix is a list of recent studies relevant to nitrate removal from potable water. Given the many years of successful full-scale operating experience with IX, current applied research is focused on brine recycling efficiency, the optimization of waste management, and improvements in resin capacity and selectivity, to improve efficiency and reduce costs.

3.1.5 Ion Exchange - Summary of Advantages and Disadvantages

A summary of advantages and disadvantages of IX in comparison with other treatment options is listed in Table A.6 of the Appendix. Significant advantages of IX include years of industry experience, multiple contaminant removal, selective nitrate removal, financial feasibility, use in small and large systems, and the ability to automate. Disadvantages include the costly disposal of waste brine, the potential for nitrate dumping and resin fouling, the possible need for pH adjustment, the potential for hazardous waste generation (i.e., brine with traces of co-contaminants like arsenic and chromium), and the possible role of resin residuals in DBP formation (Kemper et al. 2009).

3.1.6 Modifications to Conventional Ion Exchange

Modifications of conventional IX have led to the emergence of low-brine IX processes including magnetic ion exchange (MIEX®), ion exchange separation (ISEP®), Envirogen (formerly Basin Water) systems, and weak base ion exchange (WBA IX). Despite their potential advantages, it is important to note that proprietary technologies may have inherent disadvantages, such as a lack of flexibility to use better technology when it becomes available, vulnerability if the manufacturer goes out of business or discontinues supporting the product, and a lack of competition to keep O&M costs down.

Counter Current Flow with Specialized Resin

Magnetic ion exchange technology (MIEX®), developed by Orica Watercare, offers a low brine alternative to conventional IX, using a unique SBA Type I resin. The MIEX® process (Figure 3) differs from conventional IX in that the resin is fluidized in a contactor with spent resin removed from the contactor for regeneration outside of the process water stream and then returned to the contactor. This is in contrast to conventional IX, where the resin is stationary. Unlike conventional IX, the MIEX® fluidized bed process is tolerant of suspended solids and low levels of oxidants.

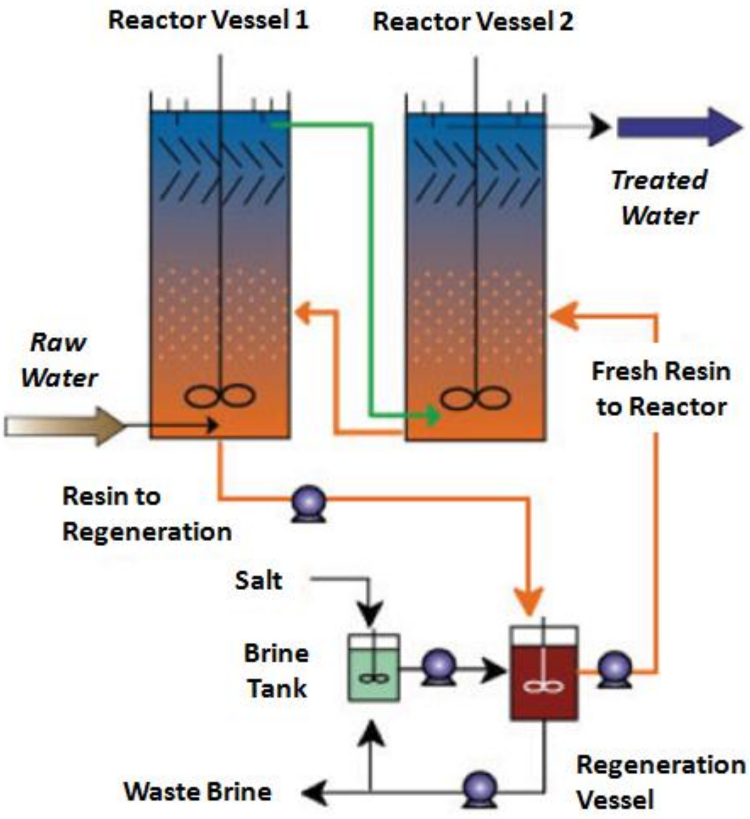

Figure 3. Process flow diagram for counter current MIEX® process. (Source: reprinted with permission, Orica Watercare 2008b.)

The minimization of waste brine is accomplished through frequent batch regeneration. The magnetized resin encourages agglomeration of loaded resin particles, resulting in faster settling. Loaded resin is removed from the bottom of the IX vessel and is passed to regeneration, while regenerated resin is added at the top of the IX vessel. This configuration removes the risk of nitrate peaking because clean resin, added at the end point of the system, captures any displaced nitrate, while competing ions, such as sulfate can be removed early on in the resin vessel. The MIEX® process has been proven to effectively address various water quality concerns including nitrate, organic carbon, arsenic, chromium-6 and perchlorate (Seidel et al. 2004; Humbert et al. 2005; Boyer & Singer 2006; and Watercare 2008).

A detailed case study of a MIEX® treatment plant in Indian Hills, CO is included in Section 3.1.7 Ion Exchange - Case Studies.

Improved Hydraulics and Nitrate Selective Resins

The Layne Christensen Company and Rohm and Haas offer their Advanced Amberpack® system which utilizes nitrate selective resins and their patented Fractal Distribution Technology to increase the treated water volume and decrease the waste brine. Nitrate selective resins have a greater affinity for nitrate than non-selective resins, resulting in an improvement in removal efficiency, especially in waters where the sulfate to nitrate ratio is greater than one.

The distributor system is designed to provide uniform flow through the ion exchange vessels in both treatment and regeneration modes. As a result of the uniform flow, the exchange capacity of the resin can be maximized while the brine and rinse values can be minimized, thus increasing the water efficiency of the system (Rohm and Haas Company 2007).

Multiple Vessel Carousel Configuration

Calgon Carbon's Continuous Ion Exchange Separation System (ISEP® System) utilizes a carousel configuration. This configuration has the potential to avoid downtime for regeneration, minimize the amount of resin needed, and maximize regeneration efficiency. Illustrated in Figure 4, multiple resin vessels are rotated from active treatment to resin regeneration and rinsing and back to active treatment. The vessels rotate in the opposite direction of the water movement (Figure 5). The counter-current and counter-flow system can provide consistent product water, operating uninterrupted with up-flow regeneration and down-flow treatment. This configuration results in four zones within the system (Calgon Carbon Corporation ISEP® for Nitrate Removal Brochure 2003). In the Adsorption Zone, the feed stream is passed through 14 ports in parallel for single pass flow. Nitrate and other anions are removed from the feed water as they transfer to the resin. In the Displacement Zone, softened water is used to displace the hard feed water to prevent scale build-up in the regeneration zone. In the Regeneration Zone, a combination of brine and rinse is directed through the beds for a true counter-current regeneration to maximize the regeneration efficiency. In the Rinse Zone, a small amount of softened feed water is used to prevent any of the salt from the regeneration zone from reaching the product water.

A detailed case study of an ISEP® treatment plant in Chino, CA is included in Section 3.1.7 Ion Exchange - Case Studies

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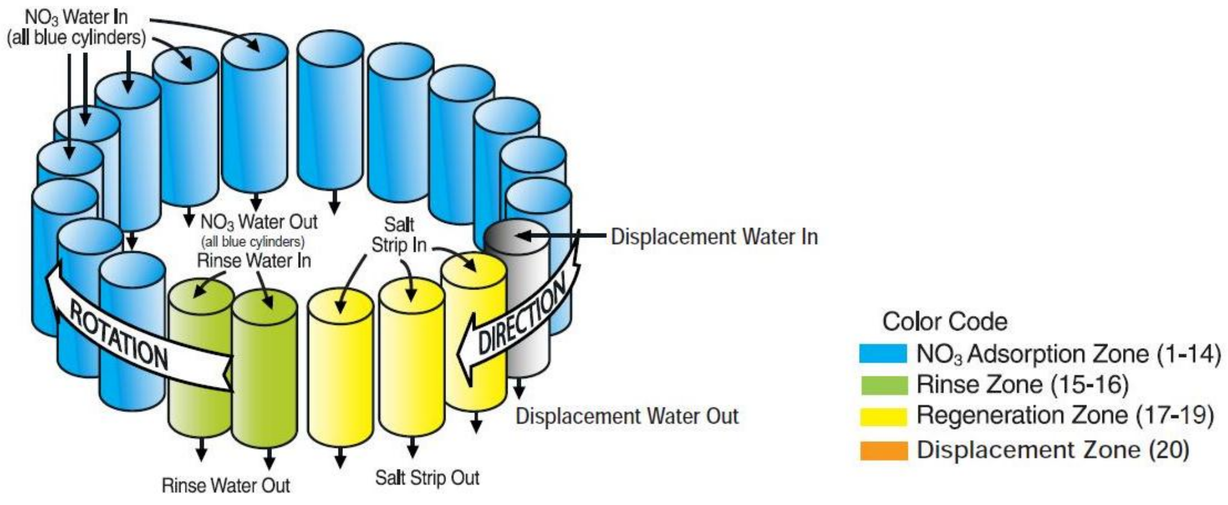

Figure 4. Vessel rotation in Calgon Carbon countercurrent ISEP® system. (Source: Calgon Carbon Corporation 2003.)

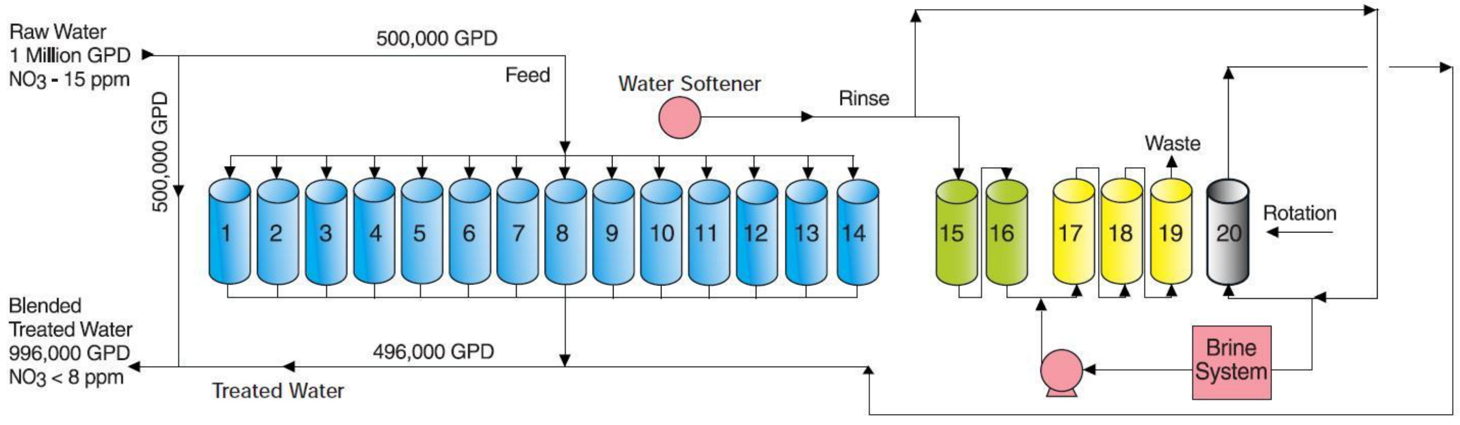

Figure 5. Example of flow through the ISEP® system. (Source: Calgon Carbon Corporation 2003.)

Multiple Vessel Staggered Configuration

Envirogen Technologies, Inc. offers proprietary ion exchange (IX) systems using multiple beds operated in a staggered design (Figure 6). Such a configuration has the potential to maximize resin capacity and minimize waste and chemical use.

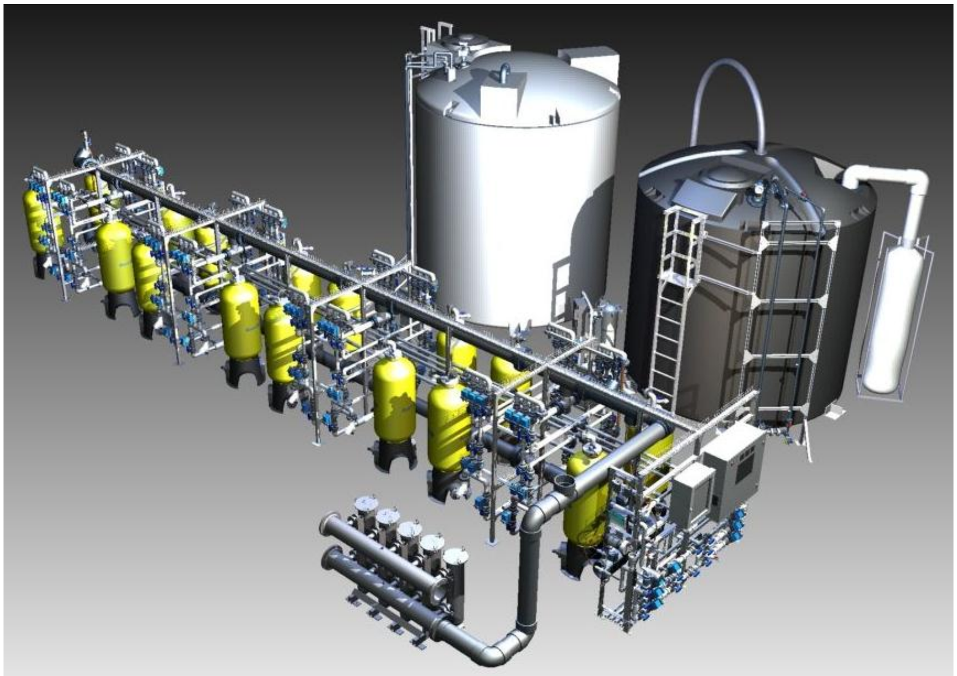

Figure 6. Example of an Envirogen multiple bed proprietary anion exchange system. (Source: reprinted with permission, Envirogen 2009.)

Envirogen's low brine IX system has been successfully implemented for nitrate and uranium removal in San Bernardino County, CA (Envirogen 2010). Delivering 2 MGD with nitrate levels reduced from 50 – 60 mg/L as nitrate (11 – 14 mg/L as N) to 35 mg/L as nitrate (8 mg/L as N) and a system footprint of 50' X 50', the installation resulted in recovery of a well that had been previously decommissioned. Envirogen is contracted to handle all operation and maintenance, including waste disposal.

In Yucca Valley, CA, nitrate treatment was required due to over pumping and the resulting intrusion of septic system contaminated waters. An Envirogen IX system was installed to provide 2.8 MGD. With treatment, nitrate concentrations are decreased from 58 mg/L as nitrate (13 mg/L as N) to 20 mg/L as nitrate (4.5 mg/L as N) with 50% blending and a 0.3% waste rate (0.15% net) (Envirogen 2002).

A detailed case study of an Envirogen low-brine IX treatment plant in California is included in Section 3.1.7 Ion Exchange - Case Studies.

Weak Base Anion Exchange (WBA IX)

Weak base anion exchange (WBA IX), an emerging technology, can be an effective option for nitrate removal from potable water. Highly pH dependent, nitrate removal using WBA IX is governed by Eqn. 5 (Applied Research Associates N.D.). While SBA IX resin can remove nitrate by splitting nitrate salts, WBA IX resin removes strong acids (Remco Engineering N.D.). First, acid addition protonates the WBA resin (Eqn. 4). Next, the positively charged resin sites remove anions, like nitrate, from the treatment stream (Eqn. 5).

$$R\text{-}NH_2 + H^+ \rightarrow R\text{-}NH_3^+ \qquad \text{(Eqn. 4)}$$

$$R\text{-}NH_3^+ + NO_3^- \rightarrow R\text{-}NH_3\text{-}NO_3 \qquad \text{(Eqn. 5)}$$

Resin regeneration occurs by neutralizing the resin, in accordance with Eqn. 6. Rather than the high salt solution used to regenerate SBA resins, weak bases are used to neutralize the WBA resin.

$$R\text{-}NH_3\text{-}NO_3 + Na^+OH^- \rightarrow R\text{-}NH_2 + HOH + Na^+NO_3^- \qquad \text{(Eqn. 6)}$$

As discussed in previous sections, the use of SBA IX resin results in a high-nitrate brine waste stream. Due to the high salt content, the nitrate in the waste stream generally cannot be beneficially reused. However, with the use of alternative weak bases for regeneration (Eqns. 7 and 8), the waste stream does not have a high salt content and could potentially be recycled as fertilizer (NH_4NO_3 and $Ca(NO_3)_2$) (Clifford 2007).

$$R_3N\text{-}HNO_3 + NH_4OH \rightarrow R_3N\text{-}HOH + NH_4NO_3 \qquad \text{(Eqn. 7)}$$

$$2\ R_3N\text{-}HNO_3 + Ca(OH)_2 \rightarrow 2R_3N\text{-}HOH + Ca(NO_3)_2 \qquad \text{(Eqn. 8)}$$

The use of WBA resins is more operationally complex than the use of SBA resins. Chemical use includes acids and bases, the system is susceptible to corrosion, and pH adjustments are more significant. Adjustment of influent pH to between 3 and 6 is necessary, with subsequent product water pH adjustment as well (Clifford 2007). WBA resins can also be more sensitive to temperature, with one resin having a maximum operating level of 95°C (Dow 2010), but this should not impact their use with municipal drinking water treatment. Regeneration of WBA resin is more efficient than that of the SBA resin of conventional IX; regenerant waste volumes are minimized and waste products can sometimes be recycled as fertilizer (Clifford 2007 and ARA & Purolite N.D.a).

Weak Base Ion Exchange for Nitrate, or the “WIN” Process, was developed by Applied Research Associates, Inc. (ARA) and The Purolite Company (ARA & Purolite N.D.b) as a treatment option that can be less costly and more efficient, with significantly lower waste volumes, than conventional SBA IX (Figure 7).

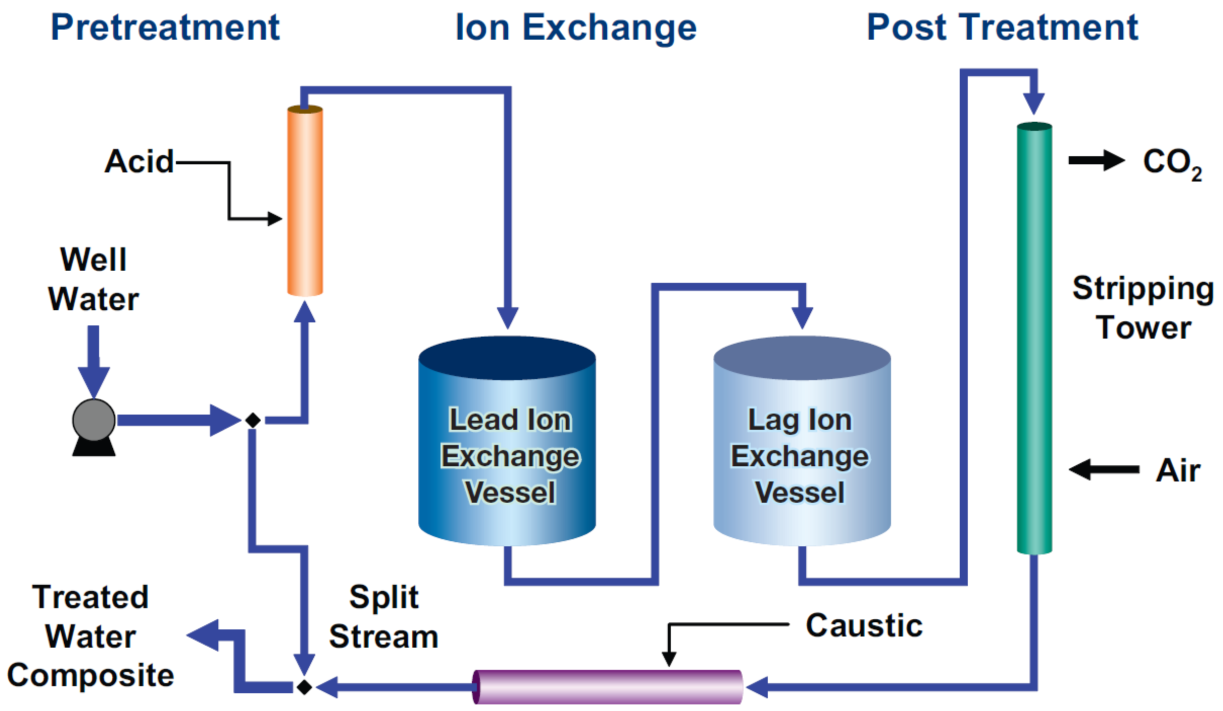

Figure 7. Process schematic of weak base ion exchange for nitrate ("WIN" process). (Source: reprinted with permission, ARA & Purolite N.D.b.)

As an emerging technology, available information is limited to that provided by the manufacturers. The process consists of a pretreatment step to decrease pH, followed by ion exchange vessels in series, and post-treatment to increase pH. The manufacturer states, “The volume of nitrate-containing effluent from the WIN process is typically less than 0.2% of the treated water and, in some cases, may be land applied as fertilizer” (ARA & Purolite N.D.b). As discussed above, the use of SBA IX resin results in a high-nitrate brine waste stream. Due to the high salt content, the nitrate in the waste stream generally cannot be beneficially reused. However, with the use of alternative weak bases for regeneration, the waste stream does not have a high salt content and could potentially be recycled as fertilizer.

3.1.7 Ion Exchange - Case Studies

Conventional Ion Exchange - Case Studies

The following case studies provide detailed information on the design and operation of full-scale conventional IX treatment plants for nitrate removal. Conventional IX is also used by the Chino Basin Desalter Authority in combination with RO. Detailed case studies for the Chino Desalter are listed separately in the RO case study section.

System Location: California

CASE #1

System Type: Community Water System

Treatment Type: Conventional Ion Exchange
Startup Date: 2007

Treatment Type	*Ion Exchange and Blending*
System Capacity	*400 gpm*
Raw Water Nitrate	*31 – 53 mg/L as nitrate* *7 – 12 mg/L as N*

System Description

A California utility is responsible for a system that has two groundwater supplies, one of which has nitrate at levels that exceed the MCL. The impacted well has a production capacity of 400 gpm and historical nitrate concentrations ranging from 31 mg/L to 53 mg/L of nitrate as NO_3^- (7 – 12 mg/L as N). The utility has implemented a blending program and installed a conventional IX treatment system in 2007 for nitrate control and treatment.

The nitrate impacted sources also have arsenic levels above the MCL of 10 ug/L, which influenced the decision to install IX. IX can simultaneously remove nitrate and arsenic. The treatment system is comprised of three pressure vessels. Two vessels contain Purolite A300E for arsenic removal and the third vessel contains the nitrate selective Purolite A520E resin. The system was originally installed in 2007 and was further modified in 2009. The system is designed to decrease nitrate levels to less than 22 mg/L as nitrate (5 mg/L as N) prior to blending. The maximum distribution system water goal for nitrate is 35 mg/L as nitrate (8 mg/L as N). To assure the system maintains the treatment goals, online nitrate analyzers have been installed at two locations, after the IX system and after blending.

Source Water Quality

- Nitrate - mg/L as NO_3^- (mg/L as N)
  - Average – 35 (8)
  - Minimum – 31 (7)
  - Maximum – 53 (12)
- Co-contaminants
  - Arsenic (15 ug/L)
  - Sulfate (66 mg/L)

Treatment Technology Selection

The conventional IX system was selected to treat both nitrate and arsenic. The decision was further influenced by the ability to discharge the waste brine to a municipal sewer, a cost-effective disposal option. Often technology selection is limited by the costs of brine management. If a utility has the option to dispose of the waste brine to a municipal sewer it can significantly decrease the operations and maintenance cost of the system. No other technologies were pilot tested prior to installation of the IX system.

Operational Notes

Since nitrate is an acute contaminant, the reliability of the treatment system is an utmost concern. The system is equipped with online nitrate analyzers which causes a shutdown of the treatment system when nitrate is at or above 35 mg/L as nitrate (8 mg/L as N). Additionally, the treatment system has experienced arsenic breakthrough resulting in concentrations above the MCL. The treatment system has also experienced premature nitrate breakthrough as a result of faulty distributors in the ion exchange vessels which have since been replaced.

Treatment System Parameters

- Design Capacity
  - 400 gpm maximum capacity
- Pretreatment
  - None
- Treatment system footprint
  - Treatment system: 30’ x 35’
  - Building footprint: None
- Ion exchange pressure vessels
  - Number of vessels: 3
  - Diameter of vessels: 6’
  - Height of vessels: 6’
- Design Loading Rate
  - 5.2 gpm/ft^2
- Bed volumes prior to regeneration
  - 345 – 470 (approximately 220,000 – 300,000 gallons treated)
- Resin Type: Purolite A200E and Purolite A520E nitrate selective
- Previous resins used: None
- Salt Consumption: 1,700 lb/week (May – Sept) and 600 lb/week (Oct – April)
- Volume of brine generated
  - 800 gal/vessel/backwash
  - 99.7% water efficiency
- Monitoring:
  - Online nitrate analyzer
  - Laboratory samples

Residuals Management

The waste brine is discharged to the sewer and sent to a municipal wastewater treatment facility, with an annual cost of $12,000. There have not been any unexpected residuals that have impacted the disposal option.

Technology Benefits and Drawbacks

Benefits
- Ease of regeneration
- Direct-to-sewer brine disposal
- Simple, manually operated system

Drawbacks
- System has potential for breakthrough of nitrate or arsenic
- Time intensive operations
- Required increase in operator certification (California T-3)

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)	
Total:	$350,000
Annual O&M Costs (Total with explanation or component costs)	
Total:	$66,500
Resin:	$13,000
Brine Disposal or Treatment:	$12,000
Chemicals:	$5,500
Repair/Maintenance (not including Labor):	$4,500
Power:	$2,500
Labor ($):	$28,000

System Location: California

CASE #2

System Type: Community Water System

Treatment Type: Conventional Ion Exchange
Startup Date: 2006

System Description

Treatment Type	*Ion Exchange*
System Capacity	*400 gpm*
Raw Water Nitrate	*44 mg/L as nitrate* *10 mg/L as N*

A California water utility operates a system that has three groundwater supplies with varying amounts of nitrate contamination. Two of the wells require treatment as the nitrate concentration is at or above the 45 mg/L as nitrate (10 mg/L as N) MCL. The third well has nitrate ranging from 22 mg/L as nitrate (5 mg/L) as N to 31 mg/L as nitrate (7 mg/L as N) and does not require treatment, but is blended with the IX treated water prior to distribution.

In 2006, a conventional IX system was installed. The treatment system is comprised of three pressure vessels that contain the Rohm and Haas HP 555 ion exchange resin. Since the nitrate concentration in the impacted wells is typically at the MCL, but above the established water quality goal of 35 mg/L as nitrate (8 mg/L as N), a sidestream treatment approach is utilized. In sidestream treatment a portion of the flow is passed through the treatment system while the remainder is bypassed. The IX treatment results in very low nitrate concentration and allows the two streams to be combined to achieve the water quality goal. Sidestream treatment offers capital and operational costs savings.

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L N)
    - Average – 44 (10)
- Co-contaminants
    - Sulfate

Treatment Technology Selection

IX was selected for this system because the utility was familiar with the technology from other installations. No technologies were pilot tested prior to the installation of the IX system.

Treatment System Parameters

- Design Capacity
    - 400 gpm maximum capacity
- Pretreatment
    - 100 mesh strainer
    - 10 micron screen
- Treatment system footprint
    - Treatment system: 20’ x 100’
    - Residual handling : 40’ x 100’
    - Total footprint: 60’ x 100’
- Ion exchange pressure vessels
    - Number of vessels: 3
    - Diameter of vessels: 4’
    - Height of vessels: 6’
- Design Loading Rate
    - 10 gpm/ft^2
- Bed volumes prior to regeneration
    - 309 BV
    - Regeneration occurs every 12.4 hours
- Resin Type: Rohm and Haas HP 555
- Previous resins used: None
- Salt Consumption: Estimated to be 32,000 lbs of salt per month
- Volume of brine generated
    - 1,000 gal/vessel/backwash
    - 99.2% water efficiency
- Monitoring:
    - Online nitrate analyzer
    - Laboratory samples

Residuals Management

The spent brine is held in a storage tank and ultimately disposed off site. Brine volumes are minimized by using a partial flow treatment strategy where only 400 gpm is treated by the IX system while the remaining 500 gpm bypasses the system. The untreated flow is recombined with the treated water prior to entering the distribution system.

Technology Benefits and Drawbacks

Benefits

- Familiarity with technology
- Simple, manually operated system

Drawbacks

- No onsite brine disposal
- Design flaws
- Inconsistent operations
- Potential for nitrate breakthrough

Treatment Technology Costs

Cost information was not included with the survey response.

Operational Notes

This particular IX system has had a series of operational challenges, many of which can be attributed to faulty engineering. Sections of exposed Schedule 80 PVC failed due to freezing, while other sections of pipe failed due to prolonged UV exposure. The brine reclaim tank experienced algal growth and was ultimately replaced with an opaque, UV-resistant tank.

The plant has experienced shutdowns due to an exceedance of the nitrate MCL. Routine sampling showed distribution system nitrate concentrations above the MCL. The system utilizes an online nitrate analyzer to prevent MCL violation; however, a calibration error prevented the system from shutting down as programmed. It is believed a second failure occurred as a result of the brine saturator having a low salt concentration resulting in incomplete regeneration of the resin.

It should be noted that, due to concerns of nitrosamine release (which can be common for IX systems), the effluent of each vessel and the entry point to the distribution system are tested before the system is placed in service.

Modifications to Conventional Ion Exchange - Case Studies

The following case studies provide detailed information on the design and operation of full-scale low-brine IX systems including MIEX®, ISEP®, and Envirogen (formerly Basin Water) systems.

System Name: Indian Hills
System Location: Indian Hills, CO
PWSID: CO 0130065
System Type: Community Water System

CASE #3

Treatment Type: Counter Current Magnetic Ion Exchange
Questionnaire completed by: Operations staff and Orica Water Care Representatives
Startup Date: 2009

Treatment Type	*Counter Current Ion Exchange*
System Capacity	*50 gpm*
Raw Water Nitrate	*53 – 71 mg/L as nitrate* *12 – 16 mg/L as N*

System Description

The Indian Hills Water District (District) utilizes a groundwater well with a production capacity of approximately 50 gpm. The well has historical nitrate concentrations ranging from 12 mg/L to 16 mg/L of nitrate as N. The District has implemented the counter-current ion exchange process developed by Orica Water Care. Unlike traditional packed bed IX, the Orica process uses a Magnetic Ion Exchange (MIEX®) resin in a series of fluidized beds which allows for a reduction in brine generation.

The minimization of waste brine is accomplished through frequent batch resin regeneration. The magnetized resin encourages agglomeration of loaded resin particles, resulting in faster settling. Loaded resin is removed from the bottom of the IX vessel and is passed to regeneration, while regenerated resin is continuously added at the top of the IX vessel. This configuration reduces the risk of nitrate spiking/chromatographic peaking because clean resin, added at the end point of the system, captures any displaced nitrate, while competing ions, such as sulfate, can be removed early on in the resin vessel.

Regeneration is performed continuously in small batches. Loaded resin is passed to regeneration tanks through the bottom of the IX vessel. The loaded resin is regenerated and then returned to the contactor vessel, thus maintaining a consistent ion exchange capacity in the contactor vessel. Numerous regenerations are performed on a daily basis, with the actual number of regenerations depending on the system's flow rate.

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L N)
    - Average – 62 (14)
    - Minimum – 53 (12)
    - Maximum – 71 (16)
- Co-contaminants
    - None noted in survey

Treatment Technology Selection

The counter-current magnetic ion exchange system was selected based on the expected low levels of brine when compared to conventional packed bed IX. Prior to full-scale implementation, the MIEX® process was pilot tested at Indian Hills. Indian Hills did not pilot test any other technology prior to implementation.

Treatment System Parameters

- Design Capacity
  - 50 gpm maximum capacity
  - 25 gpm typical
  - 11 – 14 gpm actual
- Pretreatment/Post-treatment
  - 1 micron bag filters
  - Chlorination
- Treatment system footprint
  - Treatment system: 9.75’ x 5.5’
  - Regeneration system: 13.3’ x 6.5’
  - Residuals handling system:
    - 2,000 gal storage tank
  - Total system footprint: 20.5’ x 9.75’
  - Building footprint: 30’ x 50’
- Number of contactors
  - (2) – 3’ diameter; 6.75’ height
- Design Loading Rate
  - 7.27 gpm/ft^2 at design flow
- Bed volumes prior to regeneration
  - 125 BV
- Resin Type: MIEX DW 1401- Strong base anion exchange resin in chloride form
- Previous resins used: None
- Salt Consumption: 3,500 lbs/MG treated water
- Volume of brine generated
  - 2,800 gal brine/MG treated water
  - 99.7% water efficiency
- Monitoring:
  - Online nitrate analyzer
  - Laboratory samples
  - Field colorimeter

Residuals Management

The waste brine is sent to a 2,000 gallon underground storage tank. The brine is periodically pumped from the storage tank and ultimately land applied. Indian Hills is investigating deep well injection as an alternative disposal mechanism. The waste brine was analyzed in the pilot portion of the project and no unforeseen residuals were identified that would further limit the brine disposal options.

While not specifically analyzed at this site, waste brine from other MIEX® installations has not had detectable nitrosamine concentrations.

For the MIEX® System, Bed Volumes are defined as the volume of water treated per volume of resin regenerated. BV = gal water treated/gal resin regenerated. For example, if 5 gal of resin were regenerated for every 1000 gal of water treated, the regeneration rate would be: BV = 1000gal/5gal = 200BV.
The design regeneration rate for the MIEX® System is 125 BV, meaning that 8.0 gal of resin are regenerated per 1000 gal of water treated. The system regeneration rate can be adjusted through the control system. There is approximately 1 – 2 gallons of resin attrition per 1 MG of treatment. Small amounts of resin are periodically added to the regeneration system manually.

Technology Benefits and Drawbacks

Benefits

- Consistent treatment performance
- Relatively low volumes of waste brine
- No nitrate dumping

Drawbacks

- Generation of waste brine
- Resin lost in the treatment must be removed prior to distribution

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)	
Operating and Monitoring Equipment:	Approximately $150,000 (IX treatment system, including initial resin fill)
O&M Costs (Total with explanation or component costs)	
Resin:	$0.08 – $0.15/1000 gallons treated
Resin Disposal:	N/A
Brine Disposal or Treatment:	N/A
Chemicals:	Salt (based on an estimated salt cost of $100/ton): $0.15 – $0.20/1000 gallons treated

Sources*

Vaughan, F., Orica Watercare. (2010) Personal Communication. August, 2010.
Martin, B. (2010) Completed questionnaire. September, 2010.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

System Location: California

CASE #4

System Type: Community Water System

Treatment Type: Ion Exchange, Well Modification (proposed), Well Abandonment
Startup Date: 2003 – 2009

System Description

Treatment Type	*Ion Exchange and Blending*
System Capacity	*500 – 900 gpm*
Raw Water Max Nitrate	*35 – 89 mg/L as nitrate* *8 – 20 mg/L as N*

A California district operates 27 groundwater sources, 15 of which contain nitrate at or near the MCL. The district has considered and implemented a variety of solutions including IX, well modifications, and well destruction. The nitrate impacted wells range in capacity from 500 gpm to 1000 gpm with nitrate ranging from 35 – 89 mg/L as nitrate (8 – 20 mg/L as N). It should be noted that a water quality goal of 35 mg/L of nitrate as NO_3^- (< 8 mg/L of nitrate as N) has been implemented in the district, and all sources are treated to below this level.

The district began actively treating the nitrate contaminated sources in 2003 and the most recent system was installed in 2009. The district currently has 6 wells with active IX systems, 7 wells have been destructed or made inactive, 2 wells are being considered for well modifications, and one well has an enhanced control system where there will be an automatic shut down if the nitrate levels exceed a predetermined level. The range of historical nitrate concentrations of the wells is shown below.

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L N)
    - Well A- 7 to 58 (1.5 to 13.1)
    - Well B- 15 to 48 (3.4 to 10.8)
    - Well C- 4 to 47 (0.9 to 10.6)
    - Well D- 13 to 88.4 (2.9 to 20.0)
    - Well E- ND to 40.7 (ND to 9.2)
    - Well F- 11 to 53.8 (2.5 to 12.1)
    - Well G- 2 to 52 (0.5 to 11.7)
    - Well H- 8.5 – 72.4 (1.9 to 16.3)
    - Well I- 15.0 to 40.4 (3.4 to 9.1)
    - Well J- 6 to 33.1 (1.4 to 7.5)
    - Well K- 51 (11.5)
    - Well L- 87 (19.6)
    - Well M- 37 (8.4)
    - Well N- 37.4 (8.4)
    - Well O- 64 (14.5)

Treatment Technology Selection

The first nitrate treatment system the district installed was in 2003. At the time of installation IX was deemed the best available technology as it was the most economical with respect to the well's nitrate concentrations and flow rates. When possible, the district elected to install similar systems on the wells that required treatment in an effort to establish operational parallels between their systems. When the footprint of the system or excessive nitrate concentrations made IX treatment infeasible, the district has elected for well destruction. In recent cases where treatment is necessary, the district has evaluated well modifications to determine if it is feasible to reduce the nitrate concentrations without the need for active treatment.

The following section shows the typical parameters of the district's individual IX systems with the exception of the salt consumption which represents the total salt use for the entire district.

Treatment System Parameters

- Design Capacity
    - 500 – 900 gpm
- Pretreatment
    - None
- Treatment system footprint
    - Two 35' x 20' cement slabs
    - Treatment system: 35' x 10'
    - Housed in a cargo container
- Residuals handling system
    - Three 12' x 12' dia. tanks
- Ion exchange pressure vessels
    - Number of vessels: 16
    - Diameter of vessels: 3'
    - Height of vessels: 6'
- Design Loading Rate
    - 12 gpm/ft^2
- Bed volumes prior to regeneration
    - 300 BV
- Treatment System Manufacturer
    - Envirogen Technologies
- Resin Type: Conventional type 1 ion exchange resin
- Previous resins used: None
- Salt Consumption: 25 tons/ week combined for all systems
- Volume of brine generated
    - 1.3 bed volumes per vessel regenerated
    - Approximately 99.6% water efficiency
- Monitoring
    - Online nitrate analyzer
    - Laboratory samples

Residuals Management

The waste brine is disposed of at an offsite facility. This decision has been impacted by elevated selenium and NDMA in the waste brine.

Technology Benefits and Drawbacks

Benefits

- Allows wells that would normally be offline due to nitrate contamination to be used for potable purposes
- The treatment system vendor provides on-site technical support and operations

Drawbacks

- Treatment system failure poses an acute health risk to the potable water system
- Time intensive operations if there is a treatment disruption
- The systems have numerous valves and moving parts. If there is a mechanical failure it can be difficult to identify the source
- High operating and brine disposal costs
- The district pays the system vendor for stand-by fees and service charges if the system is in stand-by mode during times of low water use

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)	
Total:	$360,000 per unit – Electrical, Piping, Setup, and Sampling/testing (construction costs) The district does not own the treatment plants, tanks, resins, etc.
O&M Costs (Total with explanation or component costs)	
Total:	$59,239.41 per month, per unit
Brine Disposal or Treatment:	$34,016.75 per month, per unit
Repair/Maintenance (not including Labor):	$3,525.00 (service fee) per month, per unit
Salt:	$13,541.41 (salt) per month, per unit
Other:	$7,500.00 (stand by fee) per month, per unit $656.25 (tax) per month, per unit

System Name: City of Chino
System Location: Chino, CA
PWSID: CA3610012
System Type: Community Water System

CASE #5

Treatment Type: Ion Exchange (IX)
Questionnaire completed by: Gilbert Aldaco, Water Utilities Supervisor, City of Chino Public Works
Startup Date: 2006

System Description

Treatment Type	*Ion Exchange and Blending*
System Capacity	*5000 gpm*
Raw Water Nitrate	*~40 – 200 mg/L as nitrate* *~9 – 45 mg/L as N*

The City of Chino, CA, operates 13 groundwater sources and 3 GWUDI (groundwater under direct influence of surface water) sources. All of the GWUDI and 8 of the groundwater sources are impacted by nitrate contamination. IX and blending are used to address high nitrate and perchlorate levels. One of the wells is inoperable due to perchlorate contamination. IX using the ISEP® system is used for the treatment of 3 wells ranging in capacity from 1100 to 2300 gpm and is being considered for a 4th well. Additionally, an Envirogen (formerly Basin Water) IX system is used for the treatment of an 800 gpm well. Treated surface water from the Metropolitan Water District of Southern California (MWDSC) and treated groundwater from the Chino Basin Desalter Authority is used for blending. The blend ratio is 3:1. The ISEP® system was built in 2005, with a capacity of 5000 gpm (7.2 MGD), and approved for operation in 2006.

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L N)
    - 40 to 100 (9 to 45)
- Co-contaminants
    - Perchlorate

Treatment Technology Selection

The costs and feasibility of several alternatives for nitrate treatment, including reverse osmosis, biological processes, and conventional (fixed bed) IX were investigated. Biological treatment and fixed bed IX were pilot tested prior to installation of the current system. The ISEP® system was selected based in part on the potential to simultaneously address perchlorate contamination and due to the efficiency of operation.

Additional Information

- Operators were not required to increase their level of certification to operate the treatment plant.
- When asked about the areas in which the nitrate treatment technology has exceeded expectations, the response was, “Reliability and ability to effectively remove contaminants at varied flow rates (i.e., 1500 gpm to 5000 gpm).”
- There has been no incidence of a plant shutdown due to an alarm or exceedance of an MCL; however, the plant was shut down for approximately 1 month due to theft of computer control equipment.

Treatment System Parameters

- Design Capacity
  - 5000 gpm
- Pretreatment: Filtration
- Footprint
  - Treatment system: 4400 sq.ft.
  - Residuals handling: 2300 sq.ft.
  - Total system: 6700 sq.ft.
- Ion exchange vessels
  - Number of vessels: 30
  - Diameter of vessels: 3’
  - Height of vessels: 6’
- Max. nitrate concentration goal for delivered water
  - ~36.3 mg/L NO_3^- (~8.2 mg/L N)
- Nitrate concentration goal for the treatment system (before blending)
  - ~19.0 mg/L NO_3^- (~4.3 mg/L N)
- Manufacturer: Calgon Carbon Corporation
- Resin Type: Purolite SBA
- Previous resins used: None
- Bed volumes prior to regeneration
  - Continuous regeneration
- Salt Consumption (@ 2400 gpm)
  - Nitrate mode – 4.9 tons/day
  - Perchlorate mode – 18.6 tons/day
- Volume of brine generated (@ 2400 gpm)
  - Nitrate mode: 12.7 gpm
    - 99.5% water efficiency
  - Perchlorate mode: 31 gpm
    - 98.7% water efficiency
- Monitoring
  - Weekly effluent testing for nitrate and perchlorate
  - Monthly effluent testing for nitrite, sulfate, and total Coliform
  - Monthly raw water testing for nitrate, perchlorate, nitrite, sulfate, total Coliform, HPC
- Grab Samples for resin byproduct testing were negative for nitrosamines

Residuals Management

Waste brine is discharged to a non-reclaimable waste pipeline leading to the LA County Sanitation District, with a total disposal cost of ~$50,000/yr.

Technology Benefits and Drawbacks

Benefits

- Flexibility of operation and High efficiency
- Reliability and Ease of O & M

Drawbacks

- None listed

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)	
Total:	~ $4.6 million
Housing:	~ $492 K
Piping:	~ $1.1 million
Resin:	~ $350 K
Testing:	~ $20 K
Other:	~ $2.4 million for ISEP Equipment and Engineering; ~ $350 K for electrical upgrade; ~ $200 K for pumps and associated equipment.
Annual O&M Costs (Total with explanation or component costs)	
Total:	Not reported
Brine Disposal or Treatment:	~ $50 K
Chemicals:	~ $364 K for salt, ~ $50 K for hydrochloric acid

3.2 Reverse Osmosis (RO)

Reverse osmosis can be a feasible option for nitrate removal in both municipal and Point-of-Use applications (Cevaal et al. 1995; Black 2003; Howe 2004). The first commercial application of RO for potable water treatment was in Coalinga, CA, in 1965 (National Academy of Engineering 2008). RO can be used to address multiple contaminants simultaneously including ionic (e.g., nitrate, arsenic, sodium, chloride, and fluoride), particulate (e.g., asbestos and Protozoan cysts), and organic constituents (e.g., some pesticides) (Dvorak & Skipton 2008). Water is forced through a semi-permeable membrane under pressure such that the water passes through, while contaminants are impeded by the membrane. A typical process schematic of RO for nitrate removal from potable water is illustrated in Figure 8.

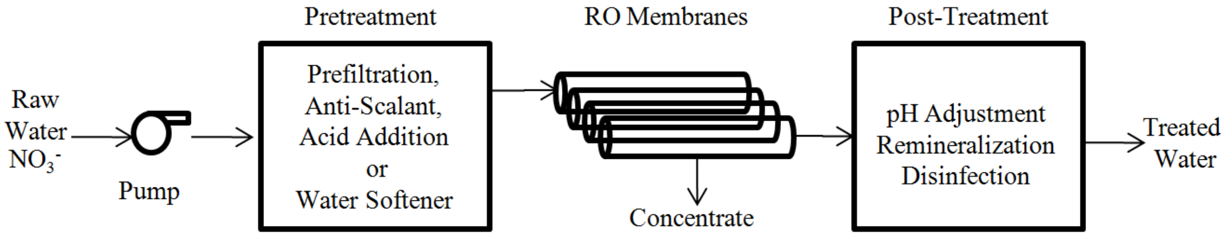

Figure 8. Reverse osmosis schematic.

The required pressure will be dependent on the concentration of solute in the feed water. The collected concentrate is high in nitrate and other rejected constituents (salts) and requires appropriate disposal. The extent to which the RO membrane removes constituents from the water is called the rejection rate. Rejection rates for sodium chloride and sodium nitrate can be as high as 98% and 93%, respectively (Elyanow & Persechino 2005). The water recovery rate of an RO system refers to the "maximum percentage of permeate produced from a given feed flow," and the flow/flux rate is "the maximum flow of permeate through a sq. ft. of the membrane" (Cevaal et al. 1995, p. 107). Modifications and improvements to standard RO have led to the emergence of more efficient RO processes including High Efficiency Reverse Osmosis (HERO™) and Ultra-low Pressure Reverse Osmosis (ULPRO) systems.

3.2.1 Reverse Osmosis - Design Considerations

Table 6 summarizes key design considerations in the application of RO to nitrate removal from potable water.

Table 6. Summary of design considerations for reverse osmosis.

RO Membranes	• Thin film membranes - Higher rejection rates, lower pressures than CTA membranes • Cellulose triacetate membranes (CTA) - Tolerant of low chlorine levels • Hollow fiber membranes - Compact configuration • Ultra-low pressure RO membranes (ULPRO) • Consider rejection rate, water recovery, and frequency of cleaning • Multiple contaminant removal

Pretreatment	• Prevent membrane damage, scaling and biological, colloidal, and organic fouling
	• Scaling
	    o Acid (e.g., sulfuric acid) and/or anti-scaling agents (e.g., poly-acrylic acid)
	    o Water softening
	• Biological fouling
	    o Upstream disinfection, but dechlorination to prevent membrane damage
	        ▪ Reducing agent (e.g., sodium bisulfate) or activated carbon
	• Colloidal fouling
	    o Pre-filtration to remove suspended solids
	    o Chemical treatment to keep suspended solids in solution

Post-Treatment	• Avoid corrosion
	    o Adjust pH, restore alkalinity for buffering capacity (see remineralization) and/or add corrosion inhibitor (e.g., poly-orthophosphate blend)
	• Remineralization
	    o Blending, pH adjustment, addition of caustic soda, bicarbonate, sodium carbonate, phosphates, and/or silicates
	• Blending, disinfection, and storage

Chemical Usage	• pH adjustment, up and down (acids and bases) • Anti-scalants • Cleaning chemicals (acids and bases)

O&M	• Frequency of membrane cleaning depends on water quality and membrane used
	    o Typically once a month for 1 hour
	• Management of chemicals and pre-filtration system
	• Waste storage and disposal
	• Membrane replacement/membrane life
	    o Up to 20 years or more with appropriate pretreatment and maintenance
	• Monitoring of nitrate levels and membrane flux rate
	• Automation can be feasible
	• Low operational complexity (though higher than IX depending on pretreatment needs)

System Components	• Maximize water recovery while minimizing energy use
	    o Pressure range of 100 to 200 psi
	    o Based on system size and feed water quality
	• Key system configuration parameters are system flow rate, number of membranes/stages, system footprint, flux rate, water recovery rate, pump selection and sizing, pressure requirement, cleaning frequency

Waste Management and Disposal	• Significant cost of waste brine/concentrate disposal of greatest concern for inland systems • Management options include sewer or septic system, drying beds, trucking off-site, coastal pipeline, deep well injection, and advanced treatment • Disposal options can be limited by waste brine/concentrate water quality (e.g., volume, salinity, metals, and radionuclides) • Optimization of recycling and treatment of waste brine/concentrate • Higher water recovery (more costly) to minimize waste volume (tradeoff between energy costs and disposal costs)
Limitations	• Need to prevent membrane scaling, fouling, and damage ○ Hardness, iron, manganese, suspended solids, silica, and chlorine • High energy demands • Disposal of waste concentrate • Complete demineralization (no control over target constituents)

Water Quality

While regulated by operational pressure, the water recovery rate depends largely on feed composition. Problematic constituents include sulfate, calcite, calcium sulfate dehydrate (gypsum), silica, colloids, and microorganisms (Cevaal et al. 1995; Elyanow & Persechino 2005; Tarabara 2007). Filtration upstream of the RO membranes is required to remove suspended solids. The life of the RO membranes and prefilters, and the frequency of membrane cleaning are also directly dependent on water quality and the efficiency of pretreatment measures.

Treatment efficiency can be compromised by membrane fouling. Anything that decreases available membrane surface area can limit the passage of water through the membrane and decrease water recovery. The four main types of membrane fouling are scaling, colloidal fouling, biological fouling, and organic fouling. When the salt concentration in the feed water exceeds the saturation point at the membrane surface, precipitation of solids on the membrane can diminish the removal efficiency (Elyanow & Persechino 2005). Scale forming constituents,[14] such as precipitates of silica, calcium, barium, and strontium salts pose a significant threat to RO by limiting the membrane surface area

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[14] Pretreatment options to prevent scaling include the addition of acid and/or anti-scaling agents and water softening. By decreasing the pH, the prevalent form of the carbonate cycle is bicarbonate rather than the carbonate ion. The precipitation of calcium carbonate will therefore be limited by the concentration of the carbonate ion. Note: the addition of acid helps only if the scale forming constituent is calcite, due to the speciation of carbonate (Lenntech, 2009c). Anti-scaling chemicals can function in three ways: threshold inhibition, crystal modification, and dispersion (Lenntech, 2009c). Threshold inhibition occurs when the anti-scalant increases the solubility of a potential scalant to super saturation, allowing for a greater concentration to remain in solution. Crystal modification refers to the interference of negatively charged functional groups on the anti-scalant with salt crystal formation and membrane attachment. Anti-scaling chemicals can also promote dispersion of crystals by attaching to them and increasing their negative charge. The most economical means of membrane scale control will depend on the system, but typically the use of anti-scaling agents alone or in combination with acid addition is the most financially prudent option. The third alternative to prevent membrane scaling is to remove the problematic constituents from the water entirely. Water softening can be used to replace calcium and magnesium cations with sodium ions. Generally the most expensive option, this requires the addition of a water softener upstream of the membranes and will result in an additional brine waste stream.

through which water can pass.[15] Development of biomass on the membrane surface can have a similar negative effect on performance.[16] Additionally, RO membranes have limited to zero tolerance for organic species, like grease and oil; organic fouling inhibits membrane performance (Lenntech 2009e). Lastly, suspended solids not removed by pretreatment filtration can inhibit membrane performance through colloidal fouling.[17] Silica can be a particularly problematic constituent for RO membranes due to the potential for colloidal silica fouling and silica scale formation, which can be very difficult to remove. Modifications to conventional RO have emerged to manage high silica source waters (Section 3.2.5 Reverse Osmosis - Improvements and Modifications, below). Common pretreatment measures used to address membrane fouling are included in Table 6.

With acid addition in pretreatment, the permeate will need to be neutralized in post-treatment to avoid corrosion in the distribution system. Alternatively a corrosion inhibitor, such as a poly-orthophosphate blend, can be used (U.S. EPA 2003). Additionally, because RO is not selective in the removal of ions, treated water is demineralized. Thus, alkalinity may need to be added to restore minerals and buffering capacity to avoid corrosion in the distribution system[18] (WHO 2004; Lenntech 2009f). In post-treatment, blending (if used) follows the RO modules, after which water is disinfected and stored (Cevaal et al. 1995).

System Components and Site Considerations

RO systems are operated in stages. Following pretreatment, water is pumped through the membranes with booster pumps (Cevaal et al. 1995). Water recovery can be improved by passing the concentrate through the membranes more than once, but higher water recovery comes at the expense of increased scaling potential. As the water becomes more concentrated, saturation can lead to precipitation on the membrane (Elyanow & Persechino 2005). The membrane flux rate can be decreased to limit scaling and increase membrane life (Cevaal et al. 1995). Key aspects of the system are pressure pumps, membrane configuration, membrane flux rate, number of stages/number of membranes, flow rate, and cleaning and anti-scalant requirements. Pump sizing is based on system size and pressure requirements. The water recovery rate can be regulated by the operational pressure. The necessary pressure is dependent on the concentration of solute in the feed water. For example, with a conductivity of 1550 µS/cm,

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[15] The Langelier Saturation Index (LSI) and other corrosion indices are used to characterize the scaling potential of calcium carbonate (calcite), a commonly problematic constituent.

[16] Upstream disinfection can limit membrane biofouling; however, additional measures must be taken to avoid membrane exposure to chlorine "by dosing with a reducing agent (such as sodium bisulfate) or by contacting with activated carbon" (Elyanow & Persechino 2005, p. 6).

[17] The potential for colloidal fouling is typically characterized by the silt density index (SDI). An SDI greater than 3 can indicate the need for further pretreatment to minimize cleaning frequency and membrane damage (Lenntech 2009e; Elyanow & Persechino 2005). Chemical treatment can keep suspended solids in solution. Alternatively, a prefilter can be used to remove solids from the feed water (Remco Engineering N.D.).

[18] It is important to note the relationship between alkalinity and pH. Alkalinity is a measure of buffering capacity or the resistance to changes in pH. Demineralized water or water with a low buffering capacity will be susceptible to more dramatic pH changes and is considered unstable. The pH of acidic product water should be adjusted and the buffering capacity of demineralized product water should be restored to avoid corrosion downstream. Options for the stabilization and remineralization of demineralized water include blending, pH adjustment, and addition of caustic soda, bicarbonate, sodium carbonate, phosphates, and/or silicates (WHO 2004; Lenntech 2009f). Corrosion indices and models can be used to determine appropriate solutions for specific scenarios.

Panglisch et al. (2005) determined the suitable pressure to be 145 – 174 psi. Similarly, at start-up, the operational pressure at an RO facility in Brighton, CO, used for nitrate removal was 170 psi (Cevaal et al. 1995).

Two types of spiral wound RO membranes are commonly used: polyamide thin film composite membranes (TF) and cellulose triacetate membranes (CTA) (Cevaal et al. 1995 and Remco Engineering N.D.). While TF membranes are capable of slightly higher rejection rates and can be operated at lower pressures, CTA membranes are tolerant of low chlorine levels (AWWA 2011). Hollow fiber RO membranes are also available, which can minimize system footprint, but can be more susceptible to fouling from suspended solids (Hydranautics 2001). Using recently developed ultra-low pressure RO (ULPRO) membranes, operational pressures can be reduced, decreasing power costs (Drewes et al. 2008). For additional information see Section 3.2.5 Reverse Osmosis - Improvements and Modifications.

Residuals Management and Disposal

The volume of the waste stream can be considerable, ranging from 15% to 50% of the starting volume depending on the operational parameters (Howe 2004). The waste stream, or concentrate, can be discharged to a wastewater treatment plant or a septic system (Bilidt 1985; Howe 2004), as long as the system can accommodate an increased salt concentration. Additional disposal options include drying beds, infiltration basins, trucking off-site, a coastal pipeline, deep well injection, advanced treatment, and most commonly, discharge to nearby surface salt-waters (i.e., oceans), when available (Howe 2004). Important water quality characteristics of the waste brine (e.g., volume, salinity, metals, and radionuclides) can affect the feasibility and costs of disposal. Options for inland communities are more limited and costly. Proximity to coastal power plants can be advantageous. Power plants using ocean water for cooling can provide a pre-existing infrastructure for disposal to ocean waters (Black 2003).

The high cost of nitrate laden concentrate disposal has led to research into optimization of recycling and treatment of this waste stream. Coupling of RO systems with biological, chemical, or catalytic denitrification enables removal of nitrate from the waste concentrate, with reduction to nitrogen gas. Additionally, the Vibratory Shear Enhanced Process (VSEP), from New Logic Research, has been explored in the context of treatment of RO concentrate from wastewater treatment. By applying a shear force across the membrane, pore clogging by colloidal particles is minimized, leading to the potential for improved water recovery (Lozier et al. N.D.). Several combined configurations of interest are discussed below in Section 3.6 Brine Treatment Alternatives and Hybrid Treatment Systems.

Maintenance, Monitoring, and Operational Complexity

RO systems are typically highly automated, accommodating the greater operational complexity of RO operation, in comparison with IX. Several operational decisions will be dictated by operator availability and training. For instance, chemical addition in pretreatment can be quite effective, but will require more intensive maintenance. In contrast, opting for the more expensive choice, installing a water softener, will require less operator time. Membrane cleaning frequency varies widely and depends on the efficiency of pretreatment measures and water quality. Interruption of operation is not always necessary as the membranes can be isolated and cleaned in place (CIP) in stages. Membranes are typically cleaned with acid or caustic solutions (WA DOH 2005). Cleaning agents are selected based on the cause of membrane fouling. Cleaning and rinsing can take an hour and with effective pretreatment, monthly membrane cleaning should be sufficient (Remco Engineering N.D.; Bates N.D.). With effective pretreatment, cleaning frequency can be significantly minimized. An RO plant in Milan, Italy, using anti-scalants, requires cleaning only once every 18 months (Elyanow & Persechino 2005). Filters should be checked weekly and, if used, the water softener should be maintained with sufficient salt every day. Effluent nitrate concentrations require monitoring to ensure compliance and to assess membrane performance. Over time, membrane degradation will lead to a gradual decrease in removal efficiency. Membrane life varies and can range from 5 to 20 years or more (Remco Engineering N.D.). Waste concentrate management consists of appropriate storage and disposal. More operationally complex than IX, operators of RO systems will typically require more training and system maintenance will demand more time and chemicals. However, with the implementation of appropriate pretreatment measures and the ability for system automation, operational complexity can be minimized.

3.2.2 Reverse Osmosis - Cost Considerations

For the efficient operation of an RO system, the fundamental objective is to maximize water recovery with the minimum amount of energy and chemical usage, while meeting necessary potable water guidelines. Factors affecting system cost include facility size (how much water), source water quality (including nitrate concentration), environmental factors (temperature and pH), and target effluent nitrate concentration (Bilidt 1985). Lower operating pressures are less costly, but result in decreased water recovery. High operating pressures maximize water recovery (decreasing disposal costs), but increase energy demands and the need for "specialized pumps" (WA DOH 2005). Thus, there is a trade-off between the costs of increasing water recovery (increased pretreatment and operational pressure) and the costs of disposal (pumping, storage, and disposal expenses). In pretreatment, the use of anti-scalants rather than acid or a water softener is generally the least expensive. The use of a water softener is the least cost competitive option (Lenntech 2009c). Regarding small water systems, "reverse osmosis is one of the most expensive forms of centralized treatment and will likely not be cost effective unless there are multiple contaminants needing removal" (WA DOH 2005, p. 27).

Capital costs for RO include land, housing, piping, storage tanks, O&M equipment, membranes, preliminary testing (pilot studies), permits, and training. O&M costs include membrane and filter replacement, membrane and filter disposal, concentrate disposal or treatment, chemical use (anti-scalant, pH adjustment, disinfection, etc.), repair, maintenance, power, and labor.

Published cost information, from existing RO installations used for nitrate treatment, is listed in Table 7. Costs have been adjusted to 2010 dollars, unless indicated otherwise. Costs can be difficult to assess due to inconsistencies in how cost information is reported. Comparison of treatment costs is not always valid due to differences in influent water quality parameters, system size, waste management options, and system configuration. Published costs do not always include comparable information. It would be inappropriate to compare the O&M costs of a facility that excludes disposal costs with others that include this information. The listed cost information is provided as an approximate range of costs for specific facilities. Costs for implementing RO may be very different from those listed here. A thorough cost analysis of design parameters for specific locations would be required for accurate cost estimation. The information gathered through the questionnaire includes detailed costs associated with the unique case studies included in this analysis. A detailed discussion of treatment costs is included below in Section 6 Treatment Cost Analysis.

Table 7. Selected costs* of reverse osmosis systems for nitrate removal.

System Flow**	< 0.5 MGD	0.5 – 5 MGD	5+ MGD
Annualized Capital Cost ($/1000 gal)	3.51 – 5.17 [1, 2]	1.00 – 1.30 [3, 4]	0.95 [3]
O&M Cost ($/1000 gal)	1.46 – 16.16 [1, 2]	1.22 – 2.01 [3, 4]	1.63 [3]
Total Annualized Cost ($/1000 gal)	5.73 – 19.70 [1, 2]	2.52 – 3.21 [3, 4]	2.58 [3]

*Costs have been adjusted to 2010 dollars with 7% interest over 20 years, unless indicated otherwise.
**When available, costs are based on actual system flow rather than design capacity.
[1] Walker (N.D.), costs not adjusted to 2010 dollars. [2] Personal communication with representatives of two small water systems (2010). [3] Conlon et al. (1995). [4] Cevaal et al. (1995).

3.2.3 Reverse Osmosis - Selected Research

Much research has focused on RO; Table A.2 of the Appendix is a list of recent studies relevant to nitrate removal from potable water and several examples of RO application. Current RO research focuses on improvements of membranes and waste management, and decreasing energy use.

3.2.4 Reverse Osmosis - Summary of Advantages and Disadvantages

A summary of advantages and disadvantages of RO in comparison with other treatment options is listed in Table A.6 of the Appendix. Advantages of RO include high quality product water, multiple contaminant removal, desalination (TDS removal), feasible automation, and application for POU applications. According to Elyanow & Persechino (2005), in their comparison of RO and EDR, "…the best economical choice for small capacity systems (<110 gpm or <25 m^3/hr) are simple RO plants, which have less electrical and hydraulic complexity than EDR and other technologies" (p. 7). In waters where salinity is a problem, RO can be better suited than IX due to the ability to remove multiple contaminants (including trihalomethane formation potential precursors (THMFPs)) (Cevaal et al. 1995).

Disadvantages of RO include high capital and O&M costs, membrane fouling susceptibility, high pretreatment and energy demands, and potentially large waste volume (lower water recovery) requiring disposal. The high cost of disposal from inland locations can result in RO treatment becoming cost prohibitive. Howe (2004) presents several alternatives to conventional disposal measures of RO waste brine, including reuse for industrial processes, processing (e.g., for salt production), or use in energy generation ("solar brine pond").

3.2.5 Reverse Osmosis - Improvements and Modifications

Process Modification

High Efficiency Reverse Osmosis (HERO™) is a patented multi-step process enabling increased water recovery (greater than 90%) and minimizing cleaning requirements. This process limits scaling by incorporating hardness reduction, CO_2 stripping, and pH adjustment (GE 2010b). Raw water is subjected to intensive pretreatment before passing through the RO membranes as follows (Engle 2007):

- Weak acid cation exchange (WAC) is used to remove hardness ions,
- CO_2 stripping is used to remove carbonate and increase pH, and
- Base addition is used to increase the pH to a level of 10.5.

An example flow diagram is illustrated in Figure 9.

With such pretreatment, water fed to the RO membranes is softened and pH is adjusted high enough to significantly increase the solubility of silica. The high chemical usage and multiple steps result in a more complicated process than conventional RO. However, benefits include increased water recovery, decreased waste volume, and the ability to treat severely impaired and poor quality source water containing multiple contaminants (Engle 2007).

The HERO™ process was initially designed to produce ultra-pure water for use in electronics applications and was patented by Debasish Mukhopadhyay with licensing rights for different applications (Engle 2007). The HERO™ process has been implemented for drinking water treatment in the small community of Yalgoo, Australia, to produce high quality drinking water from brackish groundwater high in silica and nitrate (Water Corporation 2007; Water Corporation 2009; Thomson et al. 2009). Higher removal rates result in decreased waste volume. Using the HERO™ process in Yalgoo, waste volumes are as low as "one-tenth of a conventional plant's concentrated brine residue for disposal, eliminating the need for big evaporation ponds" (Water Corporation 2007, p. 8).

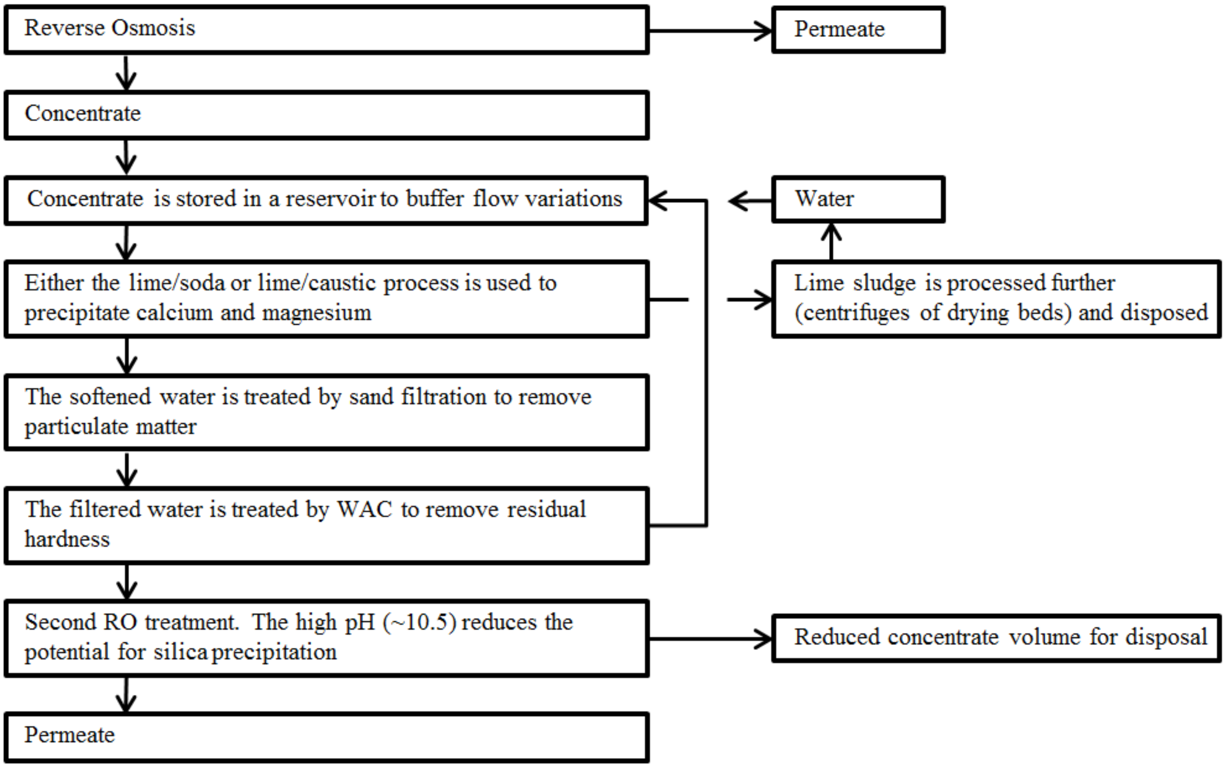

Figure 9. Flow chart of the HERO™ process. (Source: reproduced with permission, Central Arizona Salinity Study 2006.)

Membrane Modification – Low Pressure Membranes

Research and development in membrane technology has resulted in the emergence of Ultra-Low Pressure Reverse Osmosis (ULPRO). In contrast to the high pressures required for conventional RO, use of ULPRO membranes allows for lower operating pressures and improved flux rates. Energy demands can be reduced due to lower operating pressures; however, pretreatment practices to prevent membrane scaling and fouling are similar to those necessary for conventional RO membranes (Drewes et al. 2008). ULPRO membranes are available from several manufacturers. Operating pressures are in the range of 50 to 125 psi, while the pressures required for conventional RO membranes can be over 200 psi (Excel Water 2007; Drewes et al. 2008; Koch Membrane Systems 2008). Drewes et al. (2008) compared the performance of ULPRO membranes and conventional RO membranes, wherein pretreatment included nano-filtration for both RO options. Findings indicate that the ULPRO membranes included in the study were capable of successfully removing nitrate and multiple additional contaminants to potable water standards. “With regard to operating costs, operating pressure is the only TMG [ULPRO membrane] operating parameter considered to deviate from the benchmark ESPA2 [conventional RO membrane] membrane. Pretreatment requirements and recovery rate were the same. Electrical consumption will be directly proportional to the required operating pressure” (Drewes et al. 2008, p. 93). However, in the cost comparison between the two membranes, the benefits of lower operating pressures were overshadowed by the poor recovery of the ULPRO membranes after cleaning.

The authors suggest that the cleaning of the ULPRO membranes would need to be optimized for an improved cost comparison.

3.2.6 Reverse Osmosis - Case Studies

The following case studies provide detailed information on the design and operation of full-scale RO treatment plants used for nitrate removal. Chino I Desalter and Chino II Desalter are combination systems using both RO and conventional IX.

System Location: California
System Type: Community Water System

CASE #6

Treatment Type: Reverse Osmosis (RO)
Startup Date: 2002

Treatment Type	*Reverse Osmosis and Blending*
System Capacity	*120 gpm*
Raw Water Nitrate	*75 – 84 mg/L as nitrate* *17 – 19 mg/L as N*

System Description

A California water utility operates a system that has three groundwater supplies, one of which has nitrate at levels that exceed the MCL. The impacted well has nitrate concentrations that range from 75 to 84 mg/L as nitrate (17 to 19 mg/L as N) and has a typical production capacity of 100 gpm. In 2002, the utility implemented a blending program and installed an RO system for nitrate control and treatment.

In RO, raw water is forced through a semi-permeable membrane under pressure such that the water passes through, while contaminants are impeded by the membrane. The required pressure will be dependent on the concentration of solute in the feed water. The collected concentrate is high in nitrate and other rejected constituents (salts) and requires appropriate disposal. The extent to which the RO membrane removes constituents from the water is called the rejection rate. Rejection rates for sodium chloride and sodium nitrate can be as high as 98% and 93%, respectively (Elyanow & Persechino 2005).

The high nitrate supply is blended with one of the other two groundwater sources prior to RO treatment. The RO system reliably removes nitrate to below 35.4 mg/L as nitrate (8 mg/L as N) and the water delivered to consumers typically has nitrate levels below 13.3 mg/L as nitrate (3 mg/L as N). This system utilizes a leach field type system to land apply the RO concentrate.

Source Water Quality

- Nitrate - mg/L as nitrate (mg/L N)
  - Average - 75 (17)
  - Minimum - 80 (18)
  - Maximum - 84 (19)

- Co-contaminants
  - Fluoride - 3.3 mg/L
  - Arsenic
  - Radium

Treatment Technology Selection

RO was selected as the most appropriate treatment system as the technology can reliably remove nitrate in addition to the co-occurring contaminants that are present, specifically fluoride, arsenic, and radium.

Treatment System Parameters

- Design Capacity
    - 120 gpm maximum capacity
- Pretreatment
    - Anti-scalant (Hyposperse MCD 150)
- Treatment system footprint
    - Treatment system: 15’ x 30’
    - Total system footprint: 40’ x 100’
- RO System
    - System manufacturer: Aria™
    - Membrane manufacturer: Osmonics
    - Number of stages: 4
    - Number of RO elements per stage: 4
- Clean-in-place (CIP)
    - CIP frequency: Quarterly (4x/year)
    - Initiated when there is a 15% decrease in permeate flow or salt rejection or a 15% increase in trans-membrane pressure
    - CIP chemicals: Dilute phosphoric acid
- Water efficiency: 80%
- Monitoring
    - Laboratory samples
- Service life of membranes
    - Approximately 8 years

Residuals Management

The concentrate is disposed to an on-site leach field.

Technology Benefits and Drawbacks

Benefits

- Effectively removes nitrate and other co-contaminants
- On-site concentrate disposal
- Consistent operations

Drawbacks

- Energy intensive
- Relatively low water efficiency (80%)

Treatment Technology Costs

Treatment technology costs are not available for this system.

Operational Notes

The RO system has never had any extended unplanned shut downs or been shut down as the result of an alarm. There has been an exceedance of the fluoride MCL that occurred near the end of the useful life of the membranes. The membranes have since been replaced, resolving this operational issue.

References

Elyanow, D. and Persechino, J. (2005) Advances in Nitrate Removal. GE – General Electric Company, Water & Process Technologies. Accessed June 11, 2010 via < http://www.gewater.com/pdf/Technical%20Papers_Cust/Americas/English/TP1033EN.pdf>.

System Name: City of Brighton
System Location: Brighton, CO
PWSID: CO 0101025
System Type: Community Water System

CASE #7

Treatment Type: Reverse Osmosis (RO)
Questionnaire completed by: Dave Anderson, City of Brighton RO Chief Plant Operator
Startup Date: 1993

System Description

Treatment Type	*Reverse Osmosis*
System Capacity	*6.65 MGD (4,600 gpm)*
Raw Water Nitrate	*49 – 89 mg/L as nitrate* *11 – 20 mg/L as N*

The City of Brighton (City) utilizes six groundwater wells with production capacities ranging from 900 to 1500 gpm and one groundwater source which has been designated as groundwater under direct influence of surface water (GWUDI), as an emergency well, with a production capacity of 700 gpm. These seven sources are impacted by nitrate with average concentrations ranging from 49 to 89 mg/L of nitrate as nitrate (11 to 20 mg/L of nitrate as N). The City has implemented RO with blending. The design capacity of the RO system is 6.65 MGD of permeate at 80% recovery (1150 gpm/train). Green sand and cartridge filters (Graver) are used to treat the GWUDI source, primarily for the removal of manganese. (Additional sources operated by the City that are not impacted by nitrate are purchased treated surface water and additional GWUDI wells.)

Raw water enters the system with 40% of feed water bypassing the RO system and 60% of feed water passing to pretreatment. After anti-scalant addition, pretreated water is pressurized with boost pumps and passed to the RO skids. Waste concentrate exits the system for disposal and post-treatment of the permeate includes CO_2 stripping and the addition of chlorine and caustic. Post-treated water is blended with bypassed water and sent to storage and ultimately distribution.

Source Water Quality

- Nitrate - mg/L as nitrate (mg/L N)
    - Average - 46.9 to 90.3 (10.6 to 20.4)
    - Minimum - 20.02 to 69.97 (4.52 to 15.8)
    - Maximum - 78.8 to 112.9 (17.8 to 25.5)
- Co-contaminants
    - TDS: 580 to 1000 mg/L, RO TDS ~34 mg/L, Finished TDS ~280 mg/L
    - Fluoride: 1.3 mg/L
    - TOC: < 2 mg/L
    - Hardness: 370 – 480 mg/L as $CaCO_3$ (historically, Cevaal et al. 1995)

Treatment Technology Selection

IX and EDR were also considered and pilot tested prior to installation of the RO system. RO was selected due to nitrate levels and hardness. IX could have been less costly; however, the lower salt levels in RO concentrate make it possible to discharge waste to the South Platte River. "By selecting RO, the City hoped to actually reduce the salt load on the river with RO since many Brighton residents currently using home ion-exchange softening units would no longer use them" (Cevaal et al. 1995, p. 102). Biological treatment is also being explored for the treatment of nitrate in the waste brine.

Residuals Management

The waste concentrate is continuously discharged via a brine line to the South Platte River. Biological treatment is being explored for the treatment of the waste concentrate. The biological system would be located on the West side of the RO treatment system and would allow for reduction of nitrate in the waste stream. As mentioned above, the use of RO rather than ion exchange was an effort to decrease salt loading to the South Platte River. Historically, Brighton residents used in-home ion exchange units to soften water.

Technology Benefits and Drawbacks

Benefits	*Drawbacks*
• Consistent treatment performance	• Constant generation of waste stream
• Ease of operation	• High power consumption

Treatment System Parameters

- Design Capacity
    - 6.65 MGD of permeate at 80% recovery (1150 gpm/train)
- Pretreatment/Post-treatment
    - Anti-scalant:
King Lee Technologies
Pre Treat Plus 0100 phosphonate
    - 5 micron cartridge filters
        - 2.5 inch diameter
        - 90 day replacement
    - pH adjustment
        - caustic soda (NaOH)
        - Air stripping (CO_2 removal)
- Treatment system footprint
    - Treatment system:
11,000 sq. ft. at installation
(Cevaal et al. 1995)
- Number of contactors
    - 2 stages, 5 trains
    - RO elements/stage
        - 36 x 18 array
        - 6 membranes/vessel
        - 324 total
- Max. Concentration goal for delivered water
    - 35.4 mg/L as nitrate (8 mg/L as N), (always produce lower)
- Rejection Rate
    - 95 – 98% rejection
    - Nitrate goal (before blending):
~4.4 mg/L as nitrate (~1 mg/L as N)
- Flux rate of the RO membranes
    - 13 gpd/sf
- System Manufacturer
    - Hydranautics and Hydrocode
- Membrane Type
    - CPA2 (no others used in past)
- Membrane Life
    - Unknown, none have required replacement (5 yrs. ago, the manufacturer said the membranes should last 3 more yrs.).
- Membrane Cleaning
    - Clean in Place initiated by time rather than decrease in flux. Every ~157 million gals treated (~2x/yr)
    - Chemicals:
Nalco Product and Citric acid
- Waste
    - Discharge via Brine line to South Platte River
    - Recovery Rate: 80%
- Monitoring
    - Ion chromatography
        - At Source
        - At Point-of-Exit
    - Grab ISE (HACH)
        - At Blending Point
        - At Point-of-Exit
    - Testing once per year is required for compliance

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)	
Operating and Monitoring Equipment:	$8,253,000 (1993) 4MGD RO facility
Annual O & M Costs (Total with explanation or component costs)	
Total:	$2,873,293.00
Membrane:	0
Membrane Disposal:	0
Brine Disposal or Treatment:	0
Chemicals:	Approx. $100,000 year
Power:	Approx. $210,000 year for RO
Labor (Hours per Year):	10 hr/day, 7 day/wk
2 MGD Thornton treated:	$3.60/1000 gallons
COMPLETE Cost (including treatment, distribution, everything):	$3.16/1000 gallons

Additional Information

- The level of certification required for plant operators is Colorado A treatment.
- During a power outage, there is a pause before the generators start. This requires a manual restart of the system.
- This system has never produced water exceeding the nitrate MCL and has never had an unplanned shutdown exceeding one week.
- The major benefit of the RO system is the rejection rate allowing for removal of regulated contaminants.
- The most significant disadvantages are the high power consumption and the continuous brine discharge.
- The operator also noted that there has been a decreasing trend in nitrate levels in their sources.

References

Cevaal, J.N., Suratt, W.B., and Burke, J.E. (1995) Nitrate removal and water quality improvements with reverse osmosis for Brighton, Colorado. *Desalination*, **103**, 101 – 111.

Anderson, D. (2010) In-person interview and tour of the facility. July 19, 2010.

Anderson, D. (2010) Completed questionnaire. September 15, 2010.

System Name: Western Municipal Water District - Arlington Desalter
System Location: Riverside, CA
PWSID: CA3310049
System Type: Community Water System

CASE #8

Treatment Type: Reverse Osmosis (RO) and Blending
Questionnaire completed by: Joseph Bernosky, Director of Engineering
Startup Date: 1990 strictly for desalting, Upgraded to drinking water treatment in 2002

System Description

Treatment Type	*Reverse Osmosis and Blending*
System Capacity	*6.6 MGD (4,600 gpm)*
Raw Water Nitrate	*44 – 89 mg/L as nitrate* *10 – 20 mg/L as N*

The Western Municipal Water District (District) operates a system comprised of seven wells, five of which contain nitrate above the MCL. Three of the nitrate impacted wells are treated by a 6.6 MGD RO facility. The permeate, or treated water, from the RO system is blended with the remaining two wells prior to distribution. The RO and blending facilities are collectively referred to as the Arlington Desalter. Approximately 60% of the total flow is treated by the RO system and the remaining 40% is blended with the treated water. The District targets a nitrate concentration of 22 mg/L as nitrate (5 mg/L as N) in the distribution system.

The Arlington Desalter facilities were originally installed in 1990 to address the salt imbalance in the Upper Santa Ana Watershed. High salinity waters withdrawn from the South Arlington Basin were treated by the Arlington Desalter and subsequently discharged to the Santa Ana River for downstream use (and downstream drinking water treatment). The system was upgraded to a drinking water treatment facility in 2002 with the addition of disinfection, a clear well and a pump station used to pump drinking water into the distribution system for the city of Norco, CA.

Source Water Quality

- Nitrate - mg/L as nitrate (mg/L N)
    - Average - 75 (17)
    - Minimum - 44 (10)
    - Maximum - 89 (20)

- Co-contaminants
    - TDS - 1200 mg/L

Treatment Technology Selection

Due to the original intent of the system and the 2002 conversion for drinking water production, no other technologies were pilot tested prior to the installation of the RO and blending facilities. However, biological treatment of RO bypass water was recently pilot tested with full-scale implementation anticipated. A case study about this fixed bed biological pilot study is listed separately.

Treatment System Parameters

- Design Capacity: 6.6 MGD
- Pretreatment/Post-treatment
    - Anti-scalant: Y2K Anti-scalant
- Treatment system footprint
    - Treatment system: Approximately 7,500 ft^2 excluding clear well and pump station
- Number of Stages
    - Stage 1: 36 vessels each with 6 units
    - Stage 2: 12 vessels each with 6 units
- Max. Concentration goal for delivered water
    - 22 mg/L as nitrate (5 mg/L as N)
- Water Recovery
    - Original design: 75 – 76%
    - Current: 80%
- Flux rate of the RO membranes
    - 16 gpd/ft^2
- System Manufacturer
    - Hydranautics
- Membrane Type
    - Koch HR400
- Membrane Life
    - > 10 years
- Membrane Cleaning
    - Occurs 2 times per year
    - Chemicals: Low pH solution, hydrofluorosilic acid, high pH solution
- Waste: The RO concentrate is disposed of offshore via the Santa Ana Regional Interceptor (SARI) brine line
- Monitoring
    - Ion chromatography
    - Online nitrate analyzers

Residuals Management

Waste is discharged to the Santa Ana Regional Interceptor (SARI) (Brine Line). The SARI line prevents degradation of natural waters caused by increased salinity. Managed by the Santa Ana Watershed Project Authority (SAWPA), the SARI line is a dedicated interceptor line built to help users meet discharge requirements. In addition to the Arlington Desalter, the SARI line is used by other dischargers including industrial and domestic sources. The District's contribution to the total flow of the SARI line is approximately 5%. The SARI line carries water to the Orange County Sanitary District for wastewater treatment with ultimate offshore discharge.

Having access to the SARI line for brine disposal is a benefit of this system; however, there have been complications with the SARI line. Tremendous scaling problems in Arlington Desalter's reach of the SARI line have resulted in the need to address calcium carbonate buildup and to consider additional cleanout points. Due to multiple discharge sources, the SARI line combines waters having very different water quality characteristics. Reactions within the mixed water can vary based on water chemistry. Several cleaning and maintenance options are being considered.

Technology Benefits and Drawbacks

Benefits

- Reliable
- Access to SARI line for disposal
- The treatment potential of the pre-existing RO system has been maximized through conversion for drinking water treatment

Drawbacks

- Costly brine disposal
- Complications with SARI line (scaling, etc.)

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)	
Total Capital Costs:	Unavailable for initial installation in 1990
Annual O&M Costs (Total with explanation or component costs) (2009/2010)	
Total:	$2,931,228
Labor and Overhead:	$836,530
Outside Services:	$200,000
Emergency Repairs:	$25,000
General & Administrative:	$85,000
Vehicle and Equipment:	$9,000
SARI Fixed Cost:	$151,800
Materials and Supplies:	$5,500
Permits and Fees:	$27,000
SARI Variable Cost:	$470,000
Chemicals:	$150,000
Energy:	$971,398

Additional Information

Regarding water recovery, the original design recovery rate was 75 – 76%. With modifications to anti-scalant use, the recovery rate increased to 78% and then to the current operational water recovery rate of 80%. To further improve the water recovery rate, additional testing is anticipated which will require engineering work and a cost-benefit analysis.

Currently used membranes are 10 years old and are still performing adequately with respect to operational parameters (flux rate, rejection rate, etc.). The District has budgeted for membrane replacement in this fiscal year; however, due to adequate performance, the current membranes may be used for an additional year. Membranes are actually attaining better water recovery than manufacturer specifications. Membrane life is also exceeding initial expectations.

Sources*

Bernosky, J. (2010) Personal communication. November 5, 2010.
Bernosky, J. (2010) Completed questionnaire. October, 2010.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

System Name: Chino Basin Desalter Authority (CDA) – **Chino I Desalter**
System Location: Chino, CA
PWSID: CA31610075
CASE #9
System Type: Community Water System

Treatment Type: Reverse Osmosis (RO) and Ion Exchange (IX)
Questionnaire completed by: Timothy Mim Mack, CDA Coordinator, City of Ontario
René Cruz, Engineering Project Manager, Project Partners Inc., serving the (CDA).
Startup Date: RO in 2000, IX added in 2005

Treatment Type	*Reverse Osmosis,* *Ion Exchange* *And Blending*
System Capacity	*RO: 4940 gpm* *IX: 3400 gpm*
Raw Water Nitrate	*147 – 303 mg/L as nitrate* *33 – 68 mg/L as N*

System Description

The Chino Basin Desalter Authority (CDA) in southern California is a conglomerate of the following agencies: Inland Empire Utilities Agency (IEUA), Jurupa Community Services District (JCSD), City of Chino, City of Chino Hills, City of Ontario, City of Norco, Santa Ana River Water Company (SARWC), and Western Municipal Water District (WMWD). The CDA drinking water treatment facilities include two desalters: Chino I Desalter (discussed here) and the Chino II Desalter (discussed in the next case study) to address high TDS levels as well as nitrate contamination. The Chino I Desalter operates 14 source wells, 11 of which have raw nitrate levels well above the MCL. Treatment consists of a combination of RO, conventional anion exchange and blending. Sixty percent of total flow is treated with RO, 27% with IX and 13% passes only through VOC/Air-stripping prior to blending. The RO system was installed in 2000 and the IX system was added in 2005.

Source Water Quality

- Nitrate - mg/L as nitrate (mg/L N) (of nitrate impacted wells)
    - Average: ~147 – 303 (~33 – 68)
    - Minimum: ~114 – 289 (~26 – 65)
    - Maximum: ~161 – 351 (~36 – 79)
- Co-contaminants
    - TDS: 1100 mg/L

Treatment Technology Selection

RO and IX were selected because, combined, they were determined to be the best mode of technology to adequately treat the high-TDS, high-nitrate source water. No other technologies were pilot tested or considered prior to the installation of the system.

Treatment System Parameters

- Design Capacity
    - RO: 4940 gpm
    - IX: 3400 gpm
- Pretreatment
    - Anti-scalant: threshold inhibitor
    - Filtration: 1 micron pre-filters
    - pH adjustment: sulfuric acid
- Post-treatment
    - pH adjustment: sodium hydroxide
- Treatment system footprint
    - RO system: 143’ X 80’
    - IX system: 190’ X 60’
- Number of contactors
    - 4 trains, 2 stages/train
    - RO elements/stage
        - stage 1: 196 elements
        - stage 2: 98 elements
- Ion exchange pressure vessels
    - Number of vessels: 4
    - Diameter of vessels: 12’
    - Height of vessels: 11’
- Design Loading Rate: 1.66 gpm/ft^2
- Max. nitrate concentration goal for delivered water: 36 mg/L as nitrate (8.13 mg/L as N)
- Nitrate concentration goal for the treatment system (before blending): 10 mg/L as nitrate (2.25 mg/L as N)
- RO recovery rate: 80%
- RO membrane flux rate: 0.9 gfd/psi
- System Manufacturer
    - RO: Code-line
    - IX: Hungerford and Terry
- Membrane Type: Dow 400 BW-30
- Membrane Life: 5 years
- Membrane Cleaning
    - Flux decrease initiates CIP
    - Every 6 month
    - Chemicals: for pH adjustment based on manufacturer recommendation
- Resin Type: Rohm and Hass Amberjet 4400 CL SBA
- Volume treated prior to regeneration
    - 700,000 gallons
    - Regeneration once every 12 hrs
- Salt consumption: 75 tons per week
- Volume of brine/backwash
    - 53,000 gallons
    - 92.4% water efficiency
- Resin life: Has not been replaced (online for 5 years)
- Monitoring
    - Online nitrate analyzers
        - Treatment train
        - Blending point
        - POE
    - Laboratory samples
    - Quarterly testing for NDMA

Residuals Management

Concentrate/brine is discharged into a regional brine line called the Inland Empire Brine Line (IEBL) and formerly known as the Santa Ana Regional Interceptor (SARI).

Technology Benefits and Drawbacks

Benefits

- RO provides better removal
- IX is inexpensive
- IX has very low energy demands

Drawbacks

- RO is expensive
- High waste rate of RO
- IX does not address TDS
- Resin replacement will be costly

Treatment Technology Costs

Capital Costs (Based on projected costs in 2004)	
Treatment Plant Expansion Total (5000 afy expansion):	$6,379,530
Ion Exchange Treatment (4.9 MGD):	$4,031,900
Onsite Modifications:	$1,735,000
SARI Discharge Upgrades & Storm Drain:	$612,630
Additional SARI Capacity Purchase (not included in above total):	$4,140,000
O & M Costs (Based on CDA 2010/11 Budget, for complete plant operation, not just the treatment system)	
Total:	$7,496,315
Chemicals:	$662,257
Electricity, Plant Total:	$2,843,000
Operating Fees:	$1,353,439 (includes SARI fees, permits and other fees)
Labor ($):	$1,370,698

Additional Information

- The RO treatment system is described as falling short of expectations with respect to the high waste rate. 15% of all incoming water is sent to the brine line and delivered to a treatment plant outside of the local watershed at the Orange County Sanitation District.
- Plant shutdown has been required in the past due to high or low pressure, high nitrate, and high TDS.
- In the event of insufficient treatment and the production of water in exceedance of an MCL, the plant has an MOV that closes automatically, sending water to a storm drain.

Sources*

Listed costs are based on:
Chino Basin Desalter Authority. (2005) Presentation: Chino I Desalter Expansion & Chino II Desalter Project Update.
Chino Basin Desalter Authority. (2010) Fiscal Year 2010/11 Budget Adoption.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

System Name: Chino Basin Desalter Authority (CDA) – **Chino II Desalter**
System Location: Mira Loma, CA
PWSID: CA3310083
CASE #10 System Type: Community Water System

Treatment Type: Reverse Osmosis (RO) and Ion Exchange (IX)
Questionnaire completed by: Timothy Mim Mack, CDA Coordinator, City of Ontario
René Cruz, Engineering Project Manager, Project Partners Inc., serving the (CDA).
Startup Date: 2006

System Description

Treatment Type	*Reverse Osmosis, Ion Exchange And Blending*
System Capacity	*RO: 4167 gpm IX: 2778 gpm*
Raw Water Nitrate	*70 – 224 mg/L as nitrate 16 – 51 mg/L as N*

The Chino Basin Desalter Authority (CDA) in southern California is a conglomerate of the following agencies: Inland Empire Utilities Agency (IEUA), Jurupa Community Services District (JCSD), City of Chino, City of Chino Hills, City of Ontario, City of Norco, Santa Ana River Water Company (SARWC), and Western Municipal Water District (WMWD). The CDA drinking water treatment facilities include two desalters: Chino I Desalter (discussed above) and the Chino II Desalter (discussed here) to address high TDS levels as well as nitrate contamination. The Chino II Desalter operates 8 source wells, all of which have raw nitrate levels well above the MCL. Treatment consists of a combination of RO, conventional anion exchange, and blending. The combined RO/IX system was installed in 2006.

Source Water Quality

- Nitrate - mg/L as nitrate (mg/L N) (of nitrate impacted wells)
    - Average: ~70 – 224 (~16 – 51)
    - Minimum: ~53 – 190 (~12 – 43)
    - Maximum: ~81 – 260 (~18 – 59)

- Co-contaminants
    - TDS

Treatment Technology Selection

RO and IX were selected because, combined, they were determined to be the best mode of technology to adequately treat the high-TDS, high-nitrate source water. No other technologies were pilot tested or considered prior to the installation of the system.

Treatment System Parameters

- Design Capacity
    - RO: 4167 gpm
    - IX: 2778 gpm
- Pretreatment
    - Anti-scalant: threshold inhibitor
    - Filtration: 1 micron pre-filters
    - pH adjustment: sulfuric acid
- Post-treatment
    - pH adjustment: sodium hydroxide
- Treatment system footprint
    - RO system: 30'X 188'
    - IX system: 30' X 188'
    - Total system: 60' X 300'
- Number of contactors
    - 3 trains
    - 48 vessels/train
    - RO elements/stage:
        - 7 elements per stage
- Ion exchange pressure vessels
    - Number of vessels: 4
- Design Loading Rate: 10.1 gpm/ft^2
- Max. nitrate concentration goal for delivered water: 25 mg/L as nitrate (5.7 mg/L as N) (goal), 35 mg/L as nitrate (7.9 mg/L as N) (max.)
- Nitrate concentration goal for the treatment system (before blending): 4 mg/L as nitrate (0.9 mg/L as N)
- RO membrane flux rate: 0.30 – 1.70 gfd/psi
- System Manufacturer
    - RO: PROTEC Bekaert
    - IX: Hungerford and Terry
- Membrane Type: Dow/Filmtec Model BW30-400
- Membrane Life: ~5 years
- Membrane Cleaning
    - Flux decrease initiates CIP
    - Every 6 months to 1X per year
    - Chemicals: King Lee (anti-scalant), high/low pH, Silica Cleaner
- Resin Type: Rohm and Hass Amberjet 4400 CL SBA
- Volume treated prior to regeneration
    - 0.8 – 1.4 MGD
    - Regeneration is based on nitrate levels
- Salt consumption: 50 tons per week
- Volume of brine/backwash
    - NA
- Resin life: NA
- Monitoring
    - Online nitrate analyzers
        - At source
        - Blending point
        - POE
    - Laboratory samples

Residuals Management

Concentrate/brine is discharged to an industrial sewer that drains to the Inland Empire Brine Line (IEBL). Waste is transported 45 miles to the Orange County Sanitation District.

Technology Benefits and Drawbacks

Benefits

- RO works well for nitrate removal
- IX is less expensive

Drawbacks

- RO is expensive
- IX does not accomplish contaminant removal as well as RO

Treatment Technology Costs

Capital Costs (Based on projected costs in 2004)	
Treatment Plant Total (10,400 AFY):	$19,171,837
Ion Exchange Treatment:	$4,346,900
Chino II Desalter:	$14,284,500
RO Membranes:	$540,437
Ion Exchange Land (not included in above total):	$1,730,138
Ion Exchange SARI Fee (not included in above total):	$10,105,000
O & M Costs (Based on CDA 2010/11 Budget, for complete plant operation, not just the treatment system)	
Total:	$6,111,799
Chemicals:	$615,000
Electricity – Plant Total:	$2,331,000
Operating Fees:	$706,154 (includes SARI fees, permits and other fees)
Labor ($):	$1,133,615

Additional Information

- Operator certification levels range from T-3 to T-5.
- Plant shutdown has been required in the past due to chemical pump failure and a high clearwell.
- Overall the combined system is described as working well but at a high price.

Sources*

Listed costs are based on:
Chino Basin Desalter Authority. (2005) Presentation: Chino I Desalter Expansion & Chino II Desalter Project Update.
Chino Basin Desalter Authority. (2010) Fiscal Year 2010/11 Budget Adoption.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

3.3 Electrodialysis (ED/EDR/SED)

The use of electrodialysis (ED), including electrodialysis reversal (EDR) and selective electrodialysis (SED), in potable water treatment (Figure 10) has increased in recent years, offering the potential for improved water recovery, the ability to selectively remove nitrate ions, and the minimization of chemical and energy requirements (Kneifel & Luhrs 1988; Kapoor & Viraraghavan 1997; Hell et al. 1998; Koparal & Ogutveren 2002; Midaoui et al. 2002; Sahli et al. 2008; Banasiak & Schafer 2009).

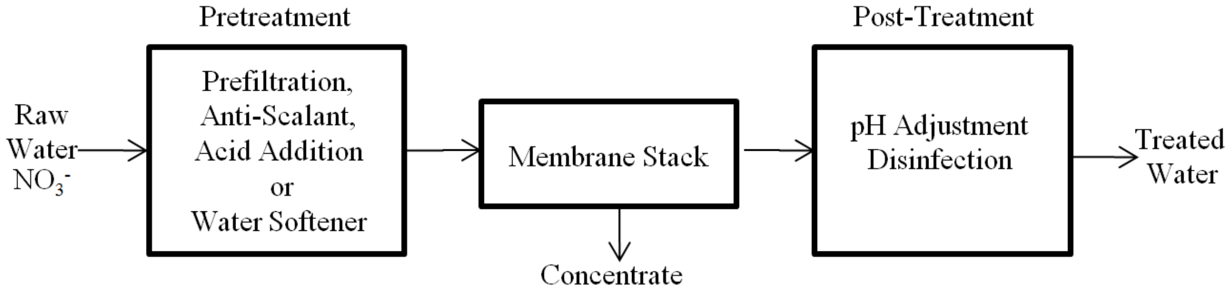

Figure 10. Electrodialysis reversal schematic.

Nitrate removal is accomplished by passing an electrical current through a series or stack of anion and cation exchange membranes, resulting in the movement of ions from the feed solution to a concentrated waste stream. Illustrated in Figure 11, nitrate ions (and other anions) move through the anion exchange membrane toward the anode. Continuing toward the anode, nitrate is rejected by the anion-impermeable cation exchange membrane and trapped in the recycled waste stream. Cations can be removed in a similar manner, migrating toward the cathode through the cation exchange membrane and rejected by the cation-impermeable anion exchange membrane. Nitrate selective membranes allow for treatment without significantly altering the balance of other ions in the water.

The electrical current is passed through the system with the migration of ions across the membranes. For every anion that leaves a compartment, a cation of equivalent charge also leaves, maintaining the charge balance in each compartment. Across the system, the flow of electrons, moving from the cathode to the anode (negative to positive), is governed by the movement of ions through the membrane stack and by the reactions in the electrode compartment. Small levels of gaseous byproducts must be removed. Electrolysis of water generates oxygen at the anode and hydrogen gas at the cathode and chloride can be reduced at the anode, producing chlorine gas (AWWA 1995). The electrode compartment is rinsed to restore ions for current transfer and to remove unwanted reaction products.

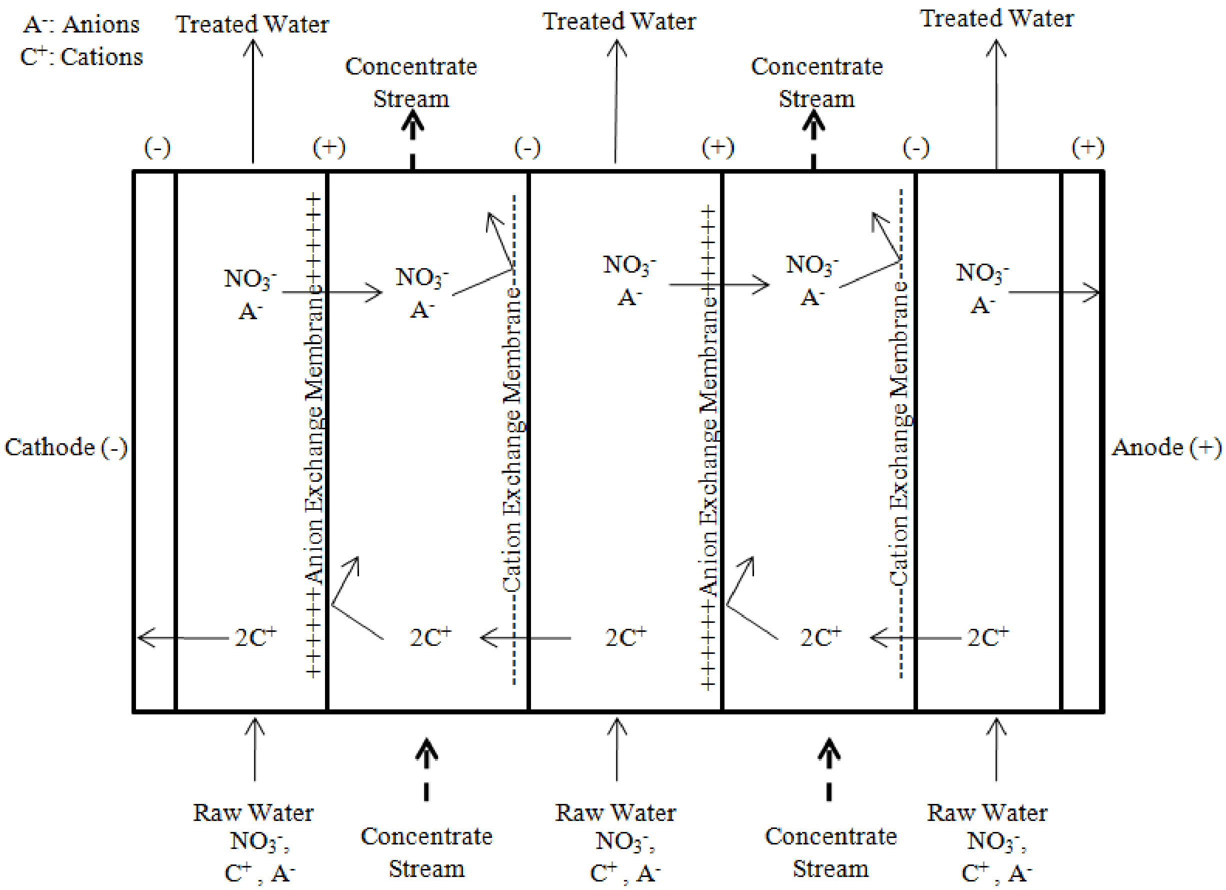

Figure 11. Illustration of electrodialysis membrane stack.

Requiring constant electrical current and low-pressure water, ED has inherent energy demands. However, voltage adjustment enables selective demineralization. "Plants can be designed to remove from 50 to 99 percent of source water contaminants or dissolved solids. Source water salinities of less than 100 mg/L up to 12,000 mg/L TDS can be successfully treated to produce finished water of less than 10 mg/L" (AWWA 1995, p. 7).

A detailed case study of an EDR plant in Spain is included in section 3.3.6 Electrodialysis - Case Studies.

3.3.1 Electrodialysis - Design Considerations

Table 8 summarizes key design considerations in the use of ED for nitrate removal from potable water.

Table 8. Summary of design considerations for electrodialysis/electrodialysis reversal.

Membranes	• Use of anion and cation exchange membranes • Selective membranes ○ Monovalent versus multivalent • Consider water recovery and frequency of cleaning
Pretreatment	• Lower pretreatment requirements because this is not direct filtration • EDR systems can avoid or limit chemical use • Prevention of scaling and fouling ○ Filtration to remove suspended solids ○ Treatment for iron and manganese removal ○ Water softening or use of anti-scalants or acid to prevent scaling
Post-Treatment	• pH adjustment to avoid corrosion (if acid used to prevent scaling) • Disinfection
Chemical Usage	• Possible pH adjustment (acids and bases) • Possible anti-scalants • Possible cleaning chemicals
O&M	• Highly automated • Frequency of membrane cleaning depends on water quality and membrane used ○ Polarity reversal (electrodialysis reversal) multiple times per hour minimizes fouling ○ ED systems can require weekly cleaning • Management of chemicals and pre-filtration system ○ Including electrode compartment rinse solution • Waste storage and disposal • High monitoring demands • Potentially higher operator demand than IX and RO, due to system complexity
System Components	• Maximize water recovery while minimizing energy use • Key aspects of the system are pretreatment requirements, number and configuration of electrodialysis stacks and stages, membrane selection and configuration, operating voltage and pressure, reversal frequency (for EDR), gas venting of anode and cathode compartments, disinfection, "brine loop, electrode rinse loop, concentrate discharge, and dosing station" (Hell et al. 1998, p. 178).
Waste Management and Disposal	• Concentrate disposal of greatest concern for inland systems ○ Close proximity to coastal waters is beneficial for brine/concentrate disposal • Management options include sewer or septic system, reuse for irrigation, drying beds, trucking off-site, coastal pipeline, deep well injection and advanced treatment • Disposal options can be limited by waste brine/concentrate water quality (e.g., volume, salinity, metals, and radionuclides) • Optimization of recycling and treatment of waste concentrate
Limitations	• Need to prevent membrane scaling and fouling ○ Hardness, iron, manganese, and suspended solids • Disposal of waste concentrate • High system complexity

Water Quality

Membrane life, cleaning frequency, and pretreatment needs are dependent on feed water quality. Pretreatment may be needed for iron levels above 0.3 mg/L, manganese levels above 0.1 mg/L, and hydrogen sulfide levels exceeding 0.3 mg/L (WA DOH 2005). Specifications for an example EDR system from GE indicate feed water turbidity levels should be < 0.5 nephelometric turbidity units (NTU) with typical TDS levels between 100 and 3,000 mg/L (maximum 12,000 mg/L) (GE 2008). SDI limits are generally higher for EDR than for RO, with typical limits of 12 and 4 – 5, respectively (Elyanow & Pereschino 2005). Softening may be necessary to reduce hardness, while pre-filtration diminishes suspended solids. The potential for scaling increases with increasing TDS and is exacerbated by increased solids precipitation with higher water recovery goals. To minimize fouling/scaling, membranes can be treated with anti-scaling chemicals and cleaned with acid (AWWA 1995). However, in comparison with other membrane processes, fouling is minimal because the membrane is subjected to the transfer of ions (directed by the electrical current), rather than the transfer of the entire feed stream. Unfortunately, because ED does not serve as a filter (the water does not pass through the membrane), ED fails to remove microbial contamination (AWWA 1995). Pre-filtration in pretreatment and disinfection in post-treatment address these concerns.

To further minimize fouling and thus the need for chemical addition, the polarity of the system can be reversed with electrodialysis reversal (EDR). By reversing the polarity (and the solution flow direction) several times per hour, ions move in the opposite direction through the membranes, minimizing buildup and the need for chemical addition to control scaling. Biological fouling concerns are lower than other separation processes due to development of membranes that are "more organic resistant and chlorine tolerant" (Elyanow & Persechino 2005, p. 8). ED depends on the transfer of an electrical current and is therefore more efficient when used for brackish waters. In low conductivity feed waters, the ion removal efficiency declines. In contrast to conventional RO, EDR is unaffected by silica.

System Components and Site Considerations

ED and EDR systems are operated in stages. Water recovery can be improved with stages operated in series while capacity can be increased with stages operated in parallel. Key aspects of the system are pretreatment requirements, the number and configuration of electrodialysis stacks and stages, membrane selection and configuration, operating voltage (based on desired removal), reversal frequency (for EDR), gas venting of the anode and cathode compartments, disinfection, "brine loop, electrode rinse loop, concentrate discharge, and dosing station" (Hell et al. 1998, p. 178).

The membranes used in ED/EDR are anion and cation exchange membranes. Membranes have been designed for selective removal based on valency (monovalent versus multivalent) to screen for particular constituents (AWWA 1995). Alternating different selective membranes in the membrane stages can avoid precipitation in the concentrate stream. For example, one stage can remove calcium and a second stage can remove sulfate (to an alternate concentrate stream), this prevents calcium sulfate precipitation (AWWA 1995).

Residuals Management and Disposal

Waste management requirements are similar to RO and IX; however, the burden of disposal in ED/EDR systems is not as significant due to higher water recovery, selective removal, and the lack of direct filtration (Reahl 2006). Disposal options include sewers, septic systems, drying beds, off-site trucking, coastal pipeline, deep well injection, reuse for irrigation, and advanced treatment. Important water quality characteristics of the concentrate (e.g., volume, salinity, metals, and radionuclides) can affect the feasibility and costs of disposal options. Several combined configurations of interest are discussed in Section 3.6 Brine Treatment Alternatives and Hybrid Treatment Systems.

Maintenance, Monitoring, and Operational Complexity

Although ED/EDR systems are amenable to automation, operator demands can be higher than other separation processes (AWWA 1995). While ED systems have greater pretreatment demands and can require membrane cleaning once a week, EDR systems minimize pretreatment demands and scaling issues, but can still have higher maintenance demands than RO, due to the complexity of the system (Kapoor & Viraraghavan 1997). Appropriate gas venting is important to avoid hazardous conditions (AWWA 1995). Membrane life will depend on water quality and pretreatment measures. However, due to the lack of direct filtration and operation under low pressure, membranes are long lasting, and do not require frequent replacement.

3.3.2 Electrodialysis - Cost Considerations

For the efficient operation of an ED system, the fundamental objective is to maximize water recovery with the minimum amount of energy and chemical usage, while meeting necessary potable water guidelines. Factors affecting system cost include facility size (how much water), source water quality (including nitrate concentration and other contaminants), target effluent nitrate concentration, and disposal options.

Capital costs for ED/EDR systems include land, housing, piping, storage tanks, O&M equipment, cation and anion exchange membranes, preliminary testing (pilot studies), permits, and training. O&M costs include membrane replacement, membrane disposal, concentrate disposal or treatment, chemical use (limited: anti-scalant, etc.), repair, maintenance, power, and labor.

Very little published cost information from existing ED systems used for nitrate removal is available in the literature, due to the limited number of full-scale systems. Costs of ED systems are most comparable to RO. However, in some instances, ED can be the less costly choice due to the greater pretreatment and post-treatment demands (higher chemical use and post-treatment pH adjustment) of RO (Reahl 2006). EDR can be chosen over RO when high water recovery is a priority, especially if land must be purchased for concentrate ponds. “New technology has also reduced the capital and operating cost of EDR nitrate removal by increasing the hydraulic efficiency of the EDR stacks and pumping system” (Elyanow & Persechino 2005, p. 8). Costs listed here have been adjusted to 2010 dollars, unless indicated otherwise. In a technical paper from GE Water & Process Technologies (the primary supplier of EDR systems in the U.S.), Werner & Gottberg (2005) present O&M costs of an electrodialysis plant in Suffolk, VA (not specifically for nitrate treatment) (Table 9). It is unclear how disposal costs were included in this study. High water recovery in comparison with other removal processes and disposal of waste concentrate to a nearby estuarine tributary would maintain low disposal costs (Werner & Gottberg 2005). According to Ameridia, the American division of Eurodia Industrie (a manufacturer of EDR systems), the capital investment for a nitrate treatment EDR unit for a ~0.5 MGD system (in 2005) was $475,000 or $0.94 per gallon of daily capacity ($559,653 or $1.11 in 2010 dollars, respectively) (Ameridia). However, additional capital costs are likely not included in this figure. A detailed discussion of treatment costs is included in Section 6 Treatment Cost Analysis.

Table 9. Sample EDR O&M costs (from Werner & Gottberg 2005).

O&M Category	EDR (/1000 gallons)[1]
Fixed	$0.72 ($1.07 adjusted)
Professional Services	$0.06 ($0.09 adjusted)
Chemicals	$0.02 ($0.03 adjusted)
Utilities	$0.21 ($0.31 adjusted)
Maintenance	$0.17 ($0.25 adjusted)
Membrane Replacement	$0.23 ($0.34 adjusted)
Production (1997)	827,339,440 gallons
Total O&M Cost	$1.41 ($2.09 adjusted)

[1] Costs adjusted from 1998 dollars to 2010 dollars.

The listed cost information is provided as an approximate range of costs for specific facilities. Costs for implementing ED may be very different from those listed here. A thorough cost analysis of design parameters for specific locations would be required for accurate cost estimation.

3.3.3 Electrodialysis - Selected Research

Much research on ED has focused on desalination applications. Table A.3 of the Appendix lists recent studies relevant to nitrate removal from potable water and several examples of ED application. Research is focused on the influence of co-contaminants on system performance and improvements in exchange membranes, including nitrate selectivity.

3.3.4 Electrodialysis - Summary of Advantages and Disadvantages

A summary of advantages and disadvantages of ED in comparison with other treatment options is listed in Table A.6 of the Appendix. Advantages of ED/EDR systems include low chemical usage, long lasting membranes, selective removal of target species, flexibility in removal rate (through voltage control), good water recovery rate, feasible automation, and multiple contaminant removal (Prato & Parent 1993; AWWA 1995; Hell et al. 1998; WA DOH 2005). With the ability to selectively remove multiple contaminants, ED/EDR systems can be used to address the following constituents: TDS, total chromium, chromium-6, arsenic, perchlorate, sodium, mercury, chloride, copper, sulfate, uranium, fluoride, nitrate/nitrite, iron, selenium, hardness, barium, bicarbonate, cadmium, and strontium (AWWA 1995 and GE 2010). Using current reversal, EDR offers additional advantages, improving system performance by “detaching polarization films, breaking up freshly precipitated scale or seeds of scale before they can cause damage, reducing slime formations on membrane surfaces, reducing problems associated with the use of chemicals, and cleaning electrodes with acid automatically during anodic operation” (AWWA 1995, p. 9, 10). Additionally, in comparison with RO systems, EDR can treat waters with higher SDI, silica, and chlorine levels (Elyanow & Persechino 2005).

Disadvantages of ED/EDR systems include the possible need for pretreatment to prevent membrane scaling and fouling, waste disposal, high maintenance demands, costs (comparable to RO systems), the need to vent gaseous byproducts, the potential for precipitation (especially for high water recovery), high system complexity, and limited manufacturers with U.S. experience (e.g., GE is the primary source of EDR systems for drinking water in the U.S.). Additionally, unlike RO, ED does not remove uncharged constituents in the water.

3.3.5 Modifications to Electrodialysis

Selective Electrodialysis (SED)

Since 1997, selective electrodialysis (SED), developed by Shikun & Binui, formerly Nitron, Ltd., has been successfully implemented throughout Israel, reducing national water costs by 55%. The manufacturer indicates that SED offers high water recovery (up to 95%), thereby minimizing waste volume (Nitron 2010). The SED system is accepted by the U.S. EPA as a nitrate treatment option for large plants (Nitron 2009). While similar to traditional ED processes, SED utilizes nitrate selective membranes which have been shown to increase operational performance when used for nitrate treatment.

The nitrate selective membranes used in the SED process have been shown to remove up to 70% of nitrate from solution. At the same time, sulfate ions and carbonate ions, which have a tendency to cause scaling issues in the concentrate stream of traditional ED/EDR and RO technologies, are more readily rejected by the nitrate selective membranes used in the SED process. As a result, the scaling potential is reduced in the concentrate stream. Since scaling problems are minimized, membrane cleaning frequency, maintenance costs, and down time are reduced compared to traditional EDR installations. Another important aspect of membrane selectivity is the energy efficiency of the process. Energy efficiency is related to the extent of ion transfer in ED/EDR and SED technologies. In traditional ED/EDR, energy use is less focused, resulting in the removal of many ions, including ions that do not need to be addressed. SED specifically targets nitrate ions, avoiding energy use for the removal of non-target ions and improving energy efficiency.

Pretreatment can be limited to filtration, energy efficiency is maximized due to low pressure operation (2 – 4 bars, ~30 – ~60 psi), chemical use is limited to concentrate treatment, and low maintenance demands are possible due to automation, remote monitoring and control, and infrequent cleaning. In the SED process there is no change in the pH of the product water. This avoids the need for pH adjustment or remineralization in post-treatment (Nitron 2009; Nitron 2010). Membranes are cleaned in place (CIP) for 1 hour every 4 – 6 months and membrane life is typically 7 – 10 years (Nitron 2010). Additional advantages of SED include constant membrane performance, no chemical contact with potable water, the simplicity of the system consisting of pre-filtration and membrane stacks (UV can be added for disinfection), and a small footprint (Nitron 2009b). Potential drawbacks of SED include the lack of full-scale application in the U.S. for nitrate removal from drinking water and, unlike RO, ED does not remove uncharged constituents in the water.

A detailed case study of the use of SED for nitrate removal at locations in Israel is included in the following section.

3.3.6 Electrodialysis - Case Studies

The following case studies provide detailed information on the design and operation of full-scale EDR and SED treatment plants used for nitrate removal.

System Name: Gandia EDR
System Location: Valencia, Spain
System Type: NA

CASE #11

Treatment Type: Electrodialysis Reversal (EDR)
Questionnaire completed by: GE Water & Process Technologies
Startup Date: 2007

Treatment Type	*Electrodialysis Reversal*
System Capacity	*2 systems at 4.7 MGD each*
Raw Water Nitrate	*80 mg/L as nitrate* *18 mg/L as N*

System Description

Gandia is a tourist area on the Mediterranean coast of Spain. The area sees peak demand during the summer months when the population almost triples. The Spanish Legislature released royal Decree 140/2003 which changed the nitrate limit to 50 mg/L as nitrate (11.3 mg/L as N). This new law required treatment of the existing system to achieve the new nitrate limits. Additionally, the existing well systems had deteriorated over time, forcing the municipality to find alternate wells to feed the community.

Upon analysis of the wells (old and new), it was determined that the nitrate levels were too high to meet the drinking water standard. The well samples had up to 80 mg/L as nitrate (18.1 mg/L as N). Treatment was necessary to produce acceptable levels of nitrate in the product water. An evaluation was conducted and EDR was selected as the technology of choice for the Gandia treatment plants. EDR offered high recovery while effectively reducing the nitrate levels below 25 mg/L as nitrate (5.6 mg/L as N).

EDR was piloted on the wells to verify the nitrate removals and operating cost estimates for power requirements and chemical consumption. The pilot study was successful, and the final systems were designed around 90% water recovery with the overall nitrate removal of 73%.

Source Water Quality

- Nitrate (mg/L N)
    - < 80 mg/L as NO_3^-

Treatment Technology Selection

EDR, Selective ED, and Reverse Osmosis were considered for treating the Gandia Wells. EDR was eventually selected for the high recovery and reduced operating costs compared to RO. SED was ruled out based on the high capital costs of the system.

Treatment System Parameters

- Design Capacity
    - 3,260 gpm
- EDR System
    - No. of modules: 4
    - Lines per module: 5
    - Stages per line: 2
- Water recovery rate
    - 94.3%

Water Quality Results

The table below summarizes the water quality of the raw water, finished water, concentrate stream, and the total waste from the Gandia EDR facility. Total waste includes concentrate blowdown, electrode waste, and off-spec product from the system.

Ion	Raw Water	Treated Water	Percent Removal	Concentrate Stream	Total Waste
Ca	82	24.9	70%	772.6	544.3
Mg	24	8.3	65%	213.7	151.0
Na	23	10	57%	180	128.1
K	1.0	0.3	70%	9.2	6.5
HCO_3	250	99.1	60%	2074.1	1471.2
SO_4	58	12.7	78%	605.2	424.4
Cl	29	7.5	74%	289.3	203.3
NO_3	60	16.6	72%	584.4	411.1
TDS	527	179.3	66%	4728.6	3339.8
pH	7.5	7.1		8.3	8.1

Technology Benefits and Drawbacks

Benefits

- EDR membranes are chlorine tolerant, providing means to control biological growth.
- Relatively low operational expenditures:
    - Membrane life expectancy is 15 years.
    - Low chemical consumption compared to other technologies.
    - Lower energy consumption compared to RO.
- High water recovery, small concentrate stream for disposal compared to other technologies.

Drawbacks

- Higher capital cost than RO.
- System footprint larger than competitive technologies.

Operating Costs

Capital costs for the EDR system were not provided.

O & M Costs (Total with explanation or component costs)		
	Unit	Cost
Labor:	$/1,000 gal	0.17
Energy:	$/1,000 gal	0.15
Maintenance:	$/1,000 gal	0.03
Chemicals:	$/1,000 gal	0.10
Consumables:	$/1,000 gal	0.19
Overhead:	$/1,000 gal	0.04
Total:	$/1,000 gal	0.67

Operational Notes

Operating costs were based on estimates prior to plant start-up. After four years of operating, the plant has not replaced any membranes. In 2010, another facility (L'Eliana) was commissioned in the Valencia area for 2.9 MGD production rate for nitrate removal using the EDR technology.

Sources*

Cháfer , V.S., Carbonell, J.S., and de Armas Torrent, J.C. Nitrate and Hardness Removal with Electrodialysis Reversal (EDR) in Gandia, (Valencia, Spain).

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

System Name: Weizmann Institute
System Location: Rechovot, Israel
System Type: NA

CASE #12

Treatment Type: Selective Electrodialysis (SED)
Questionnaire completed by: Shikun & Binui Environmental Group
Startup Date: 2008

Treatment Type	*Selective Electrodialysis*
System Capacity	*310 gpm*
Raw Water Nitrate	*84 – 89 mg/L as nitrate* *19 – 20 mg/L as N*

System Description

The Weizmann Institute of Science (Institute), located in Rechovot, Israel, is one of the top-ranking multidisciplinary research institutions in the world. In 2007, the nitrate MCL in the Israeli National drinking water regulations changed from 90 mg/L (20.3 mg/L as N) to 70 mg/L (15.8 mg/L as N). Because of this change, two of the Institute's wells were removed from the drinking water supply and the Institute had to rely on external water suppliers. Over time, municipal and national water costs increased. The Institute's management looked for solutions to solve nitrate problems and enable them to reopen the Institute's wells.

Prior to treatment, the wells were used for potable purposes and the Institute's irrigation needs. Selective Electrodialysis (SED) was identified as the Institute's most appropriate treatment technology. An SED system was implemented for the 310 gpm well. The Institute opted for treatment of one well and uses the second well as a dedicated irrigation supply source. The nitrate enriched concentrate from the SED process is fed into the non-potable irrigation system where the nitrate enhances plant growth.

SED has been successfully implemented throughout Israel since 2007. Developed by Nitron, Ltd., SED offers high water recovery (up to 95%), thereby minimizing waste volume. Pretreatment can generally be limited to filtration, energy efficiency is maximized due to low pressure operation (30 – 60 psi), chemical use is limited to concentrate treatment (no need for chemical addition to feed or product water), and low maintenance demands are possible due to automation, remote monitoring and control and infrequent cleaning. Membranes are cleaned in place (CIP) for 1 hour every 4 – 6 months and membrane life is typically 7 – 10 years.

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L N)
    - Average – 84 (19)
    - Minimum – 84 (19)
    - Maximum – 89 (20)

Treatment Technology Selection

SED and RO were considered for treating the Weizmann Institute well. A 10 year life cycle cost analysis that included capital and operations costs identified SED as the more cost-effective solution.

Treatment System Parameters

- Design Capacity
    - 310 gpm
- Pretreatment
    - Cartridge filtration
- Post-treatment
    - Chlorination
    - Acid addition (pH 4.5 – 5) to concentrate to prevent the precipitation of calcium carbonate and calcium sulfate in the concentrate cells
- Treatment system footprint
    - Treatment system: 8’ x 5’
    - Building footprint: Butler building - 50’ x 13’ as per client request
- Water recovery rate
    - 94.3%
- SED unit information:
    - Number of SED units: 1
    - Membrane pairs: 240
    - 59% Nitrate reduction
    - 30% TDS reduction
    - Energy consumption: 2.3 KWh/1,000 gal
- Monitoring:
    - Online nitrate analyzer
    - Laboratory nitrate samples
    - Online pH meters
    - Online turbidity
    - Online conductivity meters

Water Quality Results

The table below summarizes the water quality of the raw water, finished water, and concentrate stream from the Weizmann Institute.

Ion	Raw Water	Treated Water	Percent Removal	Concentrate Stream
Cl	194	105	45.9%	2051
SO_4	95	92	**3.0%**	140
HCO_3	232	165	28.7%	442
NO_3	92	44	**52.2%**	835
Na	109	76	30.4%	627
Ca	118	73	37.8%	808
Mg	24	15	36.6%	160
K	3.6	2.5	30.6%	20.5
Ba	0.149	0.093	37.8%	1.020
Sr	0.84	0.52	37.8%	5.75
TDS	869	574	**33.9%**	5090
pH	7.8	7.8		6.5

Residuals Management

The concentrate from the SED process is fed into the Weizmann Institutes non-potable system which is used for irrigation purposes. Since the SED process selectively removes nitrate, the Total Dissolved Solids (TDS) of the concentrate is less than an RO system would be if it were treating the same water, which allows the concentrate to be used for irrigation without the salinity adversely affecting plant growth. This management approach is also beneficial since the concentrated nitrate solution has limited the amount of fertilizer applied by the Weizmann Institute.

Technology Benefits and Drawbacks

Benefits

- Ease of regeneration
- Fewer chemicals than comparable technologies
- Membranes 7 – 10 year life span
- Energy consumption less than that of RO
- High water recovery
- Concentrate solution has relatively low TDS which may increase disposal options

Drawbacks

- Capital intensive technology
- Requires specific operator training

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)		
	$	Comments
Total:	650,000	
Housing:		Light building 50X13 feet. According to customer demands.
Piping:	50,000	Pipes, electric valves, storage tanks (the largest would be with a volume of about 180 cu. ft. for product water).
Storage Tanks (include description of uses):		See above.
Operating and Monitoring Equipment:	90,000	Control system, remote assistance for the control system, on line nitrate measurement, conductivity, pH, turbidity.
Membranes Modules:	300,000	Complete SED membrane stack.
Permits:		According to local regulations.
Other:	210,000	Design; Electricity boards – design, manufacturing & installation; Pumps and blowers.

O & M Costs (Total with explanation or component costs)		
	Unit	Comments
Membrane:	%/year	10% per year is the replacement rate.
Concentrate Disposal or Treatment:	Gallons/year	Site specific; 4 – 5% of the SED system capacity.
Chemicals:	Total, [lb/1000 gallon]	0.77
Specific acid consumption	[lb/1000 gallon]	0.72
Specific chlorine consumption	[lb/1000 gallon]	0.04
Specific caustic soda consumption	[lb/1000 gallon]	0.01
Repair/Maintenance (not including Labor):	$/year	7,000
Specific power consumption	[kWh/1000 gallon]	1.4
Labor ($):	$	16,000 (40$ per hour basis)
Labor (Hours per Year):	hours	400

Operational Notes

The SED system is highly instrumented and the PLC has over 200 monitored inputs. As a result there have not been any failures that have resulted in MCL violation. While there have been failures resulting in alarms and shutdowns, the control system has been robust enough to shut down the system and prevent water with high nitrate entering the distribution system. Typical shutdowns can be rectified in a matter of hours and normal operation is resumed.

Sources*

Merhav, Neta. (2010) Completed questionnaire. October, 2010.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

3.4 Biological Denitrification (BD)

Commonly used in wastewater treatment, biological reactors are emerging as a method for denitrification of potable water with the potential to address multiple contaminants including nitrate, chromate, perchlorate, and trace organic chemicals (Brown 2008). Biological denitrification (BD) in potable water treatment has been implemented in Europe since 1804 (Lenntech 2009), with recent full-scale systems in France, Germany, Austria, Poland, Italy, and Great Britain (Meyer 2009; Dördelmann 2009). To date, full-scale drinking water applications in the United Sates have been limited to a single plant in Coyle, OK (no longer online). However, two full-scale biological denitrification systems are anticipated in California within the next couple of years.

Denitrification occurs naturally in the environment as part of nitrogen cycling. Application of biological denitrification to potable water treatment (Figure 12) utilizes denitrifying bacteria to reduce nitrate to innocuous nitrogen gas in the absence of oxygen (anoxic conditions).

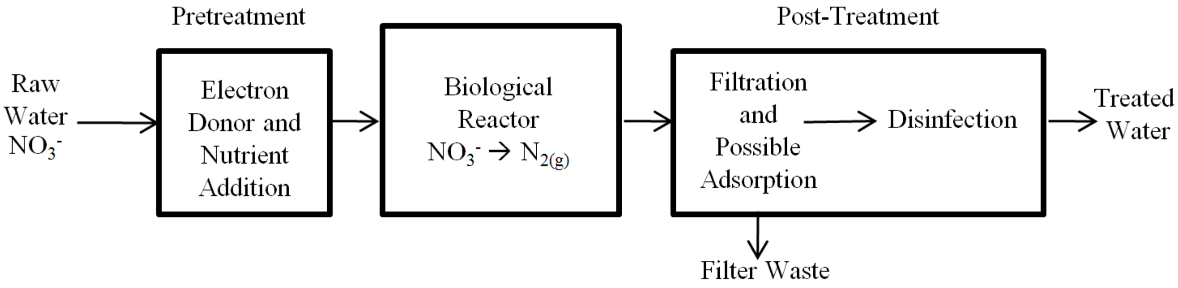

Figure 12. Biological denitrification schematic.

The reduction of nitrate proceeds stepwise in accordance with Eqn. 9. In contrast with the separation processes of IX, RO, and ED/EDR, nitrate is reduced and thereby removed from the system rather than simply being displaced to a concentrated waste stream.

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \qquad \text{(Eqn. 9)}$$

Denitrifying bacteria require an electron donor (substrate) for the reduction of nitrate to nitrogen gas. In conventional wastewater treatment, substrate addition is not typically needed, because the wastewater contains sufficient carbon for denitrification to occur. However, depending on the source, substrate addition is often required for the biological denitrification of potable water. The addition of a carbon substrate in potable water treatment is somewhat counter intuitive. In fact, one principal objective of potable water treatment is to minimize dissolved carbon in the water to minimize growth of microbes (e.g., biofilms) and production of disinfectant byproducts (e.g., THMs). Feed water composition may need to be further augmented with the addition of nutrients required for cell growth (phosphorus for example). Autotrophic bacteria utilize sulfur or hydrogen as an electron donor and inorganic carbon (typically carbon dioxide) as a carbon source for cell growth (Eqns. 10 and 11), while heterotrophic bacteria consume an organic carbon substrate, like methanol, ethanol, or acetate (Eqn. 12) (Mateju et al. 1992; Kapoor & Viraraghavan 1997).

Eqns. 10 through 12 illustrate the overall denitrification reaction defining the stoichiometric relationship between electron donor, carbon source, and nitrate in the production of cells and the conversion of nitrate to nitrogen gas. Not all nitrogen is converted to nitrogen gas. Some nitrogen is required for cell growth. The governing stoichiometric equation indicates the necessary dose and varies with the substrate used. For example, the stoichiometric factor for acetic acid is 0.82 moles of acetic acid per mole of nitrate (Dördelmann et al. 2006).

$$11\ S^0 + 0.5\ CO_2 + 10\ NO_3^- + 2.54\ H_2O + 1.71\ NH_4^+ \rightarrow 0.92\ C_5H_7O_2N + 11\ SO_4^{2-} + 5.4\ N_2 + 9.62\ H^+ \qquad \text{(Eqn. 10)}$$

$$H_2 + 0.35\ NO_3^- + 0.35\ H^+ + 0.052\ CO_2 \rightarrow 0.010\ C_5H_7O_2N + 0.17\ N_2 + 1.1\ H_2O \qquad \text{(Eqn. 11)}$$

$$1.08\ CH_3OH + NO_3^- + H^+ \rightarrow 0.065\ C_5H_7O_2N + 0.467\ N_2 + 0.76\ CO_2 + 2.44\ H_2O \qquad \text{(Eqn. 12)}$$

Various species of bacteria are responsible for denitrification including *Thiobacillis denitrificans, Micrococcus denitrificans, Pseudomonas maltophilia* and *Pseudomonas putrefaciens* (Kapoor & Viraraghavan 1997). Due to slower bacterial growth rates, autotrophic denitrification offers the advantage of minimizing biomass accumulation; however, autotrophic denitrification requires alkalinity to supply the inorganic carbon source (carbon dioxide) for cell growth (Della Rocca et al. 2006). BIODEN® and DENICARB® are heterotrophic biological denitrification processes, while DENITROPUR® is an autotrophic option. Selected full-scale biological denitrification systems are listed in Table 10 (Dördelmann 2009).

Table 10. Full-scale biological denitrification systems for potable water treatment.[1]

Location	**Reactor Configuration**	**Substrate, Denitrification type**	**Flow rate m^3/h (MGD)**
Germany			
Neuss	Fixed bed, down-flow	Acetic acid, Heterotrophic	150 (0.95)
Frankfurt Airport	Fluidized bed, up-flow, DENICARB®	Ethanol, Heterotrophic	320 (2.03)
Aschaffenburg	Fixed bed, up-flow, DENITROPUR®	Hydrogen and CO_2, Autotrophic	1600 (10.14)
Föhr Island			90 (0.57)
Austria			
Obersiebenbrunn	Fixed bed, down-flow, BIODEN®	Ethanol, Heterotrophic	180 (1.14)
Poland			
Czestochowa	Fixed bed, down-flow, BIODEN®	Ethanol, Heterotrophic	500 (3.17)

[1] Dördelmann 2009.

In their review of potable water treatment methods for the removal of nitrate, Mateju et al. (1992), Kapoor & Viraraghavan (1997), Soares (2000), and Shrimali & Singh (2001) discuss previous research and applications of biological denitrification. Problems associated with the application of conventional biological denitrification to potable water treatment include additional post-treatment for removal of biomass and dissolved organics, the potential for incomplete denitrification, increased capital costs, and sensitivity to environmental conditions (Kapoor & Viraraghavan 1997). To address these concerns, several treatment configurations using biological denitrification have been developed.

3.4.1 Biological Denitrification - Design Considerations

Table 11 summarizes key design considerations in the application of biological denitrification.

Table 11. Summary of design considerations for biological denitrification for nitrate removal in potable water.

Pretreatment	• Substrate and nutrient dosing • pH adjustment
Post-Treatment	• Carbon adsorption for organic carbon removal ○ Carbon adsorption is not always required ○ Residual substrate removal can be accomplished via biological filtration • Aeration • Filtration • Disinfection
Chemical Usage	• Possible pH adjustment • Substrate and nutrient addition • Coagulant/polymer use to meet turbidity standards • Disinfection
O&M	• Historically operator intensive ○ Operator demands can be minimized with latest design configurations • Constant monitoring required to assure efficient removal, microbe health, etc. • Monitoring of nitrite and ammonia will also be necessary due to the potential for incomplete denitrification • Management of chemicals • Waste sludge storage and disposal • New plants can be highly automated • Historically viewed as operationally complex ○ More unit processes than IX ○ New design configurations can minimize complexity (e.g., fixed bed)
System Components	Important process considerations include (Dördelmann 2009): • Dosage requirements of substrate and nutrients • Reactor configuration and governing equation of the biological process • Aeration to remove nitrogen gas and provide oxygen • Filtration to remove particulate matter • Activated carbon may be used to remove substrate residual and avoid DBP formation (for heterotrophic systems) • Disinfection
Waste Management and Disposal	• Sludge disposal – Biological solids and residual organic matter • No waste brine or concentrate as in separation processes
Limitations	• Requires anoxic conditions • Chemical management • Few examples for nitrate removal in the U.S. • Post-treatment requirements • Operator training • Intermittent use of wells may be challenging due to the need for acclimation of microorganisms

Water Quality

Anoxic conditions are required for denitrification to occur. In the presence of oxygen (> 0.1 mg/L) bacteria preferentially reduce oxygen rather than nitrate, diminishing the efficiency of the process. For all configurations, the optimal growth of microbes must be considered. Control and monitoring of water quality characteristics including temperature, pH, salinity, and oxidation reduction potential (ORP) can be fundamental to the stability and efficiency of the biological denitrification system (WA DOH 2005). For biological denitrification, near neutral pH is preferred (7 – 8) and temperatures below 5^{o}C/41^{o}F can inhibit denitrification (WA DOH 2005).

Pretreatment will include addition of substrate and nutrients in the appropriate dose while post-treatment requirements can include coagulant addition, filtration, gas exchange, and disinfection for the removal of biomass, particulates, and substrate residuals (Panglisch et al. 2005; Dördelmann et al. 2006).

System Components and Site Considerations

Important process considerations in the design and operation of BD systems include (Dördelmann 2009):

- Dosage requirements of substrate and nutrients
- Reactor configuration and governing equation of the biological process
- Aeration to remove nitrogen gas and provide oxygen
- Filtration to remove particulate matter
- Activated carbon to remove substrate residual and avoid DBP formation (for heterotrophic)
- Disinfection

Conditions should be optimized to ensure complete denitrification. In addition to nitrate reduction to meet the nitrate MCL (45 mg/L as NO_3^-, 10 mg/L as N), effluent *nitrite* levels must not exceed the *nitrite* MCL of 1 mg/L as N. Incomplete denitrification, which can be associated with higher dissolved oxygen (DO) levels, can result in the production of greenhouse gases (GHG) such as NO and N_2O. Dissolved oxygen levels can be decreased using reducing agents or through the provision of sufficient electron donor to enable depletion of DO (Meyer et al. 2010).

System configurations of biological denitrification include: fluidized bed reactor, fixed bed reactor, membrane biofilm reactor, and bio-electrochemical reactors. In situ options (including bank filtration) have also been explored in the research.

Fixed Bed

Fixed bed biological reactors can be in up-flow or down-flow systems in a pressurized or open flow configuration (Brown 2008). Typical options for growth support media include sand, plastic, and granular activated carbon (Brown 2008). Accumulation of biomass in the media leads to head loss requiring periodic backwashing. Post-treatment requirements can include filtration, gas exchange, and disinfection for the removal of biomass, particulates, and substrate residuals (Panglisch et al. 2005; Dördelmann et al. 2006). The fixed bed configuration "is often coupled with pre-ozonation to improve the removal of organic material, which reduces regrowth potential and DBP formation in distribution systems" (Brown 2008, p. 137). (See Soares (2002), Panglisch et al. (2005), Aslan (2005), Dördelmann et al. (2006), Carollo Engineers (2008), Upadhyaya (2010), Meyer et al. (2010), and City of Thornton (2010), in Table A.4 for research examples of the fixed bed configuration.)

A detailed case study of the Fixed Bed configuration of biological denitrification in Riverside, CA, is included in Section 3.4.5 Biological Denitrification - Case Studies.

Fluidized Bed

The fluidized bed reactor operates in an up-flow mode, resulting in granular growth support media expansion. Fluidizing the granular media offers several advantages over the fixed bed configuration. Flow resistance is minimized and the system does not need to be taken off line for backwashing because accumulated biomass is removed by the fast flowing feed water and/or "in-line mechanical shearing devices" (Brown 2008, p. 139). "The biofilm is detached from the support material only upon strong mechanical effect, thus the excess biomass can be intermittently removed from the reactor, independently of the purified water" (Holló & Czakó 1987, p. 418).

Maintenance of the sufficient up-flow velocity can be achieved through recycled flow and reactor volumes are designed for a typical bed expansion of 25 to 30% (Brown 2008). (See Kurt et al. (1987), Holló & Czakó (1987), and Webster & Togna (2009), in Table A.4 for research examples of the fluidized bed configuration.)

A detailed case study of the Fluidized Bed configuration of biological denitrification in Rialto, CA is included in Section 3.4.5 Biological Denitrification - Case Studies.

Membrane Biological Reactor (MBR)/Membrane Biofilm Reactor (MBfR)

With the addition of membrane technology to conventional biological denitrification, common concerns of biological treatment can be minimized through physical separation of biomass and substrate from the treated water. In a comprehensive review of membrane bioreactors, McAdam & Judd (2006) present the pros and cons of a variety of MBR configurations (Table 12). MBRs can be designed for autotrophic or heterotrophic denitrification. Pressurized systems have been explored using submerged ultrafiltration membranes or external (sidestream) MBRs, while pressure neutral diffusion systems have been implemented with ion exchange membranes and microporous membranes (McAdam & Judd 2006; Brown 2008). Membrane types include hollow fiber, ion exchange, microporous, and flat sheet. (See Mansell & Schroeder (2002), Ergas & Rheinheimer (2004), Nerenberg & Rittman (2004), Chung et al. (2007), Meyer et al. (2010), and City of Thornton (2010) in Table A.4 for research examples of MBR configurations.)

Applied Process Technology, Inc. (APT) has developed an autotrophic membrane biofilm reactor (MBfR) (Gormly & Borg N.D.) using hydrogen gas as the electron donor rather than a carbon substrate. Early pilot and bench-scale studies of the APT MBfR raised several key challenges requiring optimization of the system, including the need to avoid build-up of excess biomass (Rittmann 2007). As part of a long term pilot-scale study[19] in Glendale, AZ, APT's MBfR was examined to address high nitrate levels in groundwater. This autotrophic biological denitrification system successfully reduced nitrate levels to below the MCL. Three types of hollow fiber membranes were examined for substrate delivery. Operational concerns highlighted by this study include (Meyer et al. 2010):

- Problems with leaking fibers,
- hydrogen sulfide formation due to excessive hydrogen gas pressure,
- ammonium generation from biomass decay due to operational interruption and insufficient electron donor, and
- nitrite levels above the 1 mg/L nitrite as N limit (incomplete denitrification).

To address these concerns, the authors suggest:

- The use of the latest optimized membranes,
- consistent and adequate nutrient and electron donor supply,
- oxidation of nitrite in post-treatment if necessary,
- stable loading and continuous operation to avoid system upset, and
- parallel reactors to allow for maintenance and repair.

The MBfR pilot was one of three biological configurations examined in Glendale. Two up-flow heterotrophic fixed bed bioreactors were also examined, each with a different media type. Post-treatment included filtration using biologically active carbon and ozonation. The two most promising biological treatment options, the MBfR and the up-flow fixed bed bioreactor with plastic media, were compared with each other and also with an IX system. Overall, using a multi-criteria analysis with consideration of sustainability, the MBfR scored the most favorably regarding benefits, but the least favorably regarding life cycle costs. The life cycle costs of both the IX and the fixed bed bioreactor options were lower than that of the MBfR. The Glendale study highlights several key areas of future research including "a shut-down test where biological treatment processes sit dry for a period of time and then re-start at optimal hydraulic loading rates [and] a re-acclimation test where systems are re-initiated after losing all viable biomass" (Meyer et al. 2010, p. 164).

As a promising technology, with further research and design optimization to reduce costs, the MBfR may become a more feasible treatment option. As of 2008, a demonstration project of the MBfR for nitrate and dibromochloropropane (DBCP) removal was under consideration for the City of Fresno, CA, to test performance of the MBfR as an alternative treatment option (City of Fresno 2008).

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[19] Project partners and participants: City of Glendale, Arizona, Water Research Foundation, Arizona State University, CH2M HILL, Applied Process Technology, Inc., Intuitech, Inc., KIWA Water Research, and Layne-Christensen.

Table 12. Membrane biological reactor configurations.

Diffusive Extraction Microporous Membranes	Relying on diffusion for nitrate transfer through the membrane, extractive MBRs (i.e., Fixed MBRs) do not directly filter the treatment stream, but they do provide separation of the denitrification chamber (substrate, biomass, and associated residuals) and the treatment stream. Key issues are the transfer of biomass or substrate across membrane, the potential for fouling/scaling, and the need for a high transfer rate of nitrate to the denitrification compartment.
Diffusive Extraction Ion Exchange Membranes	With the use of an ion exchange membrane rather than a microporous membrane, selectivity for nitrate and decreased mass transfer from the denitrification compartment are facilitated; however, capital and maintenance costs of ion exchange membranes can be significant and the need to manage membrane fouling persists (McAdam & Judd 2006).
Gaseous Substrate Delivery Hollow Fiber Membranes	In autotrophic systems, membranes can be used for gaseous substrate delivery, (i.e., hydrogen gas) (McAdam & Judd 2006). Previous problems with limited hydrogen gas transfer have been addressed with the use of hollow fiber membranes for delivery. Substrate passes through the membrane to the biofilm on the outer membrane surface. The membrane does not separate the biomass from the treatment stream and does not directly filter the treatment stream (McAdam & Judd 2006); the presence of "sloughed biomass" and biological residuals in the treatment stream requires further treatment downstream.
Pressure Driven Direct Filtration MBRs	Pressurized MBRs provide the advantage of direct filtration. Pressure is used to draw the denitrified water through the submerged membrane, leaving behind biomass and other undesirable constituents. However, the use of this configuration for denitrification is complicated by the fact that aeration is typically used for mixing and to minimize fouling of the external membrane surface (Brown 2008).

In Situ Denitrification

Bank filtration refers to the withdrawal of surface water through an embankment. The porous media (soil) of the bank serves as a biological reactor providing treatment through "filtration, dilution, sorption, and biodegradation processes" (Brown 2008, p. 140). Bank filtration was employed in water treatment as early as 1870 along the Rhine River in Germany (Brown 2008). As a recent example, a full-scale study in Aurora, CO, demonstrated effective nitrate removal with bank filtration of surface water from the South Platte River (Waskom, Carlson & Brauer N.D.). Bank filtration has also been implemented to address nitrate impacted waters in Saxony, Germany, and Des Moines, IA (Jones et al. 2007; Grischek et al. 2010).

Biological denitrification for in situ removal of nitrate from groundwater was explored by Hunter (2001), Haugen et al. (2002), Schnobrich et al. (2007), and many others. Through in situ denitrification, the subsurface acts as the porous media through which water is filtered. Residual organics and biomass from denitrifiers can thus be removed naturally. (See Hunter (2001), Haugen et al. (2002), and Schnobrich et al. (2007) in Table A.4 for research examples of in situ application.) See Technical Report 5, Section 2 for a discussion of in situ denitrification, permeable reactive barriers, phytoremediation, and other remediation methods (King et al. 2012).

Post-Treatment Requirements - Filtration/Taste & Odor/Disinfection

With the use of microorganisms and the addition of a carbon substrate, post-treatment is essential to meet turbidity standards, to remove biomass and residual organic matter, and to address taste and odor concerns. Post-treatment must include disinfection to address biological contamination and can also include dual media filtration and/or activated carbon filtration and aeration (individual state regulations will need to address local requirements).

Residuals Management and Disposal

In contrast to the concentrated waste stream from removal processes, biological denitrification has limited waste demands due to the conversion of nitrate to nitrogen gas. Waste sludge, consisting of biological solids and residual organic matter, requires appropriate disposal; however, with nearly 100% water recovery, the low waste volumes are not a significant burden (Kapoor & Viraraghavan 1997).

Maintenance, Monitoring, and Operational Complexity

Because biological denitrification is microbially mediated, to maximize performance, systems should be run continuously, with a consistent supply of substrate and nutrients at the appropriate dosage (Dördelmann 2009). An initial startup period may be necessary for development and acclimation of the microorganisms (Holló & Czakó 1987; Aslan 2005). This may be problematic for intermittent use of wells and wasting may be required for acclimation to occur. Backwashing, consistent maintenance, and regular monitoring of product water quality are also essential. Constituents that should be monitored frequently are nitrate, nitrite, pH, oxygen, turbidity, conductivity, dissolved organic carbon, and bacterial count (Dördelmann 2009). Operation and maintenance demands of biological denitrification systems typically exceed those of alternative treatment technologies. However, these systems are more sustainable because nitrate is reduced to innocuous nitrogen gas rather than concentrated in a waste stream that requires costly disposal (Dördelmann 2009).

3.4.2 Biological Denitrification - Cost Considerations

For efficient operation of a biological denitrification system, maintaining optimal conditions for the bacteria is essential, as is balancing the appropriate substrate and nutrient dose and managing pre and post-treatment while meeting necessary potable water guidelines. Factors affecting system cost include facility size (flow rate), source water quality (including nitrate concentration), environmental factors (temperature and pH), target effluent nitrate concentration, and possible wasting due to intermittent use of wells and associated acclimation of microorganisms.

Capital costs for biological denitrification include land, housing, piping, storage tanks, O&M equipment, preliminary testing (through extensive pilot studies), permits, and significant operator training. O&M costs include post-treatment, sludge disposal, chemical use (pH adjustment, substrate and nutrient dosing), repair, extensive monitoring and maintenance, power, and labor. Costs can be higher in certain states, depending on post-treatment requirements.

Very little published cost information from existing biological denitrification systems for drinking water is available in the literature, due to the limited number of full-scale systems (Table 13). Costs have been adjusted to 2010 dollars, unless indicated otherwise. The listed cost information is provided as an approximate range of costs for specific facilities. Costs for implementing biological denitrification may be very different from those listed here. A thorough cost analysis of design parameters for specific locations would be required for accurate cost estimation. The information gathered through the questionnaire includes detailed costs associated with the individual case studies included in this analysis. A detailed discussion of treatment costs is included in Section 6 Treatment Cost Analysis.

Table 13. Cost information* for biological denitrification of potable water.

System Flow**	< 0.5 MGD	0.5 – 5 MGD	5+ MGD
Annualized Capital Cost ($/1000 gal)	0.83 [1]	0.61 – 0.80 [2, 3]***	0.51 – 0.62 [4]
O&M Cost ($/1000 gal)	0.30 [1]	0.33 – 0.46 [2, 3]***	0.74 – 0.94 [4]
Total Annualized Cost ($/1000 gal)	1.13 [1]	1.03 – 1.13 [2, 3]***	1.25 – 1.56 [4]

*Costs have been adjusted to 2010 dollars with 7% interest over 20 years, unless indicated otherwise.
**When available, costs are based on actual system flow rather than design capacity.
***Listed costs are based on biological treatment for perchlorate and should be considered only as a rough estimate of similar systems for nitrate treatment.
[1] Silverstein (2010), not adjusted to 2010 dollars. [2] Webster & Togna (2009). [3] Carollo Engineers (2008).
[4] Meyer et al. (2010).

3.4.3 Biological Denitrification - Selected Research

Table A.4 of the Appendix lists recent research studies relevant to the use of biological denitrification in potable water treatment.

3.4.4 Biological Denitrification - Summary of Advantages and Disadvantages

A summary of advantages and disadvantages of biological denitrification in comparison with other treatment options is listed in Table A.6 of the Appendix. Advantages of the use of biological denitrification for nitrate removal from potable water include high water recovery, no brine or concentrate waste stream (reduction of nitrate rather than removal to a concentrated waste stream), low sludge waste, less expensive operation, limited chemical input, multiple contaminant removal, and increased sustainability (WA DOH 2005; Brown 2008; Upadhyaya 2010).

Disadvantages of nitrate removal using biological denitrification are post-treatment requirements for the removal of biomass and dissolved organics, high capital costs, potential sensitivity to environmental conditions (although recent pilot tests indicate robust newer designs), system footprint larger than typical IX systems, high system complexity (may be simplified with new configurations), lack of full-scale systems in the U.S., potential for incomplete denitrification and GHG production, pilot study requirements, and slow start-up (Kapoor & Viraraghavan 1997; WA DOH 2005).

3.4.5 Biological Denitrification - Case Studies

The following case studies provide detailed information on the design and operation of a pilot-scale and planned full-scale biological treatment systems that can be used for nitrate removal.

System Name: West Valley Water District
System Location: Rialto, CA
PWSID: CA 3610004

CASE #13 System Type: Demonstration System, Full-scale Installation is under construction

Treatment Type: Biological Denitrification, Fluidized Bed Bioreactor (FBR)
Questionnaire completed by: Todd Webster from Envirogen Technologies, Inc., Director of Sales & Bioreactor Applications.
Tom Crowley from the West Valley Water District.
Demonstration Dates of Operation: 2007 – 2008, Full-scale Installation Proposed Startup Date: 2012

System Description

Treatment Type	*Fluidized Bed Bioreactor*
System Capacity	*2,000 – 4,000 gpm* *(for proposed full-scale treatment)*
Raw Water Nitrate	
(abandoned well)	*17 – 19 mg/L as nitrate* *~4 – 5 mg/L as N*
(pilot)	*27.5 – 27.9 mg/L as nitrate* *6.2 – 6.3 mg/L as N*
Raw Water Perchlorate	
(pilot)	*50+ ug/L*

The West Valley Water District (District) utilizes 2 surface water sources and 5 groundwater wells. Of these supplies, one groundwater well is impacted by nitrate contamination above the MCL, with an average nitrate concentration of 18 mg/L of nitrate as nitrate (~4 mg/L of nitrate as N). The well source is the Chino Basin which has an estimated capacity of 300 – 400K ac. ft. For the immediate future this well has been abandoned due to nitrate contamination, while feasible treatment options are being explored. The primary water quality concern in the District is perchlorate contamination. Biological denitrification has been explored principally to address perchlorate levels typically in the range of 50 – 53 µg/L. However, simultaneous nitrate reduction in the biological denitrification process makes this system an appropriate example as a nitrate treatment alternative.

Unlike the removal processes of IX, RO, and EDR, biological denitrification allows for the destruction of nitrate through reduction to innocuous nitrogen gas. The fluidized bed configuration maximizes media surface area for the growth of denitrifying bacteria. After a successful 1-year demonstration study on well Rialto #2, a full-scale Fluidized Bed Bioreactor (FBR) is expected to be online in early 2012.

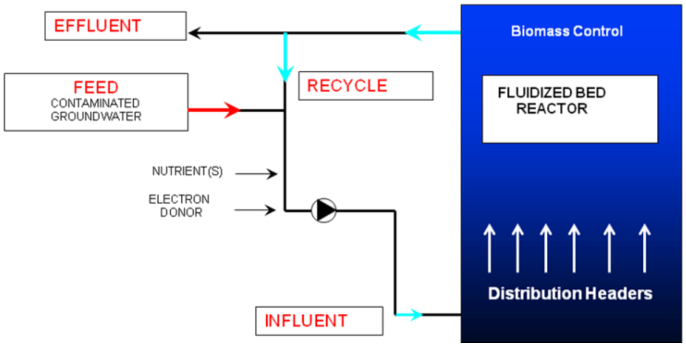

Figure 13. FBR configuration. (Source: reprinted with permission, Webster & Togna 2009.)

“The system is the fluidized bed bioreactor (FBR). The contaminated feed water is pumped from the wellhead and fed directly into a recycle line of the reactor. The feed and recycle water enters the vessel through an inlet header at the bottom of the reactor and is distributed through lateral piping and nozzles (Figure 13). The fluid passes upward through the media, causing the media to hydraulically expand approximately 28% of the settled bed height. Through a self-inoculating process from the contaminated feed water, microorganisms attach on to the fluidized media. Adequate quantities of electron donor (i.e., acetic acid) and nutrients are added to the reactor. Utilizing this electron donor and the nutrients, the attached microorganisms perform an oxidation/reduction reaction in consuming all of the dissolved oxygen, nitrate, and perchlorate. As the microorganisms grow, the amount of attached microbes per media particle also increases. Since the microbes primarily consist of water, the volume of the microbe/media particle increases, but the specific density decreases. This allows the media bed to expand and fluidize further such that longer hydraulic retention times can be achieved for contaminant removal. The treated fluid flows into a submerged recycle collection header pipe and the effluent collection header pipe at the top of the reactor. A portion of the fluid exits the FBR system to a post-aerator while the balance is recycled back to the suction of the influent pump. An in-bed biomass separation device controls bed height growth by physically separating biomass from the media particles. Typically, a bed expansion of 40 – 60% of the settled bed height is targeted. Any excess biomass that is separated from the media exits the system through the effluent collection system” (Webster & Togna 2009, p. 6).

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L as N)
  - Demonstration well: 27.5 – 27.9 (6.2 – 6.3)
  - Abandoned nitrate impacted well
    - Average: 18 (~4)
- Co-contaminants: Perchlorate
  - Demonstration well: 50 – 53 µg/L, spiked to 1000 µg/L (with appropriate substrate and nutrient adjustments)
  - Abandoned nitrate impacted well: a few ppb

Treatment Technology Selection

The fluidized bed bioreactor was selected for perchlorate removal; however, nitrate reduction is also accomplished. Due to the levels of nitrate and perchlorate, biological denitrification was deemed to be more cost-effective than alternative physico-chemical technologies. Generally, with nitrate levels well above the MCL (e.g., 100 mg/L as nitrate), removal processes can become more costly. Biological denitrification can be the more feasible option for source water containing co-contaminants and/or high levels of nitrate.

For the Rialto #6 and West Valley Water District #11 wells, the fluidized bed reactor and IX were considered. Two additional pilot studies were performed to assess nitrate and perchlorate treatment using a packed bed bioreactor and zero valent iron.

Demonstration System Parameters

- Design Capacity
  - 50 gpm capacity
- Pretreatment
  - Substrate and nutrient addition
    - Acetic Acid:
      16.2 mg/L as C (+20 – 25%)
      15 mL/min, 50% acetic acid
  - Nutrient addition
    - Phosphoric Acid:
      0.3 mg/L as P
      10.5 mL/min
- Post-treatment
  - Aeration tank
    - Increase dissolved oxygen
  - Clarifier/multimedia filter
  - GAC filtration
  - UV disinfection versus chlorination
- Water Recovery: > 99%
- Bed Expansion
  - 28% of settled bed height
  - Max: 40 – 60%
- Media: Sand or GAC
- Cleaning requirements
  - Biomass separator
  - In-bed cleaning eductor
- Manufacturer:
  Envirogen Technologies, Inc.
- Monitoring: Throughout the demonstration using online nitrate and perchlorate analyzers

Demonstration System Cost Estimation - Scaling up from 50 gpm to 1000 gpm

Capital Costs (2008 dollars) (1000 gpm)	
Total Equipment Costs:	$1,966,000
Total Contractor Costs:	$570,140
Total Home Office Costs:	$664,000
Total Installed Capital Costs (1000 gpm):	$3,200,140
Amortized Capital Cost ($/AF):	$128 (30 years, 4.9% bonding rate)
O & M Costs (2008 dollars) (1000 gpm)	
Electricity ($/yr):	$87,600
Chemicals ($/yr):	$133,187
Maintenance ($/yr):	$20,000
Total Operating Costs ($/yr):	$240,787
Operating Costs ($/AF):	$149
Total Cost (2008 dollars) (1000 gpm)	
Total Annualized Cost ($/AF):	$277

Proposed Full-scale System Parameters - See Figure 14

- Design Capacity
    - 2000 gpm capacity
    - Expandable to 4000 gpm
- Pretreatment
    - Substrate and nutrient addition
        - Acetic Acid: 10 – 15 mg/L as C
- Post-treatment
    - Aeration tank
        - Increase dissolved oxygen
    - Clarifier/multimedia filter
    - Chlorine disinfection
- Treatment system footprint
    - Treatment system: 2 FBRs
        - 14’ diameter x 24’ height
    - Residuals handling system
        - DAF for solids removal
    - Total system footprint
        - For 4000 gpm
            - 180’ x 130’
        - For 2000 gpm
            - 25% less than above
- Water Recovery: > 99% expected
- Bed Expansion
    - 28% of settled bed height
    - Max: 40 – 60%
- Media: Sand or GAC
- Cleaning requirements
    - Biomass separator
    - In-bed cleaning eductor
- Manufacturer:
  Envirogen Technologies, Inc.
- Monitoring: N/A
- Waste Volume:
  0.3 gpm waste per 2000 gpm treated

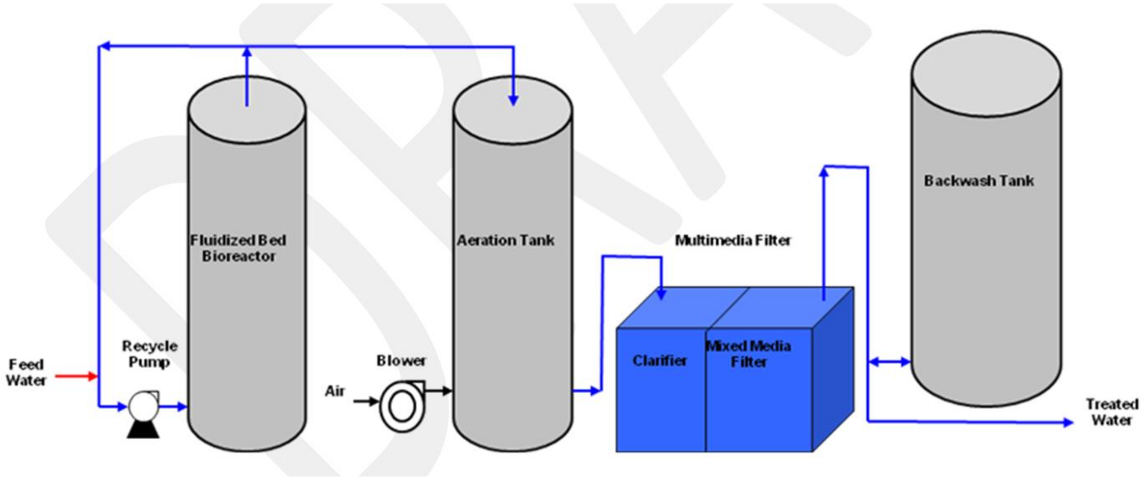

Figure 14. FBR treatment system schematic. (Source: Webster et al. 2009, reprinted from *Journal AWWA* (May 2009) by permission. Copyright © 2009 by the American Water Works Association.)

Proposed Full-scale System Cost Estimation

Capital Costs (Total with explanation or component costs)	
Treatment and Monitoring Equipment:	$1.8 million for the 2 FBR vessels, the 2 post-aeration vessels, nitrate and perchlorate monitoring, 1 DAF, chemical feed systems, pumps, valves, additional components
Filtration System:	$800,000 for the 2 clarifier and multimedia filters
O & M Costs (Total with explanation or component costs)	
Unavailable because system is not yet operating. Please see above for estimated O & M costs from the full-scale demonstration study.	

Residuals Management

Unlike removal technologies, the use of the FBR results in destruction of nitrate and perchlorate, rather than the transfer of these constituents to a concentrated waste stream requiring disposal.

Technology Benefits and Drawbacks

Benefits

- Capable of perchlorate removal and handling high nitrate levels
- Reduction of nitrate rather than just removal
- High water recovery and limited waste
- Lower operating costs than removal technologies

Drawbacks

- Limited application in the U.S.
- Large system footprint
- More complicated permitting
- Extensive pilot study necessary
- Start-up time (up to a month)
- Increased operator attention
- Sensitivity to system interruption

Additional Information

The full-scale system is being planned specifically to address perchlorate contamination, but will reduce nitrate as well. Construction is currently underway and the system should be fully operational with discharge to groundwater by Mid-2012. With approval from the California Department of Public Health (CDPH) it is expected that distribution of treated drinking water will begin in Early-2013. Regarding operator training, for this system, the operators already held the required certification due to operation of a surface water treatment plant; however, for other utilities lacking this pre-existing experience, additional training would likely be needed. Regarding permitting, the 97-005 process from the CDPH Policy Memo 97-005, (“Guidance for the Direct Domestic Use of Extremely Impaired Sources”) was followed as guidance for the FBR installation. The 97-005 process is strenuous and involved, requiring analysis of failure and worst-case scenarios and a 6 month demonstration of proper operation of the treatment plant. The system sizes for which an FBR would be appropriate vary with the concentration of contaminants. Feasible application of an FBR can be more dependent on load than on flow capacity. The FBR has been implemented or tested for systems as small as 7 to 12 gpm and across perchlorate concentrations from 12 to 13000 ppb. Typical flow rates are 250 to 5000 gpm, but with high loads FBRs can become more feasible for lower flow rates.

<u>Source</u>*

Webster, T.S., Guarini. W.J. and Wong, H.S. (2009) Fluidized bed bioreactor treatment of perchlorate-laden groundwater to potable standards. *Journal of the American Water Works Association*, **101** (5).
Webster, T.S. and Togna, P. (2009) Final Report: Demonstration of a Full-Scale Fluidized Bed Bioreactor for the Treatment of Perchlorate at Low Concentrations in Groundwater. ESTCP Project ER-0543.
Webster, T.S. and Crowley, T. (2010) Completed questionnaire and personal communication.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

System Name: Western Municipal Water District (WMWD) - Arlington Desalter
System Location: Riverside, CA
PWSID: CA3310049

CASE #14 System Type: Pilot Study, Full-scale System is currently in the Design Phase

Treatment Type: Fixed Bed Bioreactor (FXB)
Questionnaire completed by: Joseph Bernosky, WMWD, Director of Engineering
Jess Brown, Carollo Engineers, Carollo Research Group Manager
Startup Date: Proposed 2013

Treatment Type	*Fixed Bed Bioreactor*
System Capacity	*2.4 MGD (1670 gpm)*
Raw Water Nitrate	*44 – 89 mg/L as nitrate* *10 – 20 mg/L as N*

System Description

The Western Municipal Water District (Western) operates the Arlington Desalter as a drinking water supply facility. The original RO water treatment facility was constructed in the late-1980s for salt management in the Arlington groundwater basin. In 2002 the facility was upgraded and subsequently approved as a potable water supply by the California Department of Public Health (CDPH). Three wells supply raw water to the RO process and two additional wells supply bypass water for blending with the RO permeate to produce the finished water (product water).

The current capacity of the Arlington Desalter is 5 million gallons per day (MGD) of RO permeate and approximately 1.3 MGD of untreated bypass for a total product water capacity of 6.3 MGD. Western's water supply reliability program, developed through the Integrated Regional Water Management Plan (IRWMP), includes plans for expansion of the Arlington Desalter to 10 MGD capacity.

A pilot study was conducted at the Arlington Desalter from April through November 2007 that demonstrated the feasibility of nitrate removal using fixed bed biological treatment (biodenitrification) of the RO bypass stream. After completion of the initial pilot study, CDPH proposed a turbidity standard for the biodenitrification process. Supplemental pilot testing conducted between May and September 2008 demonstrated compliance with the new turbidity standard by adding a nitrogen degasification step followed by coagulation and polishing filters.

The FXB biological process utilizes a stationary bed of granular activated carbon (GAC) on which biofilms containing nitrate-reducing bacteria develop. Raw water is drawn from a well amended with an electron donor such as acetic acid. The water is then pumped through the GAC bed. Bacteria in the bed convert the nitrate to nitrogen gas and water. A one-time acclimation period is required to develop the nitrate-reducing biological activity, which is done by contacting virgin GAC with raw water and acetic acid for two to three weeks. The denitrifying bacteria used in the system are indigenous to the natural groundwater, meaning the system is naturally seeded with bacteria present in the groundwater.

During the pilot, a clone library analysis was performed on the bacteria within the biofilter to classify the various types of denitrifying bacteria present. The analysis revealed a diverse community of bacteria. At least 10 different denitrifying genera were identified, including Acidovorax, which comprised approximately 37 percent of the total bacteria in the FXB biofilters. The bacteria identified were gram negative, suggesting that they would be particularly sensitive to chemical disinfection, as gram-negative bacteria tend to have thin cell walls.

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L N)
    - Average – 75 (17)
    - Minimum – 44 (10)
    - Maximum – 89 (20)
- Co-contaminants
    - Perchlorate: 6 μg/L
    - DBCP: 0.025 μg/L

Treatment Technology Selection

The FXB process has been used successfully for removal of nitrate from drinking water supplies in Europe for decades, but has not yet been used full-scale in the United States. Reasons for using the FXB process at the Arlington Desalter include the following:

- Nitrate is not concentrated in a waste stream, as in RO or IX treatment, but is converted to nitrogen gas, which is released to the atmosphere—a harmless emission because the atmosphere is 78 percent nitrogen.
- Nitrate removal efficiencies are high. Typically, greater than 90 percent removal was achieved during the Arlington pilot studies and removal to non-detect levels was possible.
- The biodenitrification process results in simultaneous destruction of some anthropogenic contaminants. For example, perchlorate, found in the Arlington Desalter supply, was reduced to non-detect levels in the pilot study.

Historically, IX treatment has been the process of choice for nitrate removal in this country; however, it results in the replacement of nitrate with chloride in the drinking water supply. It is estimated that the equivalent IX system installed at the Arlington Desalter would add approximately 3 million pounds of salt to the basin annually. The removal of salt was the purpose for construction of the Arlington Desalter in the first place.

Treatment System Parameters

- Design Capacity
    - 1,670 gpm capacity
- Pretreatment
    - Substrate and nutrient addition
- Post-treatment
    - Degasification of N_2
    - Filtration for compliance with SWTR
- Water Recovery: expected 95%
- Media: GAC
- Treatment system footprint: TBD

Residuals Management

The residuals from this process, primarily biological growth, are accumulated in the filtration process. Once a predetermined pressure loss is experienced in filtration, the filters are backwashed to remove the accumulated solids. The solid laden backwash is then sent to the Santa Ana Regional Interceptor (SARI) line and ultimately disposed of off shore.

Technology Benefits and Drawbacks

Benefits

- Less expensive than other technologies
- No disposal of waste brine
- High water recovery rate: > 95%
- Can remove co-contaminants

Drawbacks

- No full-scale applications in operation
- More complicated permitting

Treatment Technology Costs

Detailed cost estimates are being developed. Detailed design of this system may occur in 2012, and full-scale construction of this system may be in 2013.

As an approximation, the below costs are based on a fixed-bed demonstration system for the removal of perchlorate with a similar empty bed contact time (EBCT) (Carollo Engineers 2008).

Capital Costs (2008 dollars)		
Flow Rate:	**1000 gpm**	**2000 gpm**
Total:	$4,193,000	$7,395,000
Direct Installed Costs:	$2,373,000	$4,200,000
O & M Costs (2008 dollars)		
Total Estimated Annual O&M Costs:	$175,000	$348,000
Chemicals ($/yr):	$161,000	$323,000
Other (GAC and Filter Sand) ($/yr):	$9,000	$17,000
Power ($/yr):	$5,000	$8,000
Total Costs (2008 dollars, 2.8% discount rate, 30-year lifecycle)		
Amortized Project Costs:	$209,000	$368,000
Estimated Annual Budget:	$384,000	$716,000
Total Treatment Costs ($/1000 gal):	$0.73	$0.68
Total Treatment Costs ($/AF):	$238	$222

Source*

Bernosky, J. (2010) Personal communication.
Brown, J. (2010) Personal communication.
Carollo Engineers. Final Report: Fixed-Bed Biological Nitrate Removal Pilot Testing at the Arlington Desalter Facility.
Carollo Engineers. (2008) Final Report: Direct Fixed-Bed Biological Perchlorate Destruction Demonstration. ESTCP Project ER-0544.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

3.5 Chemical Denitrification (CD)

Chemical denitrification can be accomplished with reduction of nitrate by metals. Various metals have been investigated for use in nitrate reduction including aluminum and iron (both Fe^o and Fe^{2+}), while copper, palladium, and rhodium can be used as catalysts in nitrate reduction (Shrimali & Singh 2001). The advantage of chemical denitrification over the removal technologies is that nitrate is converted to other nitrogen species rather than simply displaced to a concentrated waste stream that requires disposal. Problems with chemical denitrification of potable water are the reduction of nitrate beyond nitrogen gas to ammonia, partial denitrification, and insufficient nitrate removal (nitrite can be converted to nitrate with the use of chlorine in disinfection). No full-scale chemical denitrification systems have been installed in the United States for the removal of nitrate in potable water treatment. A significant body of research has explored the use of zero valent iron (ZVI) in denitrification. Several patented granular media options are also emerging, including SMI-III® (Sulfur Modified Iron), MicroNose™ Technology, and Cleanit®-LC.

Based on lab and pilot-scale studies, there is much variation in the configuration of chemical denitrification systems for nitrate removal from potable water. The generic mechanism of denitrification involves the transfer of electrons from an electron donating metal to nitrate. As in biological denitrification, nitrate is reduced in accordance with Eqn. 9. However, in contrast with biological denitrification, using chemical denitrification, the nitrogen in nitrate is often reduced to the least oxidized form, ammonium (Eqn. 9a) (Huang et al. 1998; Hao et al. 2005).

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \qquad \text{(Eqn. 9)}$$

$$NO_3^- \rightarrow NO_2^- \rightarrow NH_4^+ \qquad \text{(Eqn. 9a)}$$

Nitrate is exposed to an electron donating metal by passing the treatment stream through granular media. Particle size, surface area, and surface chemistry are important media characteristics related to the efficiency of nitrate removal.

3.5.1 Zero Valent Iron (ZVI)

Due to the extensive research focused on the use of zero valent iron (ZVI), ZVI will serve as a preliminary example. There is some variation in the use of ZVI. Forms of application include powdered iron, stabilized iron as nanoparticles, iron filings, and permeable reactive barriers (PRBs). Relevant reactions are listed in Eqns. 13 to 18 (Huang et al. 1998; Hao et al. 2005; Xiong et al. 2009). Nitrate can be reduced to nitrite (Eqn. 13), ammonia (Eqn. 14), or nitrogen gas (Eqn. 15) by ZVI. Following nitrate reduction to nitrite, nitrite can then be reduced to ammonia (Eqn. 16). Nitrate can also be reduced by the hydrogen gas that is produced from corrosion reactions (Eqn. 17) to ammonia (Eqn. 18).

$$Fe^o + NO_3^- + 2H^+ \rightarrow Fe^{2+} + NO_2^- + H_2O \qquad \text{(Eqn. 13)}$$

$$4Fe^{o} + NO_3^- + 10H^+ \rightarrow NH_4^+ + 4Fe^{2+} + 3H_2O \qquad \text{(Eqn. 14)}$$

$$5Fe^{o} + 2NO_3^- + 6H_2O \rightarrow N_{2(g)} + 5Fe^{2+} + 12OH^- \qquad \text{(Eqn. 15)}$$

$$3Fe^{o} + NO_2^- + 8H^+ \rightarrow 3Fe^{2+} + NH_4^+ + 2H_2O \qquad \text{(Eqn. 16)}$$

$$Fe^{o} + 2H^+ \rightarrow H_{2(g)} + Fe^{2+} \qquad \text{(Eqn. 17)}$$

$$NO_3^- + 4H_2 + 2H^+ \rightarrow NH_4^+ + 3H_2O \qquad \text{(Eqn. 18)}$$

The reduction of nitrate by iron is characterized by an increase in pH and consumption of hydrogen ions. pH is a significant controlling factor for this treatment method (Hao et al. 2005). The kinetics of nitrate reduction by ZVI have been thoroughly covered in the literature to determine the reaction rate under various conditions. For example, Alowitz & Scherer (2002) examined the nitrate reduction rates of three types of iron. Findings indicate that reduction rate increases with decreasing pH. Huang et al. (1998) investigated the use of powdered ZVI for the reduction of nitrate to ammonia. Highly pH dependent, nitrate reduction was kinetically favorable only at a pH below 4. The minimum ratio of iron to nitrate was 120 $m^2/mol\ NO_3^-$ for complete reduction within 1 hour. Nitrate reduction by ZVI can be optimized through pretreatment of iron particles. High temperature exposure to hydrogen gas and deposition of copper were explored separately as options for pretreatment of the iron surface (Liou et al. 2005). Both methods resulted in improvement of nitrate reduction in almost neutral solutions. The mechanism of improvement is due to the surface chemistry of iron. With a buildup of a surface oxide layer, the availability of sites for nitrate reduction decreases. Hydrogen gas pretreatment reduces the oxide layer, while deposited copper serves as a catalyst for the transfer of electrons. In their investigation of stabilized ZVI nanoparticles, Xiong et al. (2009) found that the end product of denitrification (nitrogen gas versus ammonium) could be controlled by the iron to nitrate ratio and the use of catalysts.

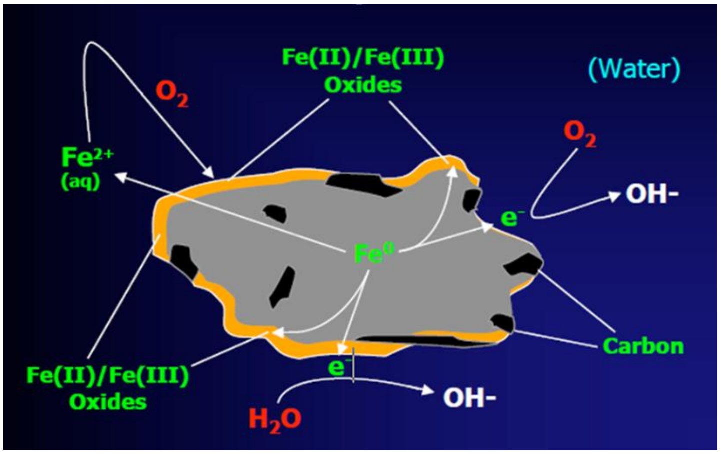

Figure 15. Surface chemistry of ZVI particles. (Source: reprinted with permission, Chiu 2009.)

Examination of the surface chemistry of ZVI particles is of the utmost importance to model and understand its use in the reduction of nitrate. Illustrated in Figure 15, relevant factors include corrosion of ZVI, complexation with water, surface complexation, reduction, precipitation, and adsorption. In the corrosion of ZVI, the formation of “green rusts” and “suspended green particles” is associated with stabilization of pH and steady decrease in nitrate (Choe et al. 2004). For nitrate reduction to occur, contact with the reducing agent is required. Reduction of nitrate by ZVI or by any surface bound species requires access to surface sites. Competition for surface sites can impede nitrate reduction; Moore & Young (2005) examined chloride as a potential competitor. Results indicate a minimal impact on nitrate removal; however, other competing ions could be important regarding both competition for adsorption sites and reduction.

3.5.2 Catalytic Denitrification

An extension of chemical denitrification, catalytic denitrification involves metal reduction of nitrate in the presence of a catalyst. Extensive research has investigated catalytic denitrification which may become more readily applicable to potable water treatment with further advances (Reddy & Lin 2000; Pintar et al. 2001; Gavagnin et al. 2002; Lemaignen et al. 2002; Pirkanniemi & Sillanpaa 2002; Chen et al. 2003; Palomares et al. 2003; Pintar 2003; Constantinou et al. 2007; Sun et al. 2010.)

3.5.3 Chemical Denitrification - Design Considerations

Table 14 summarizes key design considerations in the application of chemical denitrification for nitrate removal from potable water.

Table 14. Summary of design considerations for chemical denitrification.

Pretreatment	• pH adjustment
Post-Treatment	• Filtration for iron removal • pH adjustment • Chlorine addition for disinfection and oxidation of iron • Gas stripping or breakpoint chlorination (for ammonia)
Chemical Usage	• pH adjustment (acids and bases) • Disinfection and oxidation of iron (chlorine)
O&M	• Constant monitoring required to ensure efficient nitrate reduction ○ Nitrate levels ○ Oxidation reduction potential (ORP) • Monitoring of nitrite and ammonia will also be necessary due to the potential for incomplete denitrification • Management of chemicals ○ pH adjustment ○ Disinfection • Waste media and backwash water storage and disposal
System Components/Design Parameters	• Ratio of electron donor to nitrate for desired removal • Hydraulic Loading Rate (HLR), Empty Bed Contact Time (EBCT) • Reactor configuration (up-flow, down-flow, in series, in parallel) • Gas stripping or breakpoint chlorination to remove ammonia (if ammonia is the end-product) • Monitoring equipment • Filtration to remove iron • pH adjustment (decreased in pretreatment and increased before distribution) • Disinfection
Waste Management and Disposal	• Spent media disposal • Iron sludge management • Backwash water • No waste brine or concentrate as in removal processes
Limitations	• No examples of full-scale application for nitrate treatment ○ Unknown reliability for full-scale treatment ○ Unknown costs and operational complications • Potential for incomplete denitrification

Water Quality

The performance of chemical denitrification systems can be affected by pH, temperature, potential interference by co-contaminants, and the availability of surface sites. The reduction of nitrate by iron is characterized by an increase in pH and consumption of hydrogen ions. Thus, pH is a significant controlling factor for this treatment method (Hao et al. 2005). Alowitz & Scherer (2002) examined the nitrate reduction rates of three types of iron. Findings indicate that reduction rate increases with decreasing pH. If nitrate in the water does not come in contact with the electron donor, then reduction will not be possible. The build-up of precipitates can negatively impact nitrate reduction. The appropriate iron to nitrate ratio will be based on influent and target nitrate concentrations. Product water quality will require monitoring for nitrate, nitrite, and ammonia.

System Components and Site Considerations

Major system components include chemical storage (for pH adjustment and disinfection), the column containing the media, and post-treatment disinfection. With reduction to ammonia, post-treatment ammonia stripping may also be necessary. Design constraints include optimal temperature, sufficient EBCT length, avoidance of incomplete denitrification, monitoring requirements (nitrate, nitrite, and ammonia), appropriate iron to nitrate ratio, and adequate chlorine dosing for disinfection and iron oxidation (DSWA 2010). Incomplete denitrification, which can be associated with higher dissolved oxygen (DO) levels, can result in the production of greenhouse gases (GHG) such as NO and N_2O. Dissolved oxygen levels can be decreased using reducing agents or through the provision of sufficient electron donor to enable depletion of DO (Meyer et al. 2010, regarding biological denitrification).

Residuals Management and Disposal

In contrast to IX and the membrane technologies, the burdens of brine and concentrate disposal are minimized because nitrate is reduced through chemical denitrification. There is no concentrated brine solution requiring costly disposal. However, disposal of backwash water, spent media, and iron sludge is necessary.

Maintenance, Monitoring, and Operational Complexity

With the possibility of incomplete denitrification, monitoring is required to ensure that product water does not contain high levels of ammonia or nitrite. Exposure of these nitrogen species to chlorine in disinfection or oxygen downstream can lead to nitrification (oxidation back to nitrate) in the distribution system, unless controlled. Additional O&M demands include management of chemicals (e.g., acids, bases, and chlorine), backwashing the column to maintain flow and performance, and waste management. Despite having no full-scale installation for comparison, overall, chemical denitrification may potentially be less operationally complex than biological denitrification.

3.5.4 Chemical Denitrification - Emerging Technologies

Sulfur-Modified Iron (SMI) Media

Chemical reduction of nitrate has been demonstrated for potable water treatment using sulfur-modified iron granular media (DSWA 2010). Certified to the NSF/ANSI Standard 61 for use in drinking water treatment, SMI-III® is a patented media that is recyclable and offers the advantage of multiple contaminant removal (SMI-PS 2009). Arsenic and metals can be removed via adsorption (hexavalent chromium can also be reduced and precipitated), while nitrate is reduced to ammonia (Prima Environmental N.D.) or nitrogen gas (DSWA 2010). "The SMI-III® manufacturer believes sulfur modification regulates the environment of reactions to achieve greater and a consistent nitrate reduction" (DSWA 2010, p. 9). Nitrate reduction is governed by the following reactions (SMI-PS 2009):

$$4Fe^0 + NO_3^- + 10H^+ \rightarrow 4Fe^{2+} + NH_4^+ + 3H_2O \qquad \text{(Eqn. 19)}$$

$$5Fe^0 + 2NO_3^- + 12H^+ \rightarrow 5Fe^{2+} + N_2 + 6H_2O \qquad \text{(Eqn. 20)}$$

Key advantages of SMI-III® are the ability to remove multiple contaminants simultaneously and the limited waste disposal costs relative to other nitrate removal options (no brine waste stream is produced) (DSWA 2010). Some previous research indicates inconsistent and insufficient nitrate removal to meet potable water regulations (DSWA 2010).

Damon S. Williams Associates (DSWA) and the City of Ripon, CA, conducted a pilot study investigating the use of SMI-III® in potable water treatment. Findings suggest that this treatment option may be suitable for source nitrate concentrations slightly above the MCL (up to 70 mg/L as nitrate (16 mg/L as N)). Phase A of the pilot study was operated with the SMI-III® media in an up-flow fluidized bed across a pH range of 6.0 – 6.8 and with an EBCT of 15 to 30 minutes. Phase B tested improved media performance across the same pH range and with an EBCT of 30 minutes. Figure 16 displays a process schematic of the SMI-III® process.

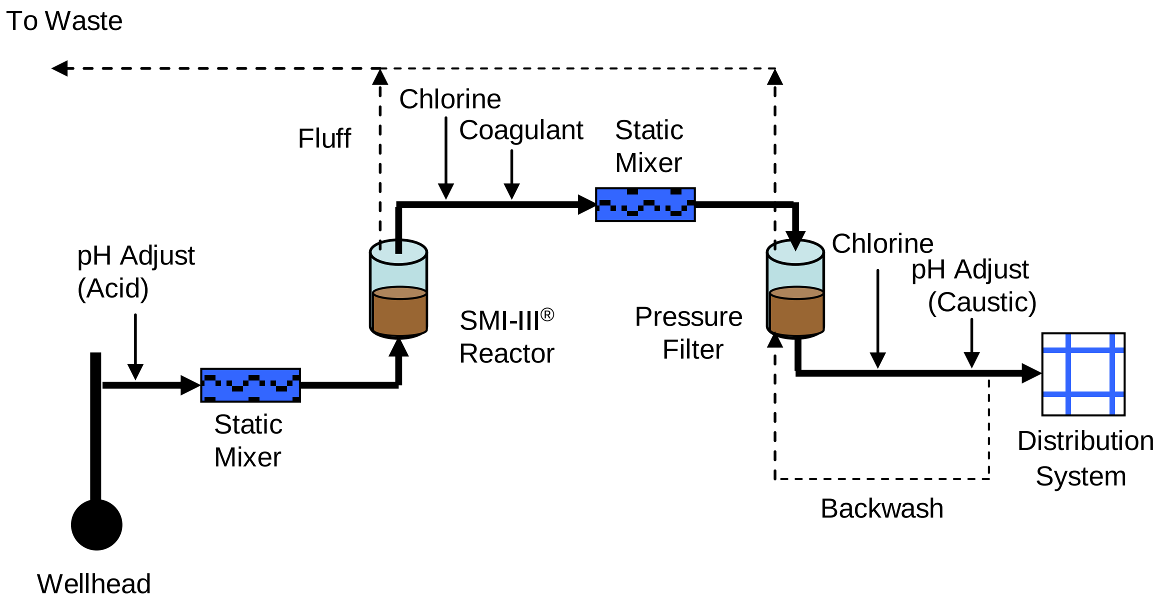

Figure 16. Process schematic for denitrification using SMI-III®. (Source: DSWA 2010.)

The media was fluffed (backwashed in up-flow mode) daily to remove oxidized iron and to avoid media agglomeration. The SMI-III® reactor was followed by coagulation and filtration for the removal of iron, arsenic, and other constituents. In phase A, the greatest nitrate removal (18 mg/L as nitrate, 4 mg/L as N) was insufficient to meet the 20% safety margin of the project (a goal of 36 mg/L as nitrate, 8 mg/L as N), with a starting nitrate concentration of 60 mg/L as nitrate (13.5 mg/L as N). Problems in phase B resulted in operation interruption; however, a maximum nitrate removal of 24 mg/L as nitrate (5.4 mg/L as N) was achieved. When the pH was reduced to 6, the system did produce water with nitrate below the MCL and the project goal nitrate level, on two occasions. Temperature was found to have the most significant impact on removal efficiency. Nitrate reduction improved with increasing source water temperatures.

Design constraints include temperature, EBCT, avoidance of incomplete denitrification, determination of the appropriate iron to nitrate ratio, chlorine requirements for released iron oxidation, clogging from precipitates, and pH. Nitrate reduction typically increases with decreasing pH, increasing EBCT, and increasing temperature (DSWA 2010).

SMI - Cost Considerations

Cost analysis in the City of Ripon report included acid, caustic soda, and media and excluded labor and waste management. In both phases, minimal variation in nitrate reduction was found with operation at a pH of 6 versus a pH of 6.8. Operation at the higher pH minimizes costs due to prolonging media life and decreasing chemical input. The production costs for operation at a pH of 6 and 6.8 were estimated to be $2.24/1000 gal. ($729/AF) and $0.88/1000 gal. ($287/AF), respectively (DSWA 2010). Due to dissolution of the media over time, media disposal is not expected to be necessary. Dissolved iron is oxidized and then removed through filtration. In the Ripon pilot study, backwash water was discharged to the sewer; however, when "…direct sewer disposal is not feasible, the backwash water must undergo solid/liquid separation with the decant liquids recycled to the head of the treatment system and the dewatered solids sent to an appropriate landfill for disposal" (DSWA 2010, p. 82). If the waste is deemed hazardous, disposal can be a major cost consideration.

Granular Clay Media

Certified to the NSF/ANSI Standard 61 for use in drinking water treatment, MicroNose™ Technology media is currently being examined in Manteca, California, for nitrate removal in potable water treatment (MicroNose™ 2010). The media consists of "absorbent and permeable pottery granules which function similar to the mucous membrane in the human nose" (University of Hawaii 2006). Limited information is available on this emerging technology. The company states, "MicroNose™ Technology media removes heavy metals, such as arsenic, manganese, and lead as well as nitrates through non-chemical processes" (MicroNose™ 2010). MicroNose™ offers removal of multiple contaminants concomitantly and claims to be cost-effective, suitable for nitrate removal, and a green technology. Additional information is needed to assess the design considerations, costs, and applications of MicroNose™ for nitrate removal from potable water.

Powdered Metal Media

As an emerging technology, Cleanit®-LC (from North American Höganäs) is a metal powder with the potential to achieve 60 – 90% nitrate removal (Lavis 2010). Certified to the NSF/ANSI Standard 61 for use in drinking water treatment, this proprietary iron-based powder could be used for removal of co-contaminants in addition to nitrate, including "arsenic, heavy metals, phosphates and pathogens" (Lavis 2010), and potentially hexavalent chromium. Cleanit®-LC media is characterized by the following: a density of 1800 – 2100 kg/m^3, a particle size of 150 – 850 microns, and a porosity of 60%. The powder can be used to adsorb arsenic in less than 10 minutes, with an arsenic capacity of 4 – 8 mg/g powder.

With an up-flow configuration, the treatment stream is pumped through a column containing the media, maximizing surface contact. The most significant consideration is the EBCT. At the particle surface, nitrate is reduced to nitrogen gas with EBCTs of 10 – 30 minutes. Additional key design factors are pH and temperature. In contrast to membrane technologies, the burdens of disposal are minimized because nitrate is reduced rather than removed. There is no concentrated brine solution requiring costly disposal. Preliminary third party results indicate nitrate removal over 7 months to below the MCL (North American Höganäs, data). However, as a new product on the market, further research is required to assess Cleanit®-LC for the removal of nitrate and other constituents in potable water treatment.

3.5.5 Chemical Denitrification - Cost Considerations

For efficient operation of a chemical denitrification system, maintaining efficient nitrate reduction is essential. Optimal performance includes managing pre and post-treatment to provide appropriate environmental conditions, while meeting necessary potable water guidelines. Factors affecting system cost can include facility size (flow rate), source water quality (including nitrate concentration and co-contaminants), environmental factors (temperature and pH), and target effluent nitrate concentration.

Capital costs for chemical denitrification include land, housing, piping, media, storage tanks, O&M equipment, preliminary testing (through pilot studies), permits, and operator training. O&M costs include pre and post-treatment, media replenishment and disposal, backwashing, chemical use (e.g., pH adjustment, chlorine), repair, monitoring, maintenance, power, and labor.

The availability of published cost information for chemical denitrification is strictly limited to pilot-scale studies, due to the lack of full-scale systems. Cost analysis in the City of Ripon report included acid, caustic soda, and media and excluded labor and waste management. In both phases, minimal variation in nitrate reduction was found with operation at a pH of 6 versus a pH of 6.8. Operation at the higher pH minimizes costs due to prolonging media life and decreasing chemical input. The production costs for operation at a pH of 6 and 6.8 were estimated to be $2.24/1000 gal. ($729/AF) and $0.88/1000 gal. ($287/AF), respectively (DSWA 2010). Due to dissolution of the media over time, media disposal is not expected to be necessary. Dissolved iron is oxidized and then removed through filtration. In the Ripon pilot study backwash water was discharged to the sewer. However, when "…direct sewer disposal is not feasible, the backwash water must undergo solid/liquid separation with the decant liquids recycled to the head of the treatment system and the dewatered solids sent to an appropriate landfill for disposal" (DSWA 2010, p. 82). If the waste is deemed hazardous, disposal can be a major cost consideration. A detailed discussion of treatment costs is included in Section 6 Treatment Cost Analysis.

3.5.6 Chemical Denitrification - Selected Research

Table A.5 of the Appendix lists recent research studies relevant to the use of chemical denitrification in potable water treatment.

3.5.7 Chemical Denitrification - Summary of Advantages and Disadvantages

A summary of advantages and disadvantages of chemical denitrification in comparison with other treatment options is listed in Table A.6 of the Appendix. Advantages of chemical denitrification for nitrate removal from potable water include the conversion of nitrate to other nitrogen species (no brine or concentrate waste stream), the potential for more sustainable treatment, and the ability to remove multiple contaminants.

Problems with chemical denitrification of potable water are the potential reduction of nitrate beyond nitrogen gas to ammonia, the possibility of partial denitrification, and the associated production of GHGs, and the lack of full-scale chemical denitrification systems.

3.6 Brine Treatment Alternatives and Hybrid Treatment Systems

Brine or concentrate disposal can be a great concern with the use of the removal processes, IX, RO, and ED/EDR, especially for inland communities. Brine treatment and recycling alternatives have the potential to address disposal concerns, improving sustainability and decreasing operating costs. Hybrid systems, combining different nitrate treatment technologies, have been explored to include the advantages of multiple treatment options, while avoiding their respective disadvantages (Pintar et al. 2001; Wisniewski et al. 2001; Matos et al. 2006; Kabay et al. 2007; Van Ginkel et al. 2008). Table 15 lists a selection of additional research related to brine treatment alternatives and the use of hybrid systems to improve nitrate treatment performance.

The combination of denitrification methods with removal technologies enables resolution of common problems with each option. The brine waste stream from IX, for example can be treated using biological denitrification to reduce nitrate to nitrogen gas. Biological denitrification of IX waste brine using an up-flow sludge blanket reactor (USBR) was demonstrated at the lab-scale by van der Hoek & Klapwijk (1987). Clifford & Liu (1993) implemented a lab-scale sequencing batch reactor (SBR) for the denitrification of waste brine resulting in the ability to recycle the treated waste brine for 15 regeneration cycles. The SBR process was pilot tested in McFarland, CA (Liu & Clifford 1996). With denitrification of spent brine followed by reuse, a 95% decrease was achieved in salt waste. Use of a membrane bio-reactor in this context has also been explored (Bae et al. 2002; Chung et al. 2007; Van Ginkel et al. 2008). Through the reduction or destruction of nitrate in spent IX brine, disposal needs may be significantly reduced as treated brine can be repeatedly recycled back for use in resin regeneration. Additionally, through treatment of RO or ED/EDR waste concentrate, removal of nitrate from the waste stream may improve disposal options.

Table 15. Selected research on brine treatment alternatives and hybrid systems for nitrate treatment of potable water.

IX and Catalytic Reduction	Pintar et al. (2001)
RO and EDR	EET Corporation (2008)
IX and EDR	Kabay et al. (2007)
ED and MBR	Wisniewski et al. (2001)
IX and Biological Denitrification	Van der Hoek & Klapwijk (1987), Clifford & Liu (1993), Bae et al. (2002), and Van Ginkel et al. (2008)
RO and VSEP	Lozier et al. (N.D.)
Electrochemical destruction of nitrate (in IX brine)	Yu & Kupferle (2008), Dortsiou et al. (2009), Goltz & Parker, 2010/2011, and Ionex SG Limited (2011)

3.6.1 *Electrochemical Destruction of Nitrate in Waste Brine*

Research by two different groups is focused on electrochemical destruction of nitrate for the treatment of waste brine.

First, two major companies are collaborating on the development of a system which incorporates IX with electrochemical destruction of nitrate. With the removal of nitrate from spent brine, the brine can be recycled for reuse in regeneration (Goltz 2010; Goltz & Parker 2010/2011). Spent brine is treated following IX treatment. With electricity distributed over a high surface area electrode, upon contacting the electrode surface, nitrate in the brine waste is reduced to nitrogen gas. In the overall reaction, nitrate is reduced and water is oxidized producing nitrogen gas and oxygen. Possible reduction to ammonia/ammonium is accounted for in the process. Laboratory tests have been successful and planning of site tests is underway to examine system performance across a variety of raw water characteristics. The potential benefits of this brine treatment alternative are in the increased sustainability and the decreased disposal costs. The objective is to optimize the process such that the savings on disposal costs, especially for inland communities, outweigh the increased electricity costs of the process.

Second, a company based in the United Kingdom, Ionex SG Limited, has developed and is currently testing a patented brine treatment system utilizing an electrochemical cell for the destruction of nitrate (Ionex SG Limited 2011). Following resin regeneration, spent brine is passed through the cell and nitrate is converted to nitrogen and oxygen gases through interaction with a rhodium catalyst (Figure 17). Treated brine can be subsequently reused for regeneration, minimizing waste brine volume as well as the overall salt consumption of the ion exchange system. The lifespan of the cell has been laboratory tested and is estimated to operate efficiently for at least 15 years through a mechanism of automatic maintenance of the reactive surface (Tucker et al. 2004; Ionex SG Limited 2011). The brine treatment system is currently being pilot tested at a location in California through collaboration with UC Davis. A cost benefit analysis by Ionex indicates that brine disposal costs would be significantly reduced; chemical and power costs of the brine treatment system balance with existing chemical costs for typical ion exchange systems.

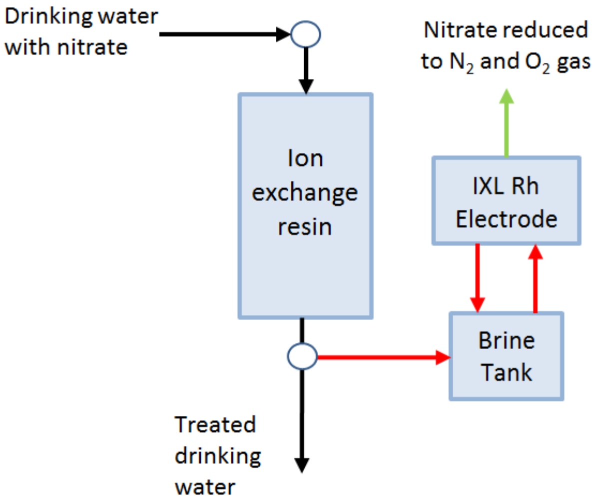

Figure 17. Schematic of the brine treatment system developed by Ionex SG Limited. (Source: Ionex SG Limited 2011.)

3.6.2 Catalytic Treatment of Waste Brine

Calgon Carbon is working with the University of Illinois to develop a brine recovery system with the goal of up to 90% brine recovery for reuse using Pd-based catalytic treatment (Drewry 2010). The following information was provided by Dr. Charles Werth (2010) from the University of Illinois.

As illustrated in Figure 18, "The spent brine solution is equilibrated with hydrogen in a gas-liquid membrane, and subsequently treated to remove nitrate in the packed bed catalyst system containing Pd-In on granular activated carbon (GAC). The treated brine is then put back into a holding tank and reused when ion exchange breakthrough occurs again" (Werth 2010).

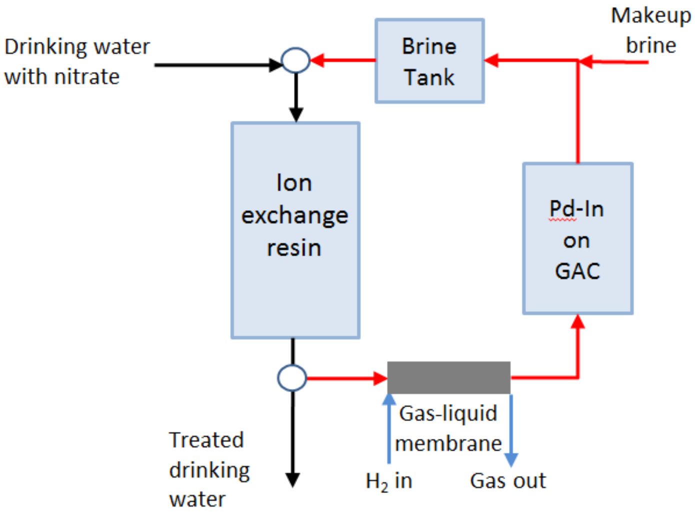

Figure 18. Schematic of an ion exchange system with brine regeneration coupled with catalytic treatment of brine for reuse. (Source: Werth 2010.)

Advantages of this brine treatment alternative include conversion of nitrate mainly to nitrogen gas, avoidance of the need for brine waste disposal, and rapid reduction of nitrate with the Pd-based catalyst. Disadvantages include the use of flammable hydrogen gas, some conversion of nitrate to ammonia, "the cost of Pd and potential fouling of the catalyst which requires regeneration using a strong oxidant like hypochlorite (Chaplin et al. 2007)" (Werth 2010). Catalytic removal of nitrate has not yet been implemented at the full-scale, but "catalytic systems have been used to remove chlorinated solvents at contaminated groundwater field sites, and they appear to be economically competitive (Davie et al. 2008)" (Werth 2010). Dr. Werth stated, "Catalytic systems show great promise for removing nitrate from IX brines, but pilot-scale studies are needed to evaluate the economic viability."

3.7 Residential Treatment (Point-of-Use, Point-of-Entry)

Point-of-Use (POU) and Point-of-Entry (POE) water treatment devices can be used to address high nitrate levels and other constituents of concern at the residential scale. A POU treatment device is installed for the purpose of reducing contaminants in drinking water at a single tap, typically the kitchen tap. A POE treatment device is installed for the purpose of reducing contaminants in drinking water entering a house or building.

Treatment technologies for POU and POE systems, used to address nitrate contamination, include IX, RO, and distillation (Mahler et al. 2007). IX is generally considered more for POE than for POU and requires disposal of concentrated waste brine. RO systems require more maintenance and have lower water recovery, resulting in a larger waste volume (Mahler et al. 2007). However, according to the New Hampshire Department of Environmental Services (2006), RO "is the most cost-effective method for producing only a few gallons of treated water per day" (NHDES 2006). While distillation can require lower maintenance, energy demands are higher than the other options. Distillation systems are generally intended as POU devices as they remove all minerals and produce water that is aggressive towards plumbing materials.

POU units are installed either under the counter or on the counter top, preferably by a licensed professional. The treatment units generally consist of several stages; for example, a POU RO system can consist of a pre-treatment filter, an RO stage, and a post-treatment filter. The system can also include a storage tank to hold treated water and a conductivity meter to indicate when maintenance is required. POE units, for the treatment of all water entering a building, are larger and require more piping.

Certification to the relevant ANSI/NSF standards by an ANSI accredited third party certifier ensures the safety and performance of the residential treatment systems. In the U.S., the following certifiers have been accredited by ANSI to certify drinking water treatment systems:

- Canadian Standards Association International (www.csa-international.org);
- International Association of Plumbing & Mechanical Officials (www.iapmo.org);
- NSF International (www.nsf.org);
- Underwriters Laboratories Inc. (www.ul.com); and

- Water Quality Association (www.wqa.org).

Numerous RO devices for nitrate removal are certified to the ANSI/NSF standard specific to RO POU devices: NSF Standard 58 - Reverse osmosis drinking water treatment systems (NSF 2009). Additionally, the Water Quality Association, an accredited certifier, lists two POU ion exchange devices for nitrate removal that are certified to NSF Standard 53: Drinking Water Treatment Units – Health Effects (Water Quality Association 2011). All the technologies listed above are capable of reducing nitrate levels; however, proper maintenance of the treatment equipment is fundamental to ensure the provision of safe drinking water. Additionally, it is important to conduct periodic testing (annually or as recommended by the manufacturer) using an accredited laboratory on both the influent water and the water produced by the treatment system to verify that it is working effectively.

CDPH provides a list of approved POU devices for nitrate treatment consisting predominantly of RO devices (CDPH 2011). Published cost information for POU systems is listed in Table 16. Based on a survey in Minnesota, “the average cost of nitrate removal systems was $800 to install and $100 per year to maintain” (Lewandowski 2008, p. 92A). Providing a more detailed cost analysis, the U.S. EPA has developed a cost estimating tool for the use of POU and POE devices (U.S. EPA 2007; U.S. EPA 2011).

It is important to note that water systems are responsible for meeting federal, state, and local requirements and the allowable uses of POU/POE devices vary by state. Current California regulations enable small public water systems to use POU devices to meet the nitrate MCL for up to three years, with certain restrictions, including the following (California Code of Regulations 2011, p. 2):

“…a public water system may be permitted to use point-of-use treatment devices (POUs) in lieu of centralized treatment for compliance with one or more maximum contaminant levels… if;

(1) the water system serves fewer than 200 service connections,

(2) the water system meets the requirements of this Article,

(3) the water system has demonstrated to the Department that centralized treatment, for the contaminants of concern, is not economically feasible within three years of the water system’s submittal of its application for a permit amendment to use POUs,

… no longer than three years or until funding for the total cost of constructing a project for centralized treatment or access to an alternative source of water is available, whichever occurs first….”

Table 16. Costs of POU treatment for nitrate removal.[1]

	Upfront Investment	Annual Costs	Comments
Ion Exchange	\$660 – \$2425	Salt costs (\$3.30 – \$4.40/bag)	Requires disposal of brine waste
Distillation 4-10 gal/d	\$275 – \$1650	\$440 – \$550/yr + electricity	Requires scale removal, higher energy use
Reverse Osmosis 2-10 gal/d	\$330 – \$1430	\$110 – \$330/yr + electricity	Requires filter replacement, high maintenance, lower water recovery

Costs have been adjusted to 2010 dollars, unless indicated otherwise.

[1] Mahler et al. (2007).

4 Tulare Lake Basin and Salinas Valley - Water Quality Analysis

As mentioned above in the discussion of treatment technologies, water quality is a key variable in both treatment selection and cost. Table 17 is a summary of water quality data of high nitrate wells across both the Tulare Lake Basin and the Salinas Valley. Constituents that interfere with treatment (e.g., sulfate) or affect treatment technology selection (e.g., TDS) are included as well as co-contaminants. Table 17 lists water quality data for all wells with raw water monitoring data in the CDPH water quality database from 2006 – 2010 having nitrate levels above the MCL, including both abandoned and inactive wells, to enable consideration of the complete range of scenarios. This water quality summary highlights the wide range of water quality characteristics encountered in the region of interest. The GAMA Priority Basin Projects provide additional water quality information including the basins within the study area (USGS 2011).

Table 17. Summary of water quality data of high nitrate wells in the Tulare Lake Basin and Salinas Valley. (Source: CDPH PICME and WQM databases.)

Constituent	Average	Min	Max	# Samples	# PWS	# Sources	Units
Nitrate as NO_3	64.69	45.1	402	1705	159	209	mg/L (as NO_3)
Ammonia	1	-	-	2	1	1	mg/L (NH_3-N)
Sulfate	116.43	3.7	2300	1594	78	111	mg/L
pH (Lab)	7.91	6.38	9.6	1582	87	120	
pH (Field)	7.12	6.52	7.85	1186	3	7	
Temperature	19.71	3	24.4	780	5	10	^{o}C
Hardness	344.7	6.7	5000	1562	88	121	mg/L as $CaCO_3$
Iron	126.34	0	5700	1586	84	117	ug/L
Manganese	11.86	0	400	1559	83	116	ug/L
Chloride	100.04	3.5	16000	2502	78	111	mg/L
Silica	1693.31	28	21000	71	4	5	mg/L
TDS (Conductance)	856.65	23	43600	4176	101	136	
TDS (TDS)	780.04	120	28700	2143	78	111	mg/L
Alkalinity (Total)	203.85	32	340	1572	79	112	mg/L as $CaCO_3$
Arsenic	3.24	0	53	2228	99	137	ug/L
Chromium (Total)	2.9	0	45	1785	96	132	ug/L
Chromium (Hex)	1.19	0.2	3.5	20	2	2	ug/L
Perchlorate	3.09	0	2000	2637	107	138	ug/L
Uranium	15.64	8.4	58	532	3	5	ug/L
TOC	3.61	1.2	6.6	85	1	1	mg/L
Turbidity (Lab)	0.55	0	8.9	1494	74	108	NTU

4.1 Water Quality - Treatment Interference

The most commonly employed nitrate treatment method, ion exchange (IX), is highly sensitive to competing anions in the treatment stream. With the use of generic anion exchange resin for nitrate treatment, sulfate will out-compete nitrate for resin sites and, as sulfate levels in source water increase, the resin capacity for nitrate will decrease. Under such circumstances, the cost of more frequent regeneration, associated chemicals and disposal may make ion exchange less feasible. Nitrate dumping can also be a problem, resulting in effluent nitrate levels higher than influent levels due to sulfate displacing nitrate on the resin. To address these concerns, nitrate selective resin can be used, which is more expensive, but can maintain a higher capacity for nitrate. At the highest levels in the above listed sulfate range, alternative treatment options would likely become more cost-effective.

High hardness, silica, TSS, turbidity, manganese, and iron can impact most nitrate treatment options requiring additional steps for pretreatment such as filtration and anti-scalant addition to avoid resin (IX) or membrane fouling (RO and EDR). Cleaning and post-treatment requirements can also be affected, thus increasing system complexity and overall costs.

4.2 Water Quality - Co-contaminants

The need for multiple contaminant removal is a key factor in the selection of the most appropriate treatment option. While IX can effectively remove several co-contaminants, this technology cannot address all constituents of concern and alternative options should be considered with particularly poor quality waters. IX has been used for simultaneous removal of perchlorate and nitrate, and arsenic and nitrate; however, specific resins may be required to optimize multiple contaminant removal, resulting in higher costs than IX for nitrate removal alone. Figures 19 – 22 highlight the co-occurrence of nitrate with other constituents of interest in the Tulare Lake Basin and Salinas Valley.

Figure 19 maps raw water nitrate levels above the MCL (45 mg/L as nitrate) including both active and inactive well data from 2006 – 2010. Figure 20 maps high nitrate wells with high arsenic levels for which IX may still be considered or for which RO may be a more suitable treatment option, depending on the priorities of a given system. Figure 21 maps high nitrate wells with high perchlorate levels; again IX may be considered under such circumstances. Alternatively, biological denitrification may be implemented to simultaneously remove both constituents while avoiding the brine disposal problem. Last, Figure 22 maps high nitrate wells in which at least 1 of 4 major pesticides has been detected (bromacil, simazine, atrazine, and DBCP). The co-occurrence of nitrate and pesticides is important on two fronts. With pesticide levels above the MCL, treatment requirements will change. RO may be implemented to address the multiple contaminants or IX could be used for nitrate and activated carbon for pesticides. Regardless of the selected option, treatment for nitrate and pesticides will be more expensive than treatment for only nitrate. Additionally, the co-occurrence may be indicative of the source of nitrate contamination.

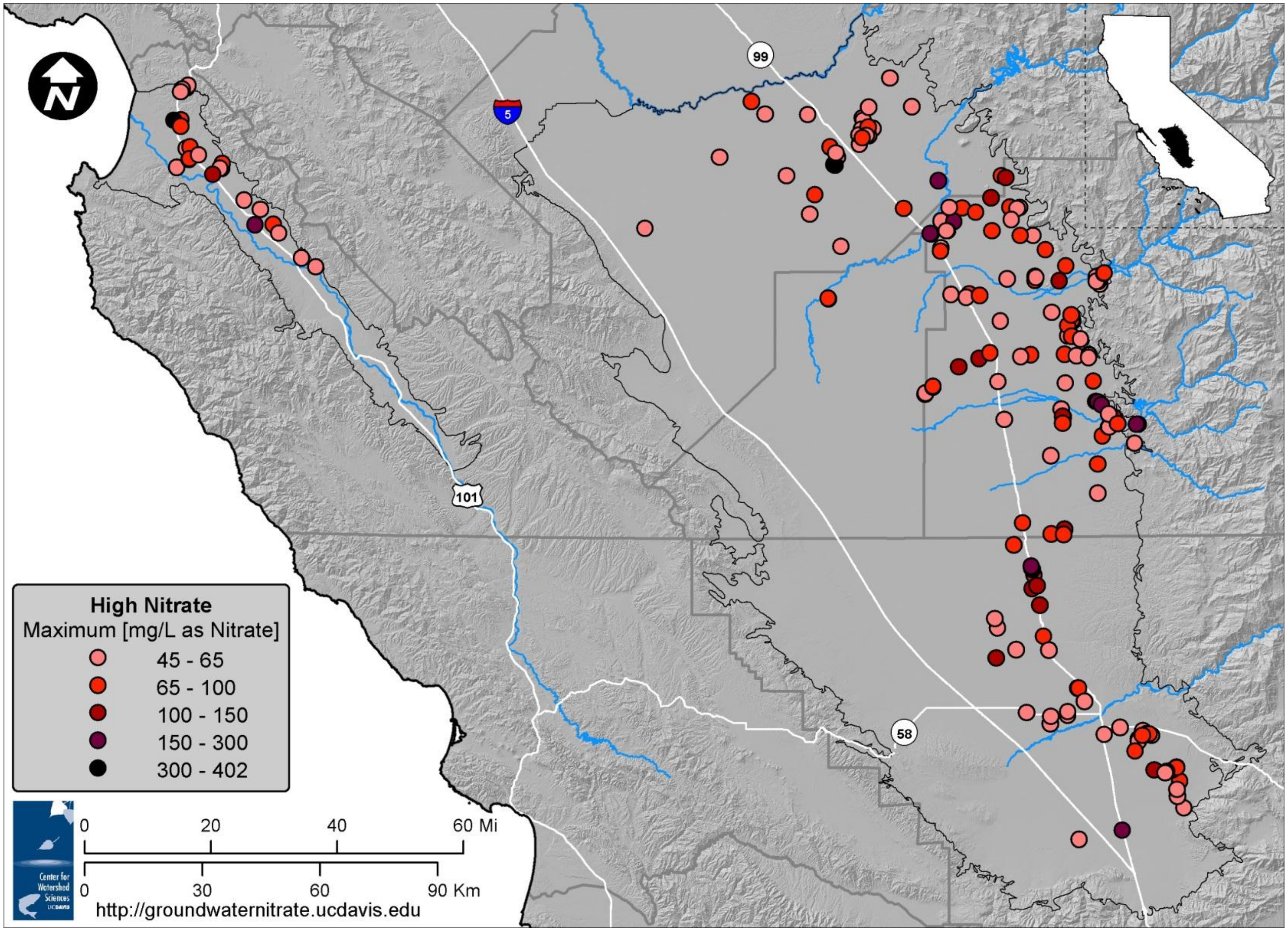

Figure 19. Raw water nitrate levels exceeding the MCL (45 mg/L as nitrate). (Source: CDPH PICME and WQM Databases.)

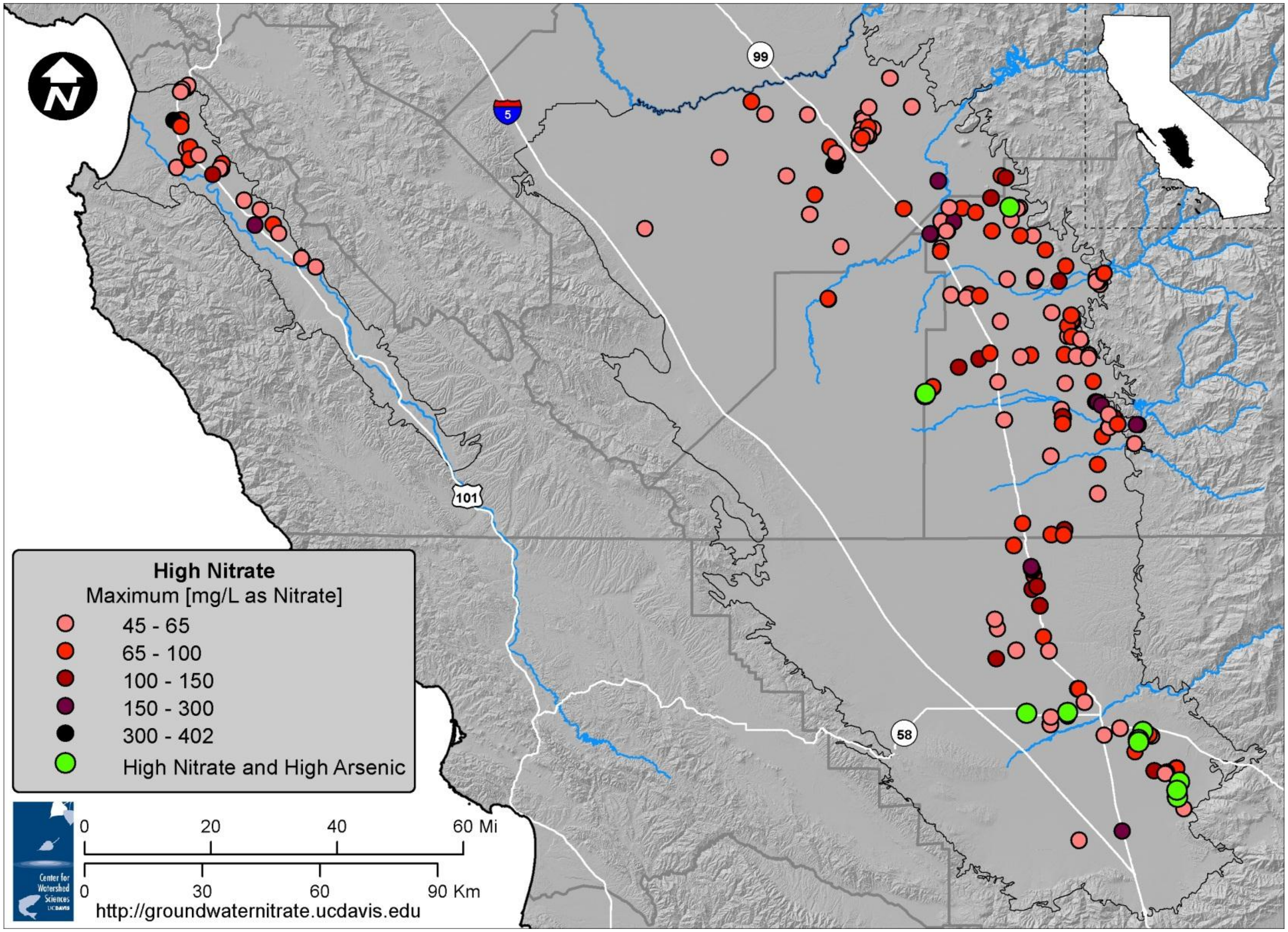

Figure 20. Raw water high nitrate wells with high arsenic levels. (Source: CDPH PICME and WQM Databases.)

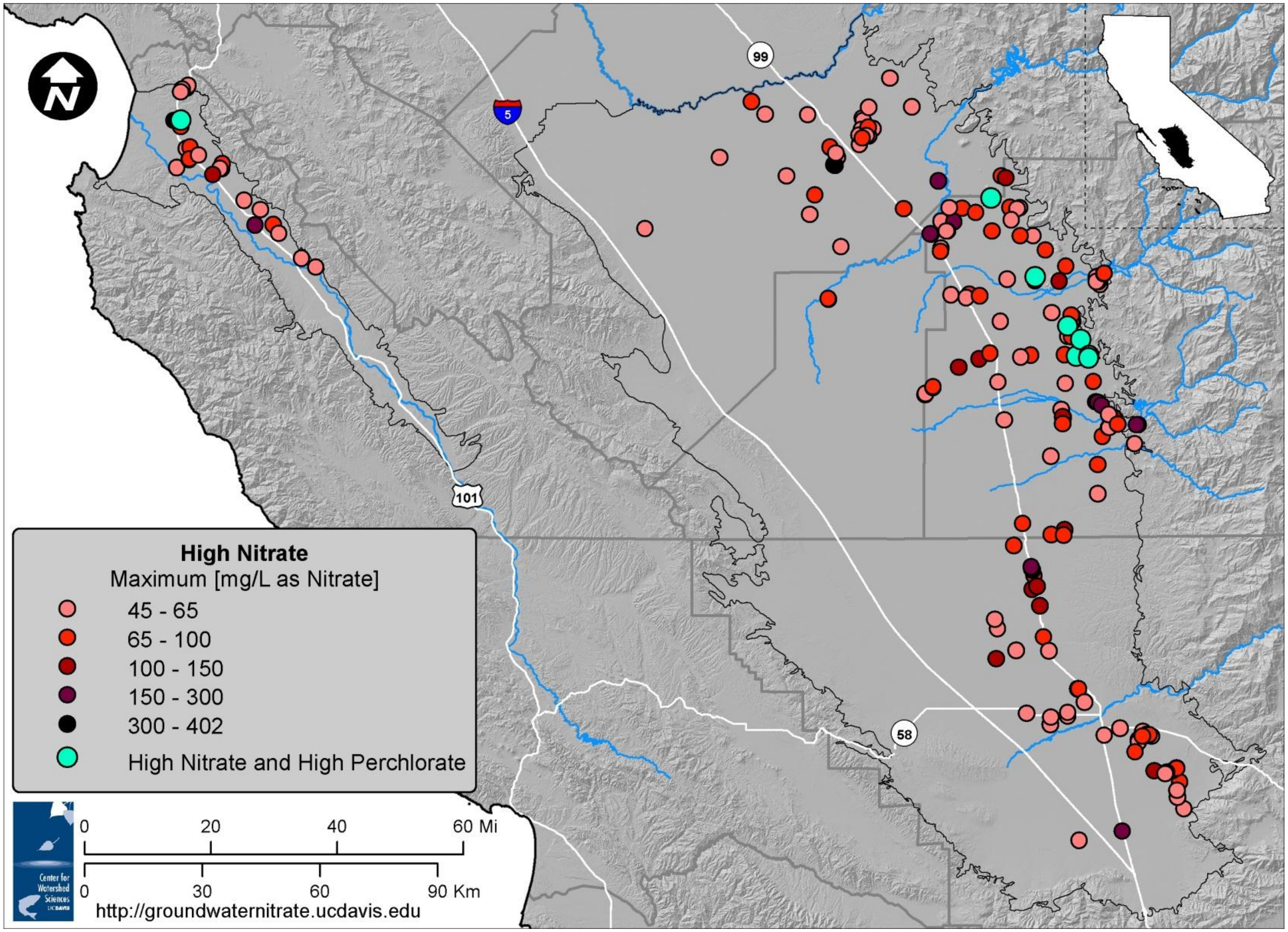

Figure 21. Raw water high nitrate wells with high perchlorate levels. (Source: CDPH PICME and WQM Databases.)

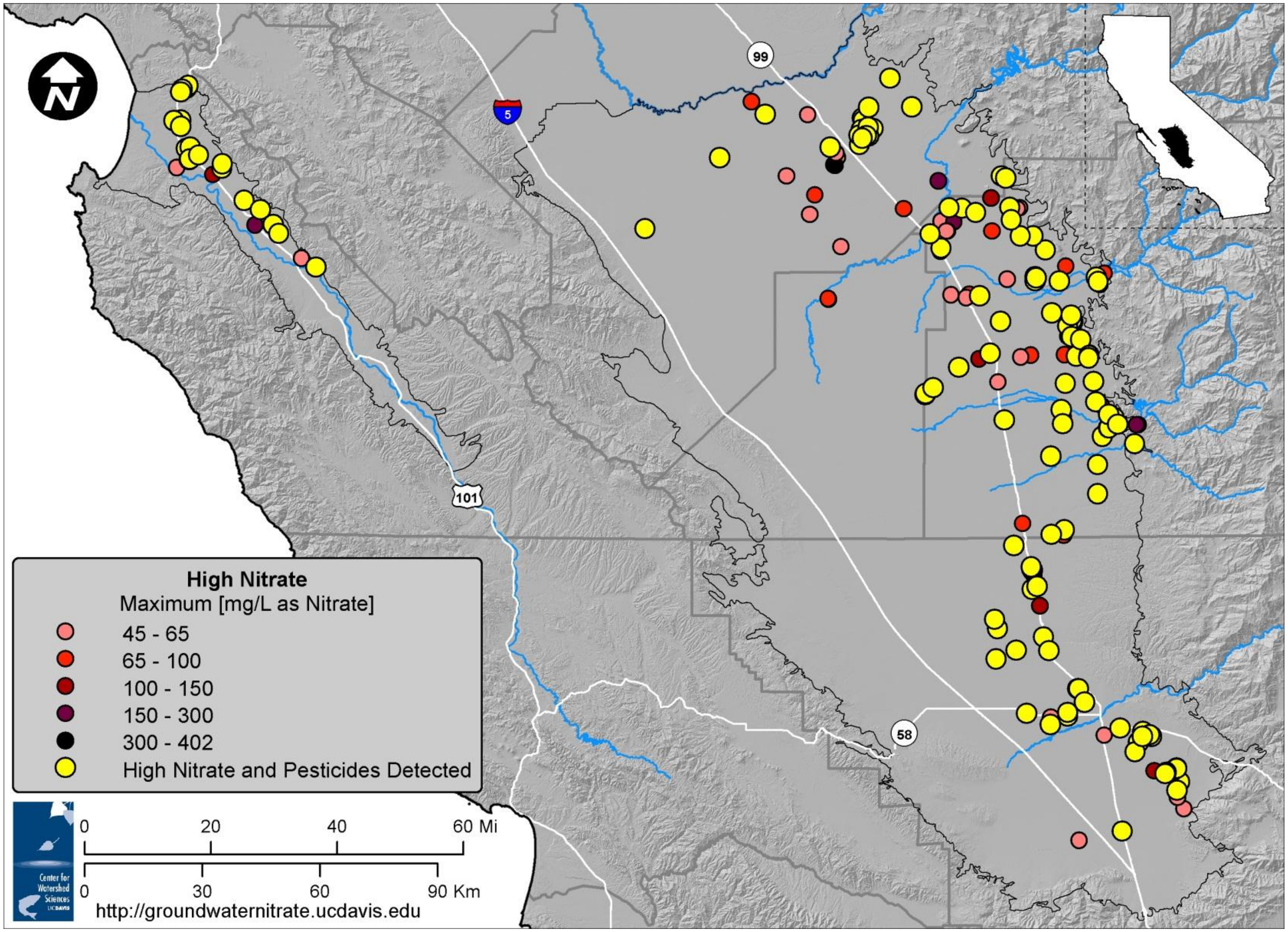

Figure 22. Raw water high nitrate wells with pesticides detected. (Source: CDPH PICME and WQM Databases.)

Through the investigation of local water quality data, the relationship between well depth and the incidence of nitrate and arsenic emerged as a potential concern for water systems drilling deeper wells to reach groundwater with lower nitrate levels. The incidence of nitrate impacted groundwater was suspected to decrease with well depth while the incidence of arsenic impacted groundwater was suspected to increase with well depth. Further analysis of this option requires that well screen depth is known in addition to well water quality. Unfortunately, for most wells with water quality information, depth information is not available (See Technical Report 4 (Boyle et al. 2012) for additional information on wells data). All wells having depth information and arsenic testing data were included, as were all wells having depth information and nitrate testing data (CDPH PICME [Permits Inspection Compliance Monitoring and Enforcement] and WQM [Water Quality Monitoring] databases). The available dataset leads to a potential bias as it excludes wells with testing data for which depth information was unavailable. For the examination of nitrate and depth, the depth to the top of the screened interval was used; depth categories are based on the minimum screened depth of the well. For the examination of arsenic and depth, well depth was calculated as the sum of the depth to the top of the screen and the length of the screen; depth categories are based on the maximum screened depth of the well. Nitrate and arsenic levels were averaged for each well across all available tests from 2006 – 2010 to avoid bias caused by having many samples for some wells and only few samples for others. Constituent concentrations were subsequently averaged across all wells in each depth category.

No relationship was found in the Salinas Valley (Figure 23); however this may be due, in part, to the more limited sample size of Salinas Valley wells (192 wells for nitrate analysis, 142 wells for arsenic analysis) in comparison with the sample size of Tulare Lake Basin wells (826 wells for nitrate analysis, 741 wells for arsenic analysis). In the Tulare Lake Basin, the variation between individual wells within each depth category and across depth categories leads to inconclusive results lacking statistical significance, despite the suggestion of the expected trend (Figure 24). However, in the context of the nitrate and arsenic MCLs, results in the Tulare Lake Basin suggest an increase in the incidence of arsenic MCL exceedance with well depth and a decrease in the incidence of nitrate MCL exceedance with well depth (Figure 25). Additional data are necessary for definitive confirmation of this trend and local conditions can vary significantly as water quality varies substantially with well location, well design, and subsurface geology.

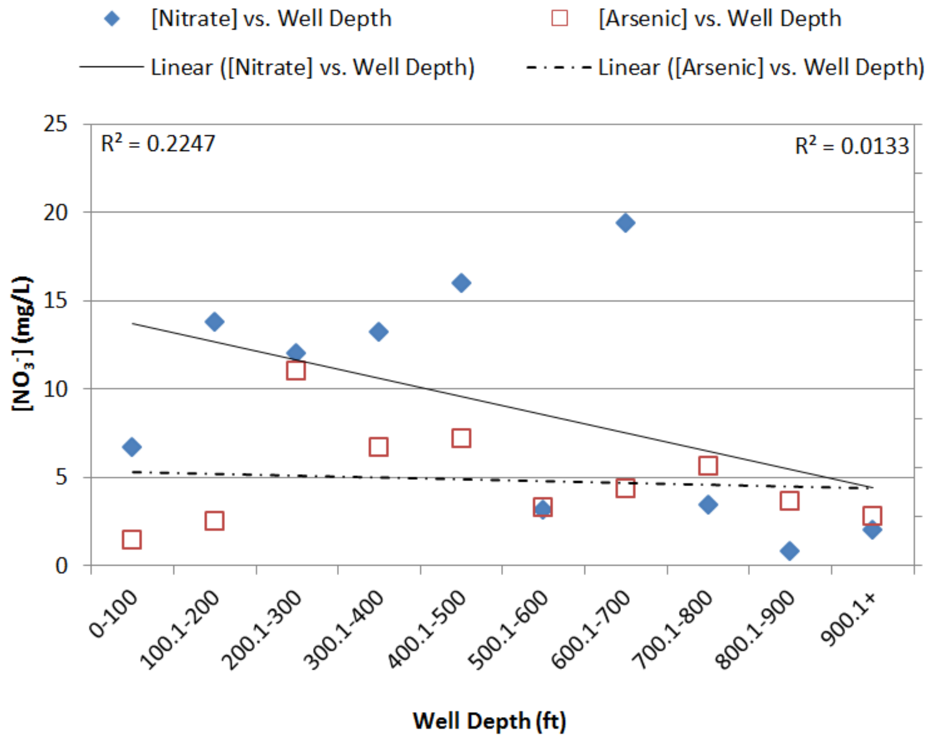

Figure 23. Salinas Valley [As] versus total well depth (deepest water) and [NO_3^-] versus depth to top of screen (shallowest water). [1 ft=0.30 meters]

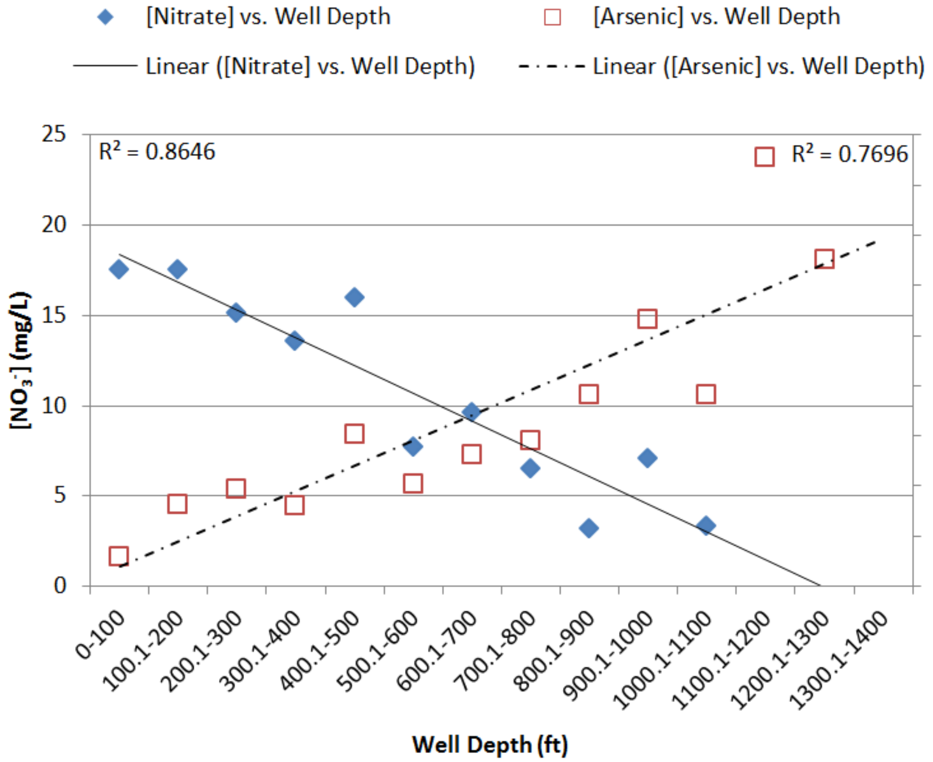

Figure 24. Tulare Lake Basin [As] versus total well depth (deepest water) and [NO_3^-] versus depth to top of screen (shallowest water). [1 ft=0.30 meters]

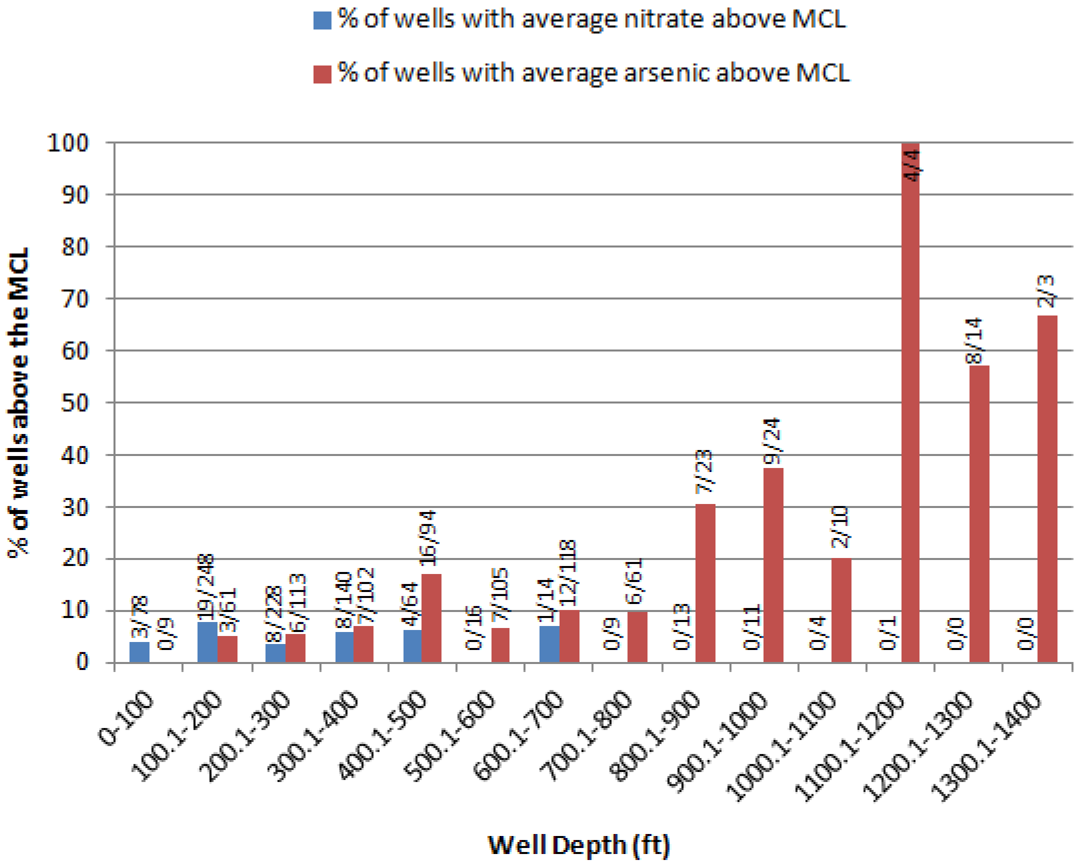

Figure 25. Tulare Lake Basin: Incidence of nitrate and arsenic MCL exceedance with well depth. [1 ft=0.30 meters]

4.3 Water Quality and Treatment Selection

Within the drinking water community, the treatment options typically considered to address nitrate contamination are IX and RO. Other technologies are available or emerging (EDR, BD, CD) because, under some circumstances, the alternatives offer advantages that IX and RO cannot. New technologies will continue to be investigated and developed because no single option is ideal for all situations. There is not a nitrate treatment option currently available that can address all possible scenarios at a lower cost than all other options. The following diagram is a rough guide for treatment technology selection based on water quality concerns and possible priorities for a given water source or system (Table 18). This diagram includes generalizations and is not intended to be definitive. In the selection of nitrate treatment technologies the unique needs of an individual water system must be assessed by professional engineers to optimize treatment selection and design.

As Table 18 shows, the most appropriate method to address nitrate contamination can be influenced by influent nitrate concentrations as well as other water quality parameters. Nitrate levels well above the

MCL may lead to the selection of one treatment option while nitrate levels just above the MCL may be more cost-effectively addressed with a different treatment option. Table 19 lists several scenarios as an example of appropriate options based on influent nitrate level and water system characteristics.

Table 18. Comparison of major treatment types.[1]

Concerns	IX	RO	EDR	BD	CD
High Nitrate Removal	🟨	🟨	🟨	🟩	
High TDS Removal	🟥	🟩	🟩	🟥	🟥
Arsenic Removal	🟩	🟩	🟩	🟨	🟩
Radium and Uranium Removal	🟩	🟩	🟩		
Chromium Removal	🟩	🟩	🟩	🟩	🟩
Perchlorate Removal	🟩	🟩	🟩	🟩	

Priorities	IX	RO	EDR	BD	CD
High Hardness Not a Major Concern	🟥	🟥	🟨	🟩	
Reliability	🟩	🟩	🟩	🟨	🟥
Training/ Ease of operation	🟩	🟩	🟩	🟨	
Minimize Capital Cost	🟩	🟨	🟨	🟨	
Minimize Ongoing O&M Cost	🟩	🟨	🟨	🟩	
Minimize Footprint	🟩	🟩	🟩	🟨	
Industry Experience	🟩	🟩	🟨	🟥	🟥
Ease of Waste Management	🟥	🟥	🟨	🟩	🟩

🟩	🟨	🟥	
Good	→	Poor	Unknown (blank)

[1] Ion Exchange (IX), Reverse Osmosis (RO), Electrodialysis Reversal (EDR), Biological Denitrification (BD), Chemical Denitrification (CD). This table offers a generalized comparison and is not intended to be definitive. There are notable exceptions to the above classifications.

Table 19. Influence of nitrate concentration on treatment selection.[1]

Option	Practical Nitrate Range	Considerations
Blend	10 – 30% above MCL	Dependent on capacity and nitrate level of blending sources.
IX	Up to 2X MCL	Dependent on regeneration efficiency, costs of disposal, and salt usage. Brine treatment, reuse, and recycle can improve feasibility at even higher nitrate levels.
RO	Up to many X the MCL	Dependent on availability of waste discharge options, energy use for pumping, and number of stages. May be more cost-effective than IX for addressing very high nitrate levels.
BD	Up to many X the MCL	Dependent on the supply of electron donor and optimal conditions for denitrifiers. Ability to operate in a start-stop mode has not yet been demonstrated in full-scale application; difficult to implement for single well systems. May be more cost-effective than IX for addressing high nitrate levels.

[1] Based on contact with vendors and environmental engineering consultants.

5 Addressing Nitrate Impacted Potable Water Sources in California

In the Tulare Lake Basin and Salinas Valley, current and historical methods to address nitrate contamination of potable water supplies include well abandonment, destruction, and inactivation; blending; and treatment. The incidence of well abandonment, destruction, or inactivation in California was explored through analysis of the CDPH PICME and WQM databases. Through collaboration with CDPH, systems treating or blending to address nitrate contamination were identified and a survey was conducted to collect additional information, including the costs of treatment.

5.1 Well Abandonment, Destruction, and Inactivation

If alternative source wells are available, costly treatment is often avoided through abandonment or inactivation of wells. However, wells must be properly destroyed or abandoned, in accordance with local requirements, to avoid hazardous conditions and the potential for groundwater contamination. The cost of proper well destruction and abandonment varies with numerous factors including depth, subsurface conditions, well type, and well construction. The minimum cost to properly destroy a 300 – 400 ft well is ~$15,000; use of best practices would increase cost (Aegis Groundwater Consulting 2011).[20]

To assess the incidence of well abandonment and inactivation due to nitrate contamination, an analysis of the CDPH PICME and WQM databases was performed. Nitrate records from 2006 – 2010 for wells labeled as abandoned, destroyed, or inactive were examined. Wells with at least one nitrate test above the MCL and wells including “NO3” or “Nitrate” in the well description were flagged. Table 20 lists the resulting number of nitrate impacted wells abandoned, destroyed, or inactive in the study area and also across California; locations are mapped in Figure 26. There is evidence of mislabeling in the PICME/WQM database. Wells missing from this analysis which may have been abandoned, destroyed, or inactivated due to nitrate may have records that are not up to date or may be mislabeled. This analysis utilizes exceedance of the nitrate MCL as an indicator of the reason for well status change; however, a portion of these wells may have been abandoned, destroyed, or inactivated for reasons other than nitrate contamination. The purpose of this analysis was to assess the incidence of well abandonment, destruction, and inactivation due to nitrate contamination. However, the small number of wells identified as abandoned, destroyed, or inactivated due to nitrate, relative to the total number of wells in these categories (which were abandoned, destroyed, or inactivated for any number of reasons) leads to two possible conclusions: the reason for well status change is not consistently identified in the CDPH database or there are simply very few wells in these categories. However, comparison of the frequency of abandonment, destruction, and inactivation of wells due to nitrate within the study area

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[20] Using best practices to properly destroy wells, “all wells would be perforated top to bottom, a high-grade cement-sand concrete would be used, and the concrete would be pressure grouted into the formation and then allowed to fill the well” (Aegis Groundwater Consulting 2011). Use of best practices for well destruction would increase the cost above the $15,000 minimum.

with those across California indicates that the study area accounts for ~22% of such wells across the state; a disproportionate number as the total number of wells in these categories accounts for only ~13% of the CA total. Analysis of the frequency of abandoned and destroyed wells and their relevance to nitrogen loading is discussed in Technical Report 2, Section 9, including agricultural wells (Viers et al. 2012).

Table 20. Incidence of abandonment, destruction, and inactivation of nitrate impacted drinking water wells.[1]

	Nitrate Impacted Wells				Total Wells	
	TLB	**SV**	**Study Area Total**	**CA**	**Study Area Total**	**CA**
Destroyed	1	0	1	9	217	2,315
Abandoned	2	1	3	28	494	2,584
Inactive	33	2	35	138	1,001	8,253
Total	***36***	***3***	***39*** ***22% of CA total***	***175***	***1,712*** ***13% of CA total***	***13,152***

[1] Source: CDPH PICME and WQM databases. There is evidence of mislabeling in the PICME/WQM database. Wells missing from this analysis which may have been abandoned, destroyed, or inactivated due to nitrate may have records that are not up to date or may be mislabeled. This analysis utilizes exceedance of the nitrate MCL as an indicator of the reason for well status change; however, a portion of these wells may have been abandoned, destroyed, or inactivated for reasons other than nitrate contamination.

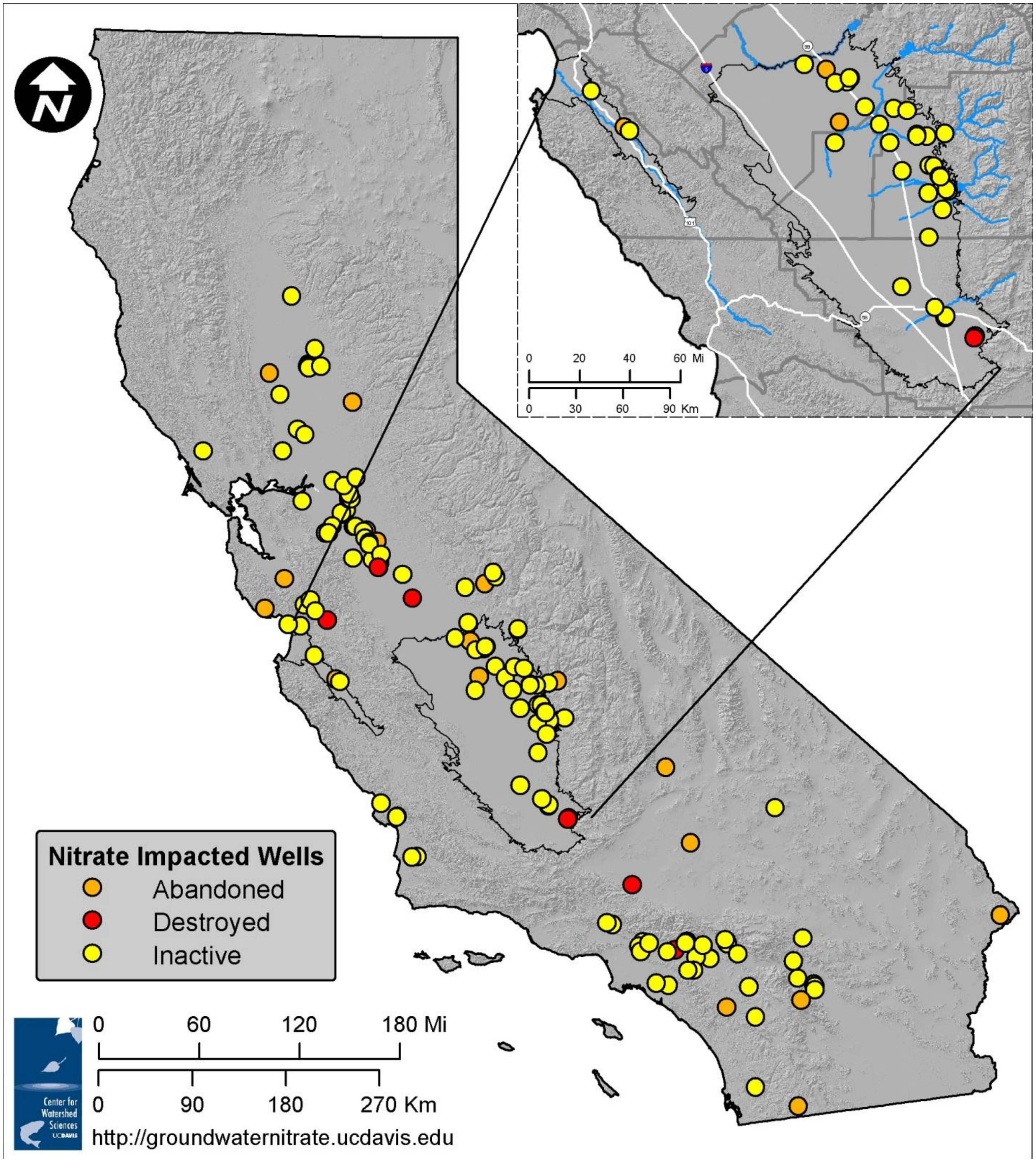
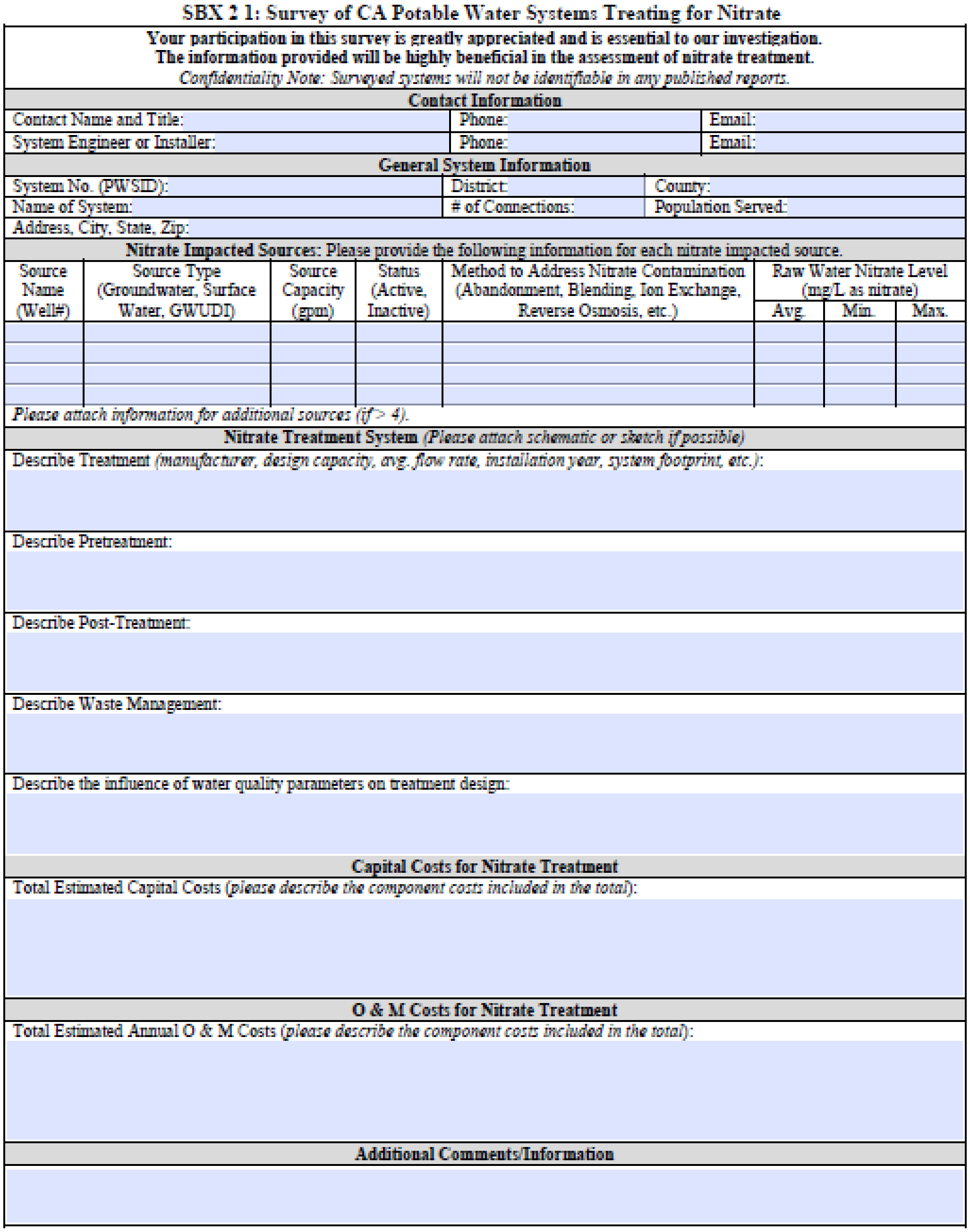


Figure 26. Location of nitrate impacted abandoned, destroyed, and inactivated wells. (Source: CDPH PICME and WQM Databases.)

5.2 Survey of Blending and Treating Systems

Complementing the detailed cases studies for each of the treatment types (above), a survey was conducted to assess full-scale application of nitrate treatment in California and specifically in the Tulare Lake Basin and Salinas Valley. The survey (Figure 27) was designed to gather detailed information on treatment type, water quality parameters affecting treatment, details of the treatment system, and cost information. Systems were contacted via phone and email and the survey packet was emailed whenever possible for ease of response. The survey packet included a letter of introduction, a brief project description, and the digital survey. Whenever possible, systems were contacted via phone and email for clarification of submitted responses and to gather additional information.

The list of water systems treating and/or blending to address nitrate contamination was developed with assistance from CDPH and analysis of the CDPH PICME and WQM databases. CDPH compiled a list of systems across CA treating and/or blending for nitrate after completing an internal review to ensure the provision of the most comprehensive list possible. Analysis of systems listed in the PICME and WQM databases confirmed treating systems based on nitrate levels and descriptions of individual systems and sources. County regulated systems treating and/or blending for nitrate were subsequently determined through contact with the individual county health departments and added to the list. The survey focus was treating systems, but several blending systems were also included.



SBX 2 1: Survey of CA Potable Water Systems Treating for Nitrate

Your participation in this survey is greatly appreciated and is essential to our investigation.
The information provided will be highly beneficial in the assessment of nitrate treatment.
Confidentiality Note: Surveyed systems will not be identifiable in any published reports.

Contact Information

Contact Name and Title:	Phone:	Email:
System Engineer or Installer:	Phone:	Email:

General System Information

System No. (PWSID):	District:	County:
Name of System:	# of Connections:	Population Served:
Address, City, State, Zip:		

Nitrate Impacted Sources: Please provide the following information for each nitrate impacted source.

Source Name (Well#)	Source Type (Groundwater, Surface Water, GWUDI)	Source Capacity (gpm)	Status (Active, Inactive)	Method to Address Nitrate Contamination (Abandonment, Blending, Ion Exchange, Reverse Osmosis, etc.)	Raw Water Nitrate Level (mg/L as nitrate)		
					Avg.	Min.	Max.

Please attach information for additional sources (if > 4).

Nitrate Treatment System *(Please attach schematic or sketch if possible)*

Describe Treatment *(manufacturer, design capacity, avg. flow rate, installation year, system footprint, etc.)*:

Describe Pretreatment:

Describe Post-Treatment:

Describe Waste Management:

Describe the influence of water quality parameters on treatment design:

Capital Costs for Nitrate Treatment

Total Estimated Capital Costs (*please describe the component costs included in the total*):

O & M Costs for Nitrate Treatment

Total Estimated Annual O & M Costs (*please describe the component costs included in the total*):

Additional Comments/Information

Figure 27. Digital survey distributed to drinking water systems treating and/or blending to address high nitrate levels.

Of the 42 systems identified as treating for nitrate throughout CA, 26 systems completed the survey. Statewide systems are mapped in Figure 28 and systems in the study area of interest are mapped in Figure 29. Whenever possible, systems are blending to address the nitrate problem, accounting for ~56% of the statewide systems in Figure 28.

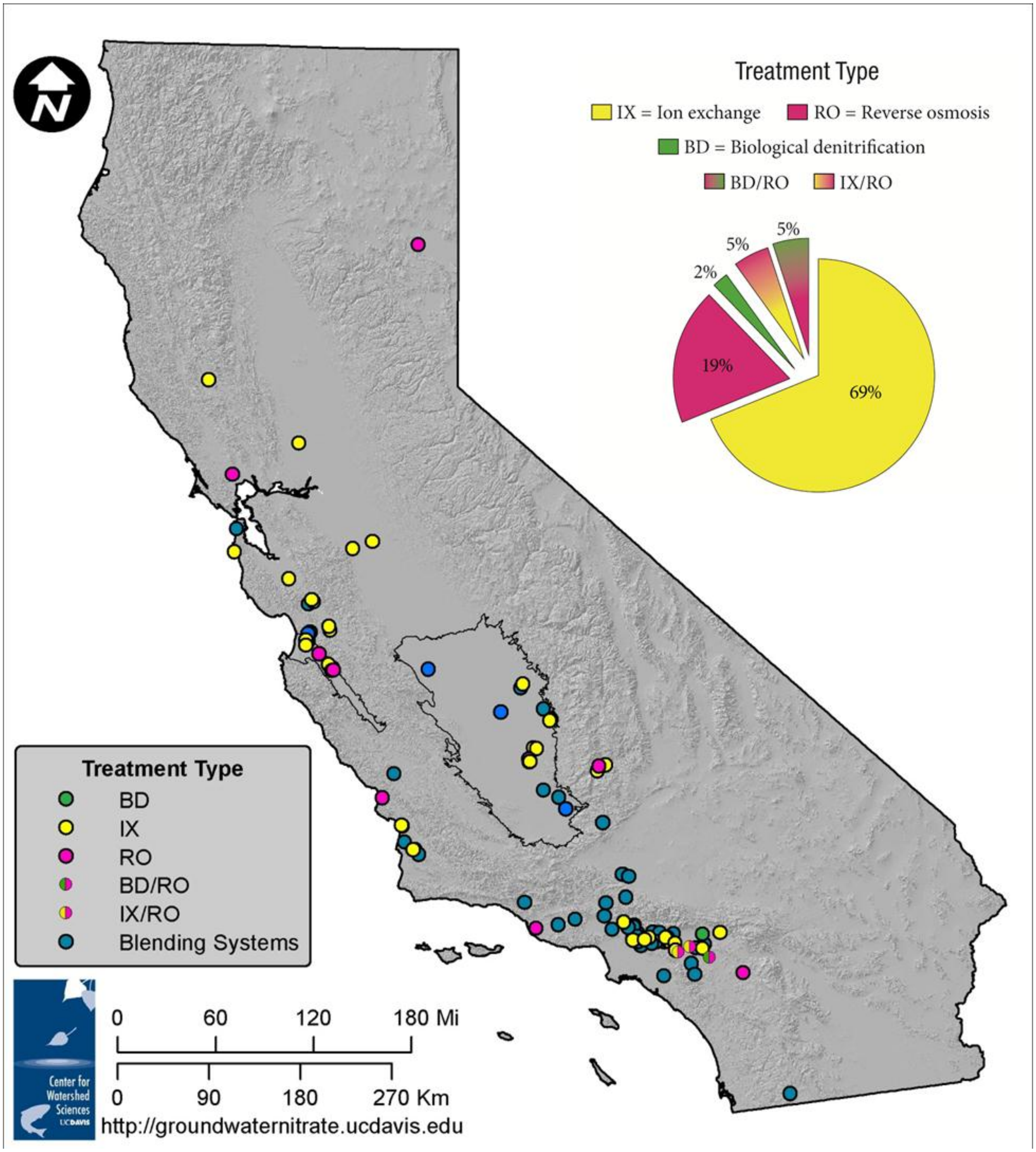


Figure 28. California drinking water systems treating and/or blending for nitrate. (Source: CDPH Internal Review of facilities and contact with facility and county representatives.)

Approximately 70% of treating systems across CA are using IX and ~20% are using RO. Several locations have implemented both RO and IX, primarily to address salinity as well as nitrate. Biological treatment is being implemented at two locations in CA. After successful completion of a one-year demonstration study, a system in Rialto, CA, is installing a biological treatment system primarily to address perchlorate contamination of drinking water. The water system also has a history of nitrate contamination and the biological treatment system provides the potential to treat their high nitrate source(s) as well. Construction of the full-scale biological treatment system is underway. In Riverside, CA, biological treatment has been investigated for the treatment of the RO bypass stream to increase total plant capacity. See Section 3.4.5 Biological Denitrification - Case Studies, for detailed case studies of these unique systems.

Focusing on the Tulare Lake Basin and the Salinas Valley (Figure 29), 23 systems were found to be treating and/or blending to address the nitrate problem (10 blending systems, 10 IX systems, and 3 RO systems).

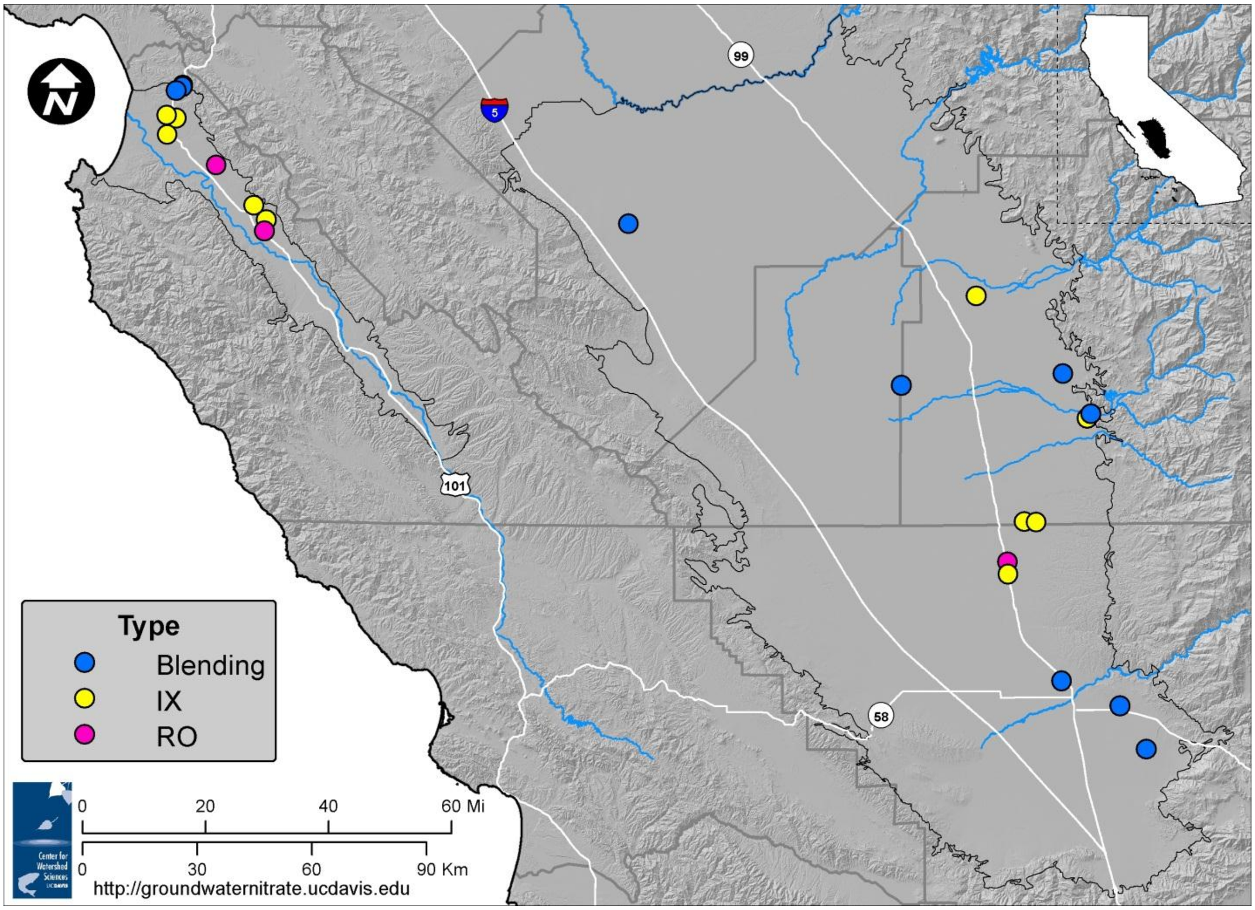


Figure 29. Drinking water systems treating or blending for nitrate in the Tulare Lake Basin and Salinas Valley. (Source: CDPH Internal Review of facilities and contact with facility and county representatives.)

Table 21 lists the population ranges and nitrate levels of blending, IX, and RO systems in the study area. The IX systems cover the widest population range; however, it is important to note that some large systems using IX for nitrate treatment also use blending. For each system the minimum, maximum, and average nitrate concentration across all active wells were determined, then the average of each of those categories across all systems for each of the treatment options was calculated to illustrate the typical maximum, minimum, and average nitrate levels for each treatment type. Of these systems, the average maximum nitrate level of blending systems is slightly lower than that of treating systems; with lower nitrate levels and access to a low-nitrate source, the possibility of blending can avoid the need for more costly treatment. Sixty percent of the IX and RO systems in Figure 29 have only a single active well and 40% of the blending systems have only two active wells. Water quality changes and increasing nitrate levels could be particularly problematic for these one- and two-well systems. Even where blending is successfully meeting today's needs, it may be precarious to assume or expect that water systems can rely solely on blending for compliance into the future.

Table 21. Population and nitrate levels of systems in the study area treating and/or blending for nitrate.

		Average Raw Nitrate (mg/L as nitrate)		
	Population Range (Total)	**Max.**	**Min.**	**Avg.**
Ion Exchange	25 – 133,750 (261,200)	71	15	40
Reverse Osmosis	45 – 6,585 (6,760)	75	24	41
Blending	45 – 25,500 (83,475)	64	3	32

An example of a blending system in Tulare County (Figure 30) illustrates the complexity of even a simple blending system. This system has seven wells, two of which are high in nitrate (wells 8 and 11 in red). Most of the year the high nitrate wells are inactive, but with high demand in summer, the system blends a high nitrate source with other wells. Table 22 lists nitrate levels, depth, and capacity of the system's source wells. It is interesting to note that the high nitrate wells have the highest capacity and are actually some of the deeper wells. Increasing nitrate levels in the low nitrate wells would be cause for concern as the system's blending program would be affected. This is one simple example of hundreds of scenarios. Even with this simple blending system, there are several complicating factors including differences in capacity, seasonal variation, and the variability of nitrate levels in wells that are very close together. Extrapolating this concept over the entire study area, the case by case nature of addressing the nitrate problem becomes more apparent.

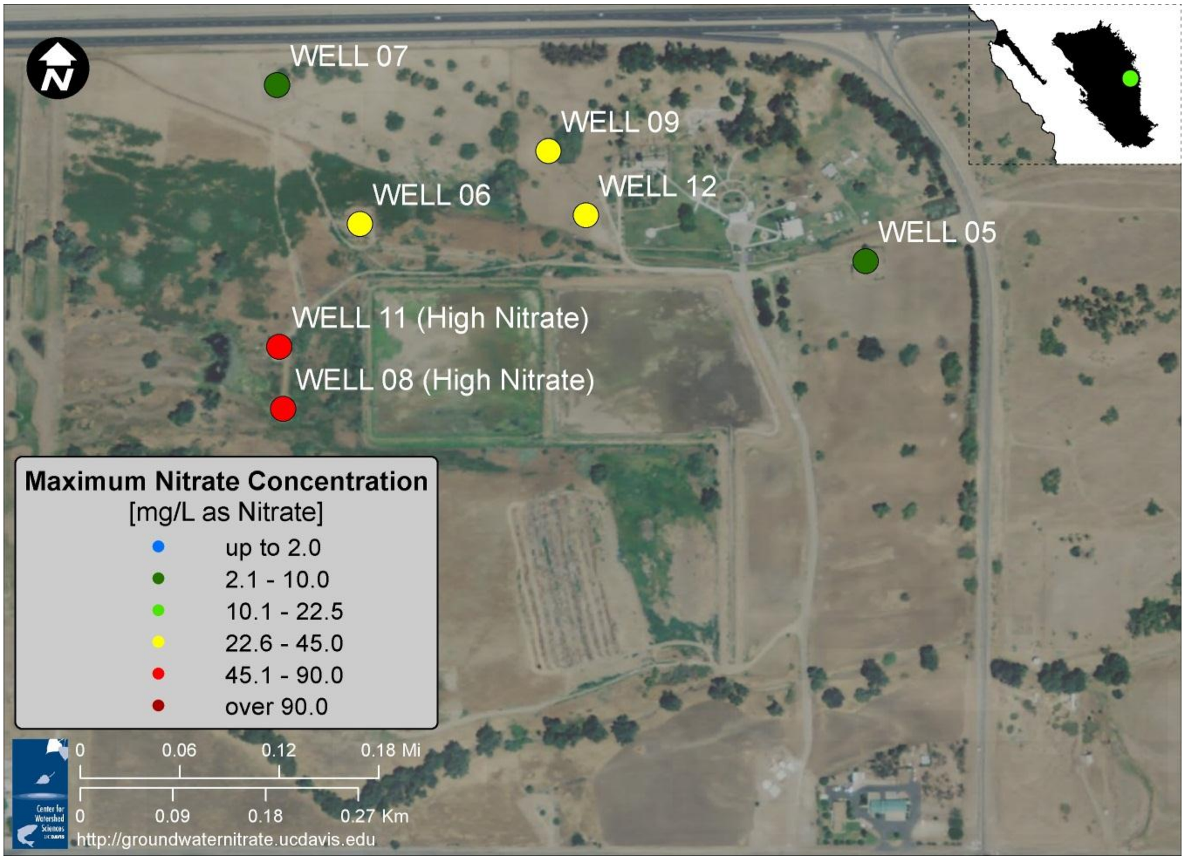


Figure 30. Wells of a blending system in Tulare County. (Source: Contact with facility/survey.)

Table 22. Nitrate level, well depth and well capacity for a Tulare County blending system.

Well #	Max Nitrate (mg/L as nitrate)	Total Depth (ft)	Depth to Top of Screen (ft)	Capacity (gpm)
05	10	255	163	153
06	12	328	30	174
07	7.8	296	94	161
08	**78**	**393**	**250**	**378**
09	11	398	160	150
11	**81**	**400**	**340**	**318**
12	11.8	470	180	170

Case studies of nitrate treatment systems are included above in the various treatment technology sections.

6 Treatment Cost Analysis

In the estimation of treatment costs, there are two major categories to consider: capital costs and O&M costs. Capital costs refer to the upfront investment required for the design, implementation, and installation of the treatment system. O&M costs refer to the annual costs for operating and maintaining the system. Based on U.S. EPA cost estimating procedures, developed through the Technology Design Panel, capital costs can be further categorized as follows (U.S. EPA 2000; U.S. EPA 2005):

Process Costs
- Manufactured Equipment
- Concrete and Steel
- Electrical & Instrumentation
- Pipes & Valves

Engineering Costs
- Contractor Overhead and Profit
- Engineering Fees
- Legal, Fiscal, and Administrative Fees

Construction Costs
- Sitework & Excavation
- Subsurface Considerations
- Standby Power
- Contingencies
- Interest During Construction

Indirect Costs
- Housing
- Permitting
- Land
- Training
- Piloting
- Public Education

The cost analysis detailed below was performed in accordance with U.S. EPA cost estimation procedures (U.S. EPA 2000). Total capital costs were converted to annualized capital costs ($/kgal) based on Eqn. 21.

$$\text{Annualized Capital Cost} = [\text{Capital Cost (\$)} * \text{Amortization Factor}]$$

$$/ \ [\text{Flow (MGD)} * 1000 \text{ kgal/mgal} * 365 \text{ days/year}] \qquad \text{(Eqn. 21)}$$

An amortization value of 0.0802 was used which corresponds with an interest rate of 5% over 20 years (Eqn. 22).

$$\text{Amortization Factor} = (1+i)^N/((1+i)^N - 1)/i) \qquad \text{(Eqn. 22)}$$

Where i = interest rate and N = number of years

Annual O&M costs were calculated based on Eqn. 23 to convert total annual O&M costs to $/kgal.

$$\text{O\&M Cost (\$/kgal)} = [\text{O\&M Cost (\$)}]$$

$$/ \ [\text{Flow (MGD)} * 1000 \text{ kgal/mgal} * 365 \text{ days/year}] \qquad \text{(Eqn. 23)}$$

Annualized Capital Cost ($/kgal) and O&M Cost ($/kgal) were summed to determine Total Annualized Cost ($/kgal).

In the U.S. EPA 1979 Cost Estimating Manual (U.S. EPA 1979), cost information is included for anion exchange for nitrate removal; however, for a more recent set of cost indices, cost information reported for anion exchange and membrane processes for arsenic treatment will be used here for comparison with collected cost information (U.S. EPA 2000). Similarly, in a recent AWWA publication examining the national impact of changing the nitrate MCL, cost curves of anion exchange for arsenic removal were used in the estimation of nitrate treatment costs. "USEPA's cost curves were chosen because they are generally used for developing national compliance costs and because arsenic and nitrate use comparable regenerable IX treatment" (Weir & Roberson 2011, p. 49). Based on the U.S. EPA cost curves of IX for arsenic removal, costs by system size using regenerable IX are listed in Table 23, ranging from $0.22/kgal for a 10 MGD system to $4.60/kgal for a 0.01 MGD system. Disposal costs were not included in the U.S. EPA cost estimates of IX for arsenic removal.

Table 23. Cost estimation using U.S. EPA cost curves of IX for arsenic removal.[1]

System Capacity	Annualized Capital Cost ($/kgal)	O&M Cost ($/kgal)	Total Annualized Cost ($/kgal)
10	0.09	0.13	0.22
2	0.13	0.23	0.36
1	0.26	0.46	0.72
0.1	0.21	0.70	0.91
0.01	0.79	3.81	4.60

[1] U.S. EPA 2000.

For the cost analysis detailed herein, treatment cost information was collected from literature, vendors, surveys, and contact with existing systems. Factors affecting system cost include facility size (flow rate), source water quality (including nitrate concentration), environmental factors (temperature), and target effluent nitrate concentration. Disposal of waste brine can be a significant portion of O&M costs for the removal technologies (see Section 6.4 Disposal Costs for a more detailed discussion of disposal costs). Capital costs for treatment can include land, housing, piping, storage tanks, O&M equipment, process equipment (i.e., vessels, resin, membranes, media, etc.), preliminary testing (pilot studies), permits, and training. O&M costs for treatment can include resin, media, or membrane replacement (due to loss or degradation) and disposal; waste residuals disposal or treatment (e.g., brine disposal); chemical use (salt, anti-scalant, pH adjustment); repair and maintenance; power; and labor. Costs can be difficult to assess due to inconsistencies in how cost information is reported. Comparison of costs across different systems is not always valid due to differences in influent water quality parameters, system size, waste management options, and system configuration. Published costs do not always include comparable information. The cost information listed in this section is provided as an approximate range of costs. Costs for implementing treatment may be very different from those listed here. A thorough cost analysis of design parameters for specific locations would be required for accurate cost estimation. The information gathered through the survey includes reported costs associated with treating systems in CA.

Assumptions and sources of uncertainty in this analysis of treatment costs include the following:

- For certain sources of cost information it is unclear if all components are included in capital costs (e.g., preliminary planning, pilot testing, installation, administration fees, engineering fees, building cost, storage, etc.) and O&M costs (e.g., pumping, disposal, labor, energy, chemicals, etc.). Whenever possible efforts have been made to ensure inclusion of all relevant costs.

- Many treatment systems blend treated water with a bypass stream; whenever possible costs were calculated on the basis of total produced water to accommodate the blending configuration. For example, an RO system may remove nitrate to very low levels, then blend the permeate with a bypass stream, raising nitrate levels to the distribution goal (and restoring other ions).
- Capital costs generally increase with design capacity, but some systems “over design” with a design capacity significantly greater than the actual flow. The calculation of annualized capital costs are based on average flow rather than design capacity whenever possible to provide capital costs per unit of produced water.
- O&M costs are based on actual average flow, rather than design capacity, whenever possible.
- Costs were adjusted to 2010 dollars.
- Costs of drilling a new well were excluded from the treatment cost analysis to ensure appropriate comparison.
- Costs of systems in the design phase are anticipated costs.
- Several sample costs of electrodialysis and biological treatment systems designed for the removal of other constituents were included; based on communication with water treatment engineers and vendors, the costs for treatment for nitrate removal should be similar.
- Costs were collected for systems with wide-ranging characteristics including variation in system size, nitrate levels, co-contaminants, and other water quality constituents.
- Several systems reported renting treatment equipment and/or contracting O&M services, resulting in very different capital and O&M costs.
- Given only equipment costs (e.g., from vendors), total capital costs were modeled based on U.S. EPA scaling factors (U.S. EPA 2000).

6.1 Costs by Treatment Type

Comparison of the average total annualized cost for IX, RO, EDR, and BD across all system sizes highlights RO as the most expensive option (Figure 31). EDR costs are for a limited number of systems including costs for the treatment of constituents other than nitrate and may not be representative of actual EDR costs for nitrate removal. Based on preliminary estimates, biological treatment has the potential to be cost competitive. Costs for IX, RO, and BD are broken down into three system size categories to illustrate the variability in cost with system size (Figure 32, Figure 33, and Figure 34, respectively). Note the much higher cost for system sizes less than 0.5 MGD for IX (Figure 32) and RO (Figure 33). Preliminary estimates of BD treatment costs do not illustrate the same degree of variability with system size (Figure 34); however, BD for nitrate removal from drinking water is an emerging technology and available cost information is limited. The very high O&M costs for small systems (Figure 32 for IX and Figure 33 for RO) are representative of low flow systems making the cost per 1000 gallons quite high. This highlights a problem faced by many small systems lacking the benefits of economies of scale; funding may be available for the initial upfront investment, but with high O&M costs long term treatment can become unsustainable. Additionally, due to insufficient funds for ongoing costs, small water systems can be faced with an inability to retain qualified operators which can lead to MCL violations and insufficient maintenance of the treatment system. (ED/EDR costs are excluded from this comparison due to insufficient cost information.)

The variability in cost information reported here is due to many factors, including variability in water quality parameters, site considerations, and the sources of uncertainty in the cost information, as discussed above. For example, one very small IX system reported the disposal of waste brine to septic, a low cost option, resulting in significantly lower O&M costs in comparison with other systems.

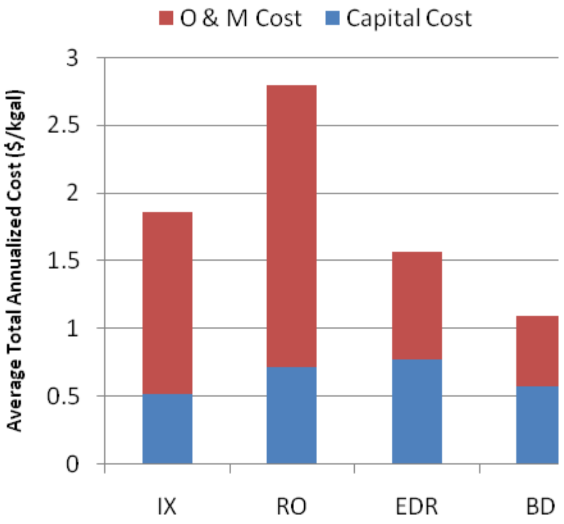


Figure 31. Average cost comparison of nitrate treatment technologies.

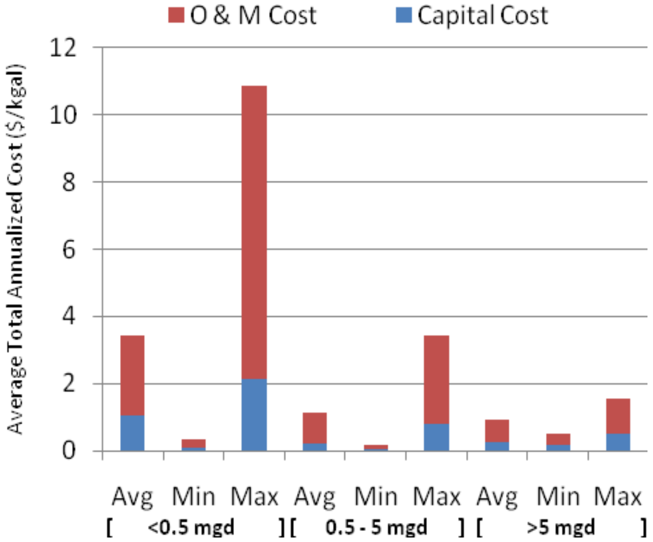


Figure 32. Costs of anion exchange for nitrate treatment.

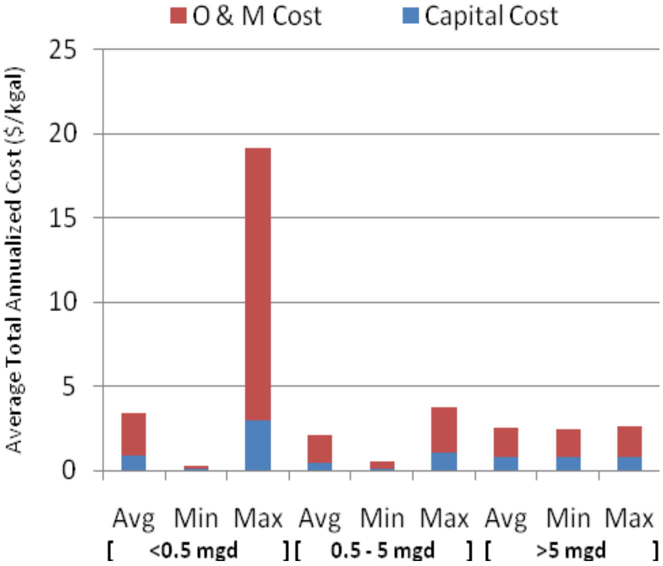


Figure 33. Costs of reverse osmosis for nitrate treatment.

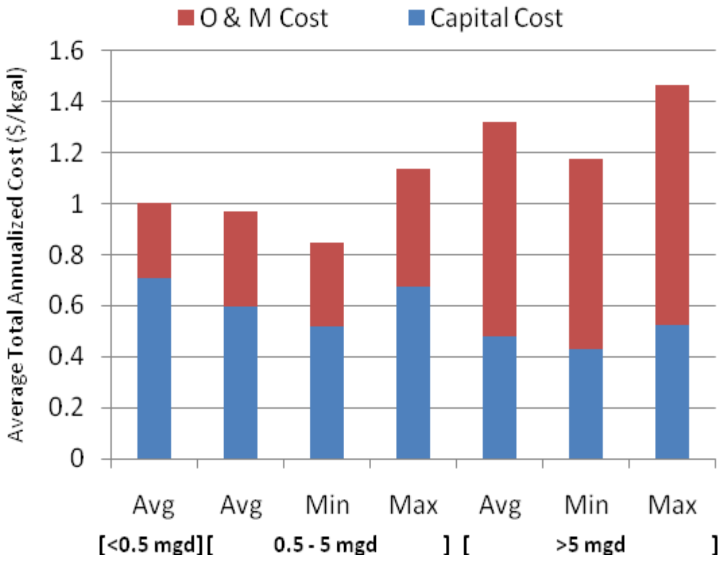


Figure 34. Costs of biological denitrification in drinking water treatment.

6.2 Costs by System Size

As indicated above, system size is a major factor in determining nitrate treatment costs. Larger treatment systems will have higher total capital and O&M costs; however, the cost per unit of produced water generally decreases as system size increases. Large treatment systems have the advantage of economies of scale. Based on cost information collected from vendors, literature, surveys, and treatment systems, treatment costs relative to system size are illustrated in the below cost curves for IX and RO (Figure 35). The development of cost curves for the other technologies was not possible due to insufficient cost information. The higher relative cost of treatment for smaller systems can be seen moving toward the vertical axis, with decreasing system size and increasing cost as the curve sweeps upward. Again, the total cost for RO treatment is higher than that of IX.

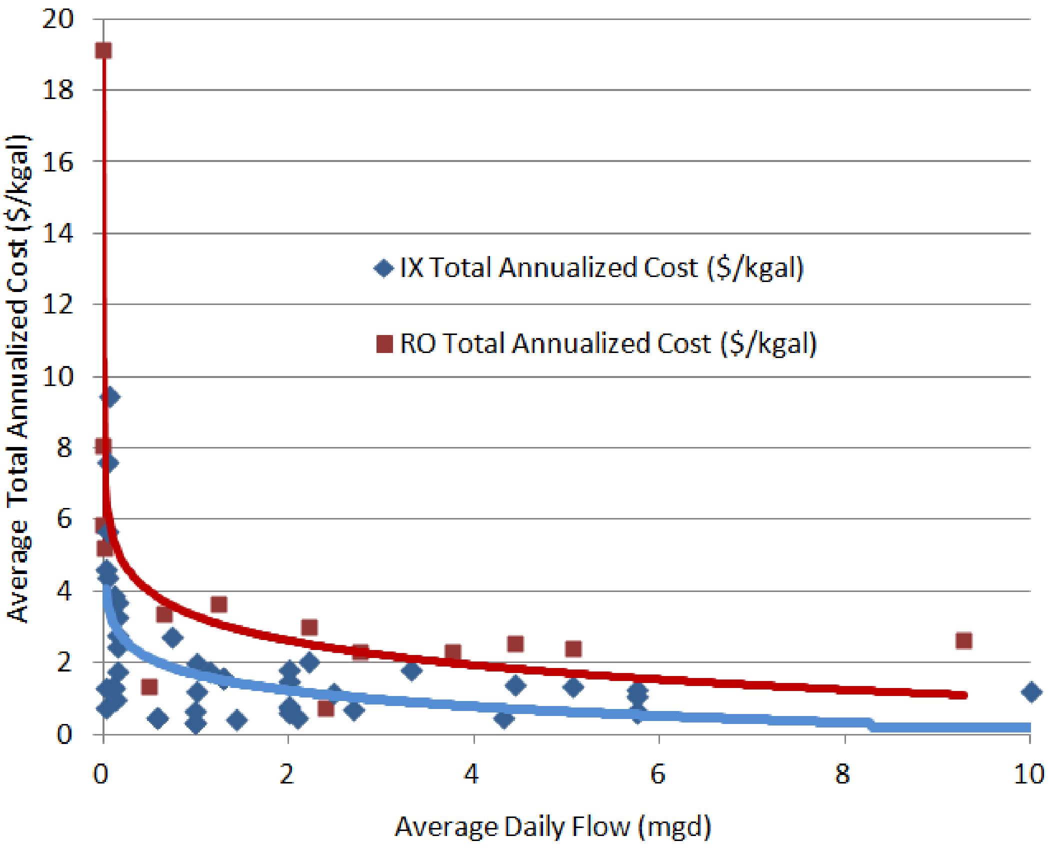


Figure 35. Cost curve of IX (blue) and RO (red) for nitrate removal.

Table 24 includes all of the most reliable treatment cost information collected for comparison of cost ranges across system size categories for IX and RO.

Table 24. Summary of anion exchange and reverse osmosis cost information by system size.

System Size (people)	Design Flow Range (typical average flow range) MGD	Treatment Type	Annualized Costs in $/1000 gallons		
			Capital Cost Range (Avg.) $/1000 gallons	O&M Cost Range (Avg.) $/1000 gallons	Total Combined Cost Range (Avg.) $/1000 gallons
Very Small (25 – 500)	0.009 – 0.17 (0.002 – 0.052)	Ion Exchange	0.05 – 1.53 (0.75)	0.28 – 3.81 (1.22)	0.62 – 4.60 (1.97)
		Reverse Osmosis	0.47 – 4.40 (2.43)	0.22 – 16.16 (4.22)	0.69 – 19.16 (6.64)
Small (501 – 3,300)	0.17 – 1.09 (0.052 – 0.39)	Ion Exchange	0.08 – 0.25 (0.15)	0.15 – 2.63 (0.87)	0.34 – 2.73 (1.05)
		Reverse Osmosis[1]	0.19 – 1.13 (0.47)	0.23 – 1.15 (0.57)	0.58 – 1.34 (0.93)
Medium (3,301 – 10,000)	1.09 – 3.21 (0.39 – 1.3)	Ion Exchange	0.06 – 0.52 (0.19)	0.12 – 1.69 (0.84)	0.36 – 2.04 (1.06)
		Reverse Osmosis[1]	0.44 – 0.63 (0.53)	0.91 – 2.76 (1.89)	1.35 – 3.39 (2.59)
Large (10,001 – 100,000)	3.21 – 30.45 (1.3 – 15.51)	Ion Exchange	0.09 – 0.41 (0.26)	0.13 – 1.39 (0.66)	0.22 – 1.81 (0.97)
		Reverse Osmosis	0.33 – 1.46 (0.97)	0.40 – 2.21 (1.48)	0.73 – 3.67 (2.38)

[1] Limited data set for the indicated system size and treatment type.

6.3 Costs by Water Quality Parameters

As highlighted above in the discussion of water quality (Section 4 Tulare Lake Basin and Salinas Valley - Water Quality Analysis), if treatment of multiple contaminants is necessary, treatment costs will generally increase. Similarly, the level of nitrate and water quality parameters that can interfere with treatment can increase O&M costs. Table 25 lists costs by system size with increasing nitrate levels and is intended as an example of nitrate treatment cost estimation based on nitrate concentration in source water. Table 25 is strictly an example and is not intended to be definitive, but only to suggest how treatment costs might change with increasing nitrate levels. The actual costs with increasing nitrate level are wide ranging and vary with numerous factors. The percent increase in O&M costs was modeled based on only two sets of vendor data in which estimates were provided based on given nitrate levels (low and high). Available data were specifically applicable to estimation of O&M increases as the nitrate concentration increases from ~1X the nitrate MCL to 2X the MCL. To extrapolate the exercise further, the same percent increase was used to predict the O&M increase from 2X the MCL to 3X the MCL. It is not possible to accurately estimate or generalize how these costs would translate for other IX systems as the two vendors provided cost estimates specifically for a system using a selective resin and a second unique system designed for low brine. Based on the information herein, O&M costs would be expected to increase even more using conventional IX under the given scenario of increasing nitrate levels. It is important to note that the treatment system could also be designed differently for higher nitrate levels (more or larger vessels, in series/in parallel, different bypass ratios, etc.); this is not included in the table as it would be pure speculation.

Table 25. An exercise in the estimation of treatment costs based on appropriate technology for various nitrate levels.[1]

System Size (people)	Raw Nitrate Level	Treatment Type	O&M Cost Range (Avg.)[2]	Annualized Combined Cost Range (Avg.)
			$/1000 gallons	$/1000 gallons
Very Small (25 – 500)	1X MCL	Ion Exchange	0.28 – 3.81 (1.22)	0.62 – 4.60 (1.97)
	2X MCL	Ion Exchange	0.35 – 10.48 (2.13)	0.69 – 11.27 (2.88)
	3X MCL	Ion Exchange	0.42 – 17.15 (3.05)	0.76 – 17.94 (3.80)
	3X MCL	Reverse Osmosis	0.22 – 16.16 (4.22)	0.69 – 19.16 (6.64)
Small (501 – 3,300)	1X MCL	Ion Exchange	0.15 – 2.63 (0.87)	0.34 – 2.73 (1.05)
	2X MCL	Ion Exchange	0.19 – 7.23 (1.52)	0.38 – 7.33 (1.70)
	3X MCL	Ion Exchange	0.23 – 11.84 (2.18)	0.42 – 11.94 (2.36)
	3X MCL	Reverse Osmosis[3]	0.23 – 1.15 (0.57)	0.58 – 1.34 (0.93)
Medium (3,301 – 10,000)	1X MCL	Ion Exchange	0.12 – 1.69 (0.84)	0.36 – 2.04 (1.06)
	2X MCL	Ion Exchange	0.15 – 4.65 (1.47)	0.39 – 5.00 (1.60)
	3X MCL	Ion Exchange	0.18 – 7.61 (2.10)	0.42 – 7.96 (2.32)
	3X MCL	Reverse Osmosis[3]	0.91 – 2.76 (1.89)	1.35 – 3.39 (2.59)
Large (10,001 – 100,000)	1X MCL	Ion Exchange	0.13 – 1.39 (0.66)	0.22 – 1.81 (0.97)
	2X MCL	Ion Exchange	0.16 – 3.82 (1.16)	0.25 – 4.24 (1.46)
	3X MCL	Ion Exchange	0.20 – 6.26 (1.65)	0.29 – 6.68 (1.96)
	3X MCL	Reverse Osmosis	0.40 – 2.21 (1.48)	0.73 – 3.67 (2.38)

[1] This table is strictly an example and is not intended to be definitive, but only to suggest how treatment costs might change with increasing nitrate levels. The estimated increase in O&M costs is wide ranging, 25% – 175%, and depends on many factors including water quality parameters, disposal options, resin capacity, resin type, and ion exchange system design. As nitrate levels increase, salt, disposal, and resin costs for IX will increase (O&M). Reverse osmosis costs will increase with increasing TDS, but not at the same rate, this cannot currently be estimated. Depending on other water quality parameters, the costs of IX are predicted to surpass those of RO. In the future, biological denitrification will likely be considered as an option for > 2X the nitrate MCL. Additionally, increasing the number and/or size of resin vessels to address higher nitrate levels would increase capital costs. O&M costs would still increase; in practice the system would be designed to optimize costs. O&M increases were considered here as an example. Actual costs with increasing nitrate levels for specific systems may vary significantly from listed costs and should be assessed by professional engineers.

[2] Increases in O&M are estimated from a limited dataset comprised of vendor cost estimates for IX costs with nitrate levels increasing from just above the MCL to slightly more than double the MCL. All available cost information was included in the 1X MCL scenario as a starting point, including systems with nitrate levels above 1X the MCL.

[3] Limited dataset for the indicated system size and treatment type.

6.4 Disposal Costs

Brine disposal costs for drinking water systems in CA using IX for nitrate treatment vary with several factors including proximity to a coastal brine line, waste brine volume (e.g., water efficiency), and the water quality characteristics of waste brine (e.g., salinity). The presence of contaminants other than nitrate (e.g., arsenic, selenium, uranium, chromium, and vanadium) in the waste stream can have a significant impact on brine disposal options and costs; disposal to a hazardous waste facility may be required at a greater cost. Methods for disposal of waste brine or concentrate reported in the survey of nitrate treatment systems in CA include discharge to a septic tank and leach fields, to a wastewater treatment plant via a sewer connection, to irrigation ponds (for RO concentrate), to a brine line, and to a wastewater treatment plant via trucking.

Disposal options are limited in the Central Valley of California due, in part, to the great distance to the coast. Trucking of waste brine to coastal wastewater facilities, although costly, is sometimes the chosen disposal option. Typical costs for trucking and disposal of spent IX brine at a coastal wastewater plant from the Central Valley can be around \$0.15/gallon (\$150/1000 gallons of waste brine). East Bay Municipal Utility District (EBMUD), in Oakland, CA, operates a wastewater management program for the disposal of high salinity and high nitrate wastewater.

O&M costs for the disposal of waste brine reported in the survey of nitrate treatment systems in CA range from \$0.015 to \$ 0.05/1000 gallons of treated water. Assuming a high efficiency of 99.5%, O&M disposal costs range from \$3 to \$11/1000 gallons of waste brine. This is consistent with the results of a recent research investigation comparing the life cycle costs of several nitrate treatment options. Meyer et al. (2010) discuss the costs of multiple brine disposal options including evaporation ponds, deep well injection, and sewer. Based on vendor estimates, results indicate total brine disposal costs (including capital and O&M costs) ranging from approximately \$7 to \$27/1000 gallons of waste brine disposal to evaporation ponds and approximately \$6 to \$11/1000 gallons of waste brine disposal to sewer (Table 26). (In the conversion of costs reported by Meyer et al. to the cost per 1000 gallons, an amortization value of 0.0802 was used which corresponds with an interest rate of 5% over 20 years.) However, it is important to note that the study by Meyer et al. (2010) was focused on the evaluation of nitrate treatment in Arizona; location specific characteristics are an important factor affecting disposal costs.

Table 26. Brine disposal costs.[1]

	Annualized Capital Cost	O&M Cost	Total Annualized Cost	Total Range
Average Cost by Waste Volume ($/1000 gallons)				
Evaporation Ponds	10.23	5.62	15.85	7 to 27
Solar Ponds	20.48	18.80	39.27	8 to 88
Well Injection	12.00	18.52	30.52	13 to 111
Sewer	2.40	5.51	7.91	6 to 11
Average Cost by Treated Volume ($/1000 gallons)				
Evaporation Ponds	0.046	0.015	0.061	0.03 to 0.14
Solar Ponds	0.063	0.047	0.110	0.07 to 0.20
Well Injection	0.051	0.077	0.128	0.03 to 0.33
Sewer	0.007	0.034	0.041	0.02 to 0.12

[1] Based on Meyer et al. 2010.

Costs of resin disposal can also vary with water quality parameters other than nitrate; IX resin removes not only nitrate, but other contaminants (e.g., arsenic) which can affect disposal options when resin needs to be replaced. High levels of other contaminants on the resin can require disposal at hazardous waste facilities and increase disposal costs, although the impact of co-contaminants is more significant on brine disposal costs than on resin disposal costs. Non-hazardous resin can be land-filled. Using regenerable resin, requiring replacement only once every 3 – 8 years (WA DOH 2005 and Dow 2010c), the cost of land-fill disposal of non-hazardous resin is expected to be minimal compared with the disposal of other waste residuals (waste brine/concentrate). Service contracts are available with various companies to manage resin replacement and disposal.

In the selection of the most appropriate nitrate treatment option, disposal costs are a significant factor; consideration of the pros and cons for the unique conditions of an individual water system is not always straightforward and can be heavily weighted by disposal options. Although other removal technologies (RO and ED) require concentrate disposal, because IX requires the addition of salt for resin regeneration, the waste stream consists of not only the nitrate and other ions that have been removed from the water, but also the spent brine solution used in regeneration. As nitrate levels in source water increase, IX resin will need to be regenerated more frequently, increasing salt use and brine waste volume. In contrast, although the recovery rate for RO is significantly lower than that of IX (~80% and >95%, respectively), the nitrate level that can be addressed with RO is theoretically much higher (in accordance with the membrane nitrate rejection rate). Recall that RO is used for desalination as well. As the nitrate concentration in the treatment stream increases, with appropriate pressure, the RO membrane will continue to reject nitrate, assuming membrane scaling and fouling are properly controlled. In the comparison of IX and RO as nitrate levels increase, theoretically there is a tradeoff between the O&M costs for each technology. With increasing nitrate levels, chemical use and waste volume will increase for IX while power use and membrane maintenance may increase for RO. Excluding all other water quality parameters, the nitrate level at which the cost of IX exceeds the cost of RO requires more research.

Lastly, several small water systems included in the nitrate treatment survey indicated disposal of waste concentrate to a septic system. This highlights an important tradeoff; while small water systems do not have the advantages of economies of scale, with a low volume waste stream (depending on chemical composition to avoid negatively impacting septic system function or underlying groundwater), disposal to septic can avoid other, more costly disposal options.

7 Guidance for Addressing Nitrate Impacted Drinking Water

7.1 Checklist for the Selection of Mitigation Strategy

The following checklist is intended as a general guide for the selection of promising mitigation strategies for nitrate impacted drinking water (adapted from U.S. EPA 2003b, WA DOH 2005a, and WA DOH 2005b). Questions regarding mitigation strategy development should be directed to the Department of Public Health.[21]

1. Monitor nitrate concentration of distributed water and determine compliance status. Quarterly monitoring may be necessary and public notification requirements must be met.
2. Develop long-term compliance schedule with the Department of Public Health.
3. Determine all pertinent water quality characteristics (nitrate, arsenic and other co-contaminants, pH, TDS, sulfate, etc.).
4. Assess non-treatment options (e.g., removing well from service, blending, consolidation, development of new sources, etc.). See Decision Tree 1 (Figure 36) and Technical Report 7 (Honeycutt et al. 2012).
5. If non-treatment options are not feasible, determine evaluation criteria for treatment (e.g., effluent nitrate goal, operator certification, water demand, and state and local requirements) and assess treatment options. Choose optimal approach to addressing nitrate impacted source(s). See Decision Trees 1 and 2 (Figure 36 and Figure 37).
6. Develop preliminary or planning-level cost estimates for capital and O&M costs.
7. Assess design considerations. See Table 4 and Table 6 for details on IX and RO, respectively. Considerations for ED/EDR and BD are listed in Table 8 and Table 11, respectively.
8. Pilot test the selected solution (engineering professional required).
9. Develop construction-level cost estimates for capital and O&M costs (engineering professional required).
10. Examine funding options and attain funding (e.g., Drinking water state revolving fund (DWSRF) loan). See Technical Report 8 (Canada et al. 2012).
11. Implement the selected solution. This may include the development of a pre-design report, design, obtaining appropriate permits, construction, inspections, and start-up tasks (engineering professional required).
12. Monitor the system to ensure safe operation and the consistent supply of compliant drinking water (engineering professional may be required).

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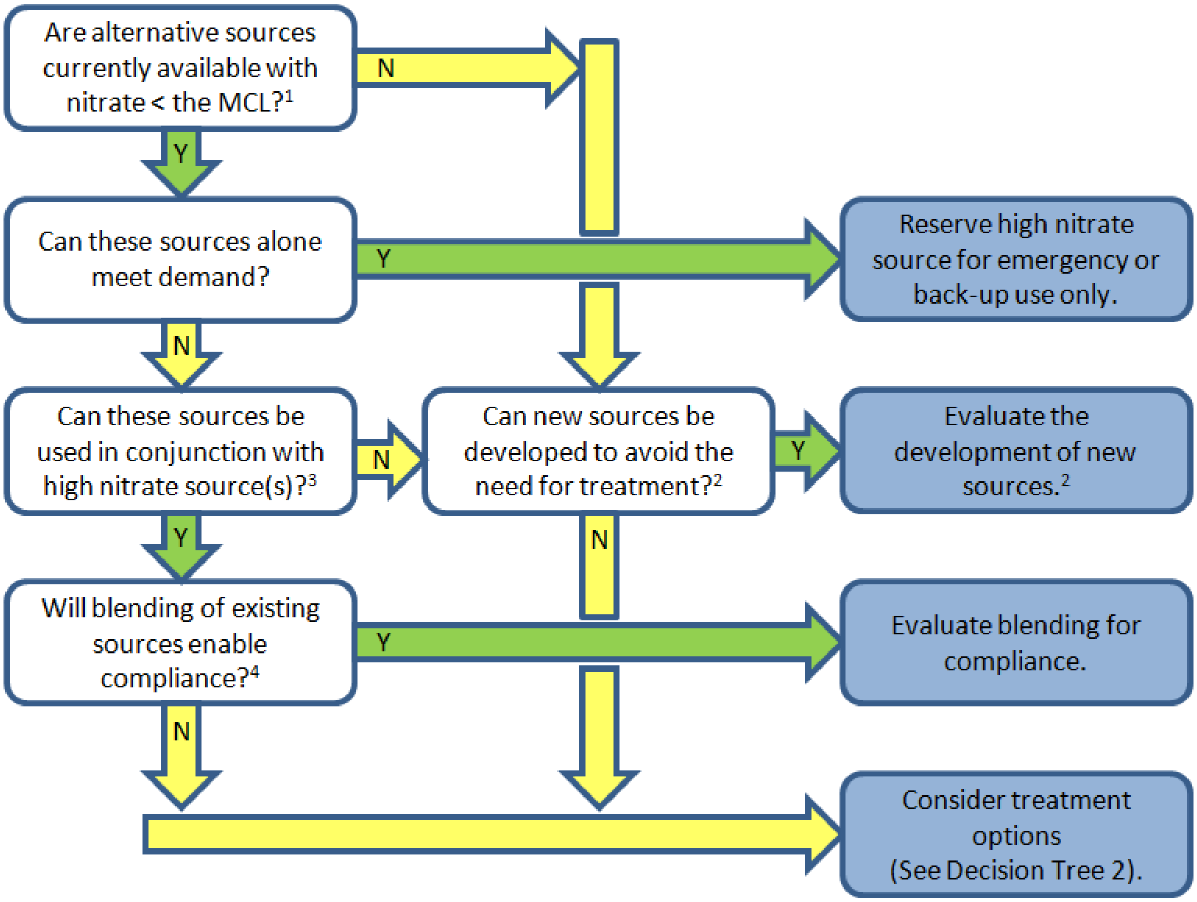
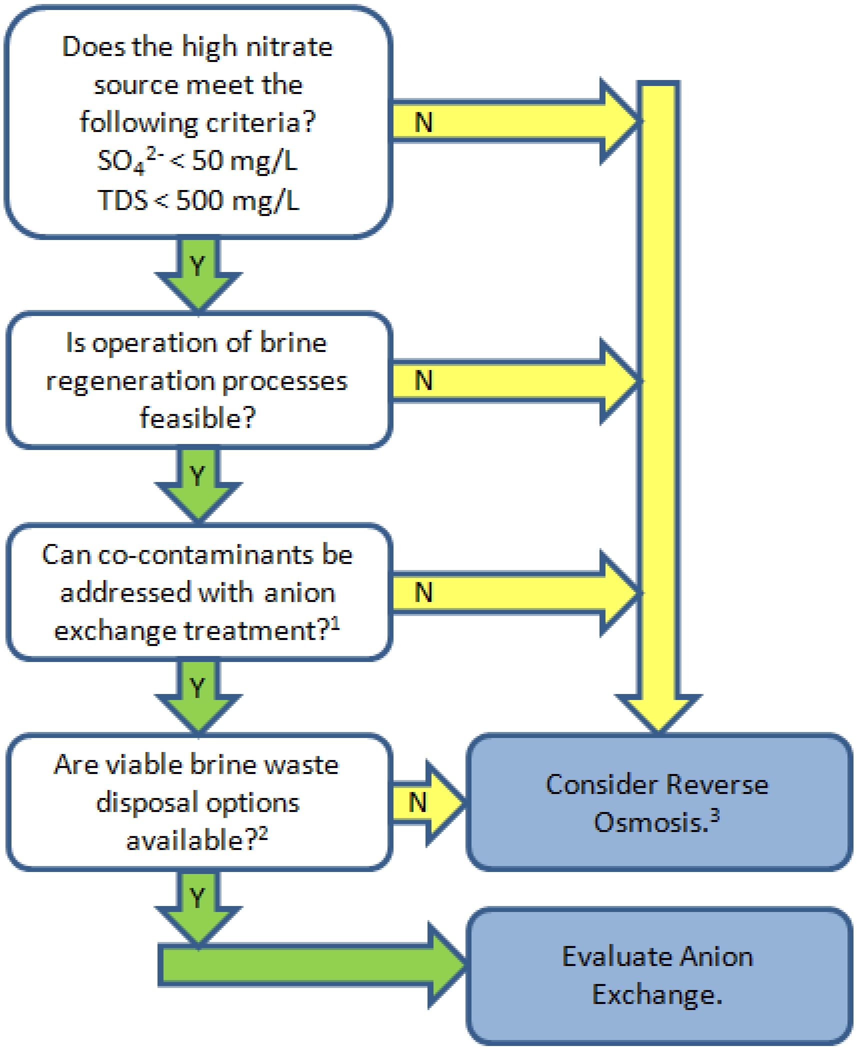
[21] Additional information can be found at http://www.cdph.ca.gov/certlic/drinkingwater/Pages/default.aspx and http://www.rcac.org/assets/.online%20materials/CA-DrinkingWater_Jan-June%202012.pdf.

7.2 Decision Trees



1. Currently available sources could include additional existing wells or surface water.
2. Potential new sources could include new well development, surface water, consolidation with nearby system(s) or connection to an existing nearby system.
3. Are there any obstacles to using sources together? For example, is the alternative source a great distance from the high nitrate source?
4. Is there a feasible blend ratio to provide compliant water that would consistently meet demand taking into consideration potential co-contaminants and seasonal variation in demand and/or water quality?

Figure 36. Decision Tree 1 - Options to address nitrate impacted drinking water sources (adapted from U.S. EPA 2003b, WA DOH 2005a, and WA DOH 2005b).



1. In addition to nitrate, certain co-contaminants can be addressed simultaneously through anion exchange treatment (e.g., arsenic and perchlorate); however, specialized resin may be required for different constituents.
2. Viable options can include disposal to sewer, evaporation ponds, and a brine line.
3. Reverse osmosis can address certain co-contaminants that anion exchange cannot and generally results in a less concentrated waste stream. For systems serving < 500 people, POU reverse osmosis is another option to consider.

Figure 37. Decision Tree 2 - Anion exchange (adapted from USEPA 2003, WA DOH 2005a, and WA DOH 2005b).

8 Summary and Conclusions

- Current full-scale nitrate treatment installations in the United States consist predominantly of ion exchange (IX) and reverse osmosis (RO). Other technologies are available because, under some circumstances, the alternatives offer advantages that IX and RO cannot. While electrodialysis (EDR) is a feasible option for nitrate removal from potable water, the application of EDR is generally limited to high TDS and/or high silica waters. The use of biological denitrification (BD) to address nitrate contamination of drinking water is more common in Europe than in the U.S. However, this option is emerging in the U.S. and two full-scale systems are expected within a few years. Chemical denitrification (CD) may become a feasible nitrate treatment option in the future; however, the lack of current full-scale implementation suggests the need for further research, development and testing. New technologies will continue to be investigated and developed because no single option is ideal for all situations. A single treatment solution will not fit every community; however, the provision of safe drinking water for all communities can be achieved using currently existing technology.

- Brine reuse and treatment are vital to the continued reliance on IX for nitrate treatment of potable water. The low brine technologies offer a minimal waste approach and current research and development of brine treatment alternatives seem to be lighting the path toward future progress.

- In regions with declining water quality and insufficient water quantity, the need to address multiple contaminants will increase in the future, suggesting the future dominance of technologies capable of multiple contaminant removal. In this context, for any individual water source or system, the most appropriate technology will vary with the contaminants requiring mitigation. Although complex, analysis of the optimal treatment option for pairs and groups of contaminants will assist in the treatment design and selection. In such scenarios, the best treatment option for nitrate may not be the most viable overall.

- Currently and into the future, selection of the optimal and most cost-effective potable water treatment options will depend not only on the specific water quality of a given water source, but also on the priorities of a given water system. If land is limited, the typical configuration required for biological treatment may not be feasible. If disposal options of brine waste are costly or limited, implementation of denitrification treatment or development of brine recycling and treatment may be the most suitable option.

- When deciding on nitrate treatment, the characteristics of the water system must be taken into account as well. With consideration of economies of scale, many rural small water systems cannot afford to install treatment. Even with financial assistance to cover capital costs, the long term viability of a treatment system can be undermined by O&M costs that are simply not sustainable. For such systems, treatment can become more affordable through consolidation of multiple small water systems into larger combined water systems that can afford treatment as a conglomerate. With a continued decline in water quality, non-treatment options alone, like blending or drilling a new well, may become insufficient measures for a water system to provide an adequate supply of safe and affordable potable water. Especially in rural small communities, perhaps the most promising approach will be a combination of consolidation and treatment. Alternatively, separate small treatment facilities can be consolidated under a single agency. For additional discussion on the comparison of alternative water supply options and associated costs see Technical Report 7 (Honeycutt et al. 2012).

- While current cost considerations are commonly the driving force in selecting nitrate treatment, it is essential to consider the long term implications of current industry decisions. For example, it may be cost-effective for a particular system to utilize conventional IX currently, but future water quality changes (e.g., increasing nitrate levels, co-contamination, high salt loading), discharge regulations, or disposal fees may lead to an unmanageable increase in costs. Environmental sustainability in drinking water treatment is being addressed with brine treatment alternatives and denitrification options. It is important to approach the future of drinking water treatment with the mindset that environmental sustainability and economic sustainability are tightly interwoven.

- Centralized treatment may not be feasible for widespread rural communities; another approach to consider is centralized management (e.g., design, purchasing, and maintenance) to minimize costs.

- Point-of-Use (POU) and Point-of-Entry (POE) treatment equipment is an important option to consider, especially for the provision of safe drinking water from private wells. Unless connecting to a nearby public water system becomes an option, users relying on domestic wells have two main alternatives: drilling a new well to attain safe drinking water or installing a POU or POE device for the treatment of contaminated water. The use of POU and POE treatment equipment by small public water systems is currently only a temporary option in California and reliance on these devices for the long-term would require regulatory changes. While POU and POE treatment equipment has been shown to effectively address nitrate and other contaminants, it is important to properly maintain these devices to ensure the supply of consistently safe drinking water.

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10 Appendix

10.1 Tables of Selected Research

Table A.1. Selected research on the use of ion exchange (IX) for nitrate removal.

IX – Nitrate selective resin	Dosing and sorption kinetics of Purolite A 520E (SBA resin). Influent nitrate concentration of 22.6 mg/L as N (100 mg NO_3^-/L). With no sulfate and chloride the treated bed volumes before breakthrough (BV) and resin capacity were 451 BV and 126.4 mg NO_3^-/g resin, respectively. With sulfate and chloride concentrations 10x that of nitrate, BV and resin capacity were 120 BV and 33.6 mg NO_3^-/g resin, respectively.	Samatya et al. (2006)
IX – Adsorption kinetics	Amberlite IRA 400 resin was found to be a suitable choice for nitrate removal with influent nitrate concentration of 0.23 – 4.07 mg/L as N (1 – 18 mg NO_3^-/L) and a 96% removal efficiency.	Chabani et al. (2006)
IX – Sulfate competition	Competition between sulfate and nitrate ions was explored. For the sulfate specific resin examined, findings indicate that sulfate selectivity increased with ionic strength.	Kim & Benjamin (2004)
IX – Co-contaminants	Compared two different anion exchange resins and studied the influence of other ions on nitrate removal.	Yoon et al. (2001)
IX – Resin residuals and DBPs	Quaternary amine groups and nitrosamines carcinogenic DBPs, 3 resins examined. With no disinfectants (chlorine and chloramine) "release 2 – 10 ng/L", max of 20 ng/L nitrosamine. "The lack of significant nitrosamine release in full-scale anion-exchange treatment system after multiple regeneration cycles indicates that releases may eventually subside." Precursors can be a problem with downstream chloramine use. Upstream disinfection releases of 20 – 100 ng/L (type 1) and 400 ng/L (type 2). Possible problem with IX in POU with influent containing chlorine/chloramines.	Kemper et al. (2009)
IX – Pilot study	3-year pilot examined RO, IX and biological denitrification in Mashhad, Iran. Treatment goal was a decrease in nitrate concentration from 26 mg/L as N (115 mg/L as nitrate) to 9 mg/L as N (40 mg/L as nitrate). Raw water with electrical conductivity of 1550 uS/cm, pH of 7.2. Optimal treatment options were deemed to be biological denitrification and RO. In the IX pilot plant pre-treatment consisted of cartridge filtration. Two IX columns containing ~200 L of nitrate selective resins were run in parallel with counter-current regeneration. System characteristics: Column diameter: 0.4 m, Bed Volume (BV): 200 L, Bed Depth: 1.6 m, Flow Rate: 1.0 to 3.0 m^3/h, Specific Flow Rate: 5 to 15 BV/h, Resin replacement every 5 years.	Panglisch et al. (2005) Dördelmann et al. (2006, and N.D.) Dördelmann (2009)
See Also	Clifford & Weber (1978), Guter (1982), Clifford (1987), Rash (1992), Clifford (2007), and Johnston & Heijnen (N.D.).	

Table A.2. Selected research on the use of reverse osmosis (RO) for nitrate removal.

RO – Pilot Study and Full-scale Installation	Brighton, CO: Use of RO in Brighton, CO, for potable water treatment including nitrate removal began operating in 1993 (11000 sq ft., 4 MGD, total cost of $8,253,000) after the performance of preliminary pilot study (4 membranes were tested). Source water nitrate concentration of 13 to 23 mg/L nitrate-N or 58 to 102 mg/L nitrate as NO_3^- (with TDS 800 – 1140 mg/L, hardness 370 – 480 mg/L as $CaCO_3$). Problems with biofouling of membrane and cartridge filters (slime forming *Pseudomonas*) minimized with anti-scalant use. Pretreatment: acid, anti-scalant and filtration. Post-treatment: CO_2 stripping (degasifier), disinfection, caustic and zinc orthophosphate (ZOP) addition. Blending ratio (untreated%:treated%) 20:80(winter) and 60:40 (summer). Waste discharge to surface water. Additional characteristics: Flux rate < 14.2 gal/sq. ft./day (helps control fouling), 2 stages: 32 : 16 vessels, pressure of 231 psi, each vessel with 6 8” diameter membranes. Cleaning required every 2 months.	Cevaal et al. (1995)
RO – HERO in Australia	Yalgoo, Australia: RO plant, online in 2007 using high-efficiency reverse osmosis (HERO). Water recovery rates as high as 95%. More than 85% waste reduction (a low as 10% the waste of conventional RO.) Removes both nitrate and silica from brackish water.	Water Corporation (2009)
RO – Full-scale uses methane	Inland Empire, CA: Cow Power in Inland Empire, CA: $80 mil. effort is powered by methane gas produced from the high population of cows in that region. Online since 2002, this plant provides 1/5th of regional demand.	Black (2003)
RO – Pilot study comparing IX, RO and BD	Mashhad, Iran: 3-year pilot examined RO, IX and BD in Mashhad, Iran. Treatment goal of 9 mg/L as N (40 mg/L as nitrate) (influent nitrate of 26 mg/L as N, 115 mg/L as nitrate). Results indicate BR and RO were best choices. Automated RO pilot plant with a “capacity [of] 3 m^3/h (RO permeate).” Pre-treatment: acid, anti-scalant, automated filtration (50 um), and cartridge filtration (1um). 2 stages: in the first stage, 2 parallel membranes, second stage, concentrate from stage one to second membrane before mixing with stage 1 permeate. Post-treatment: Blending, CO2 removal. New membranes every 5 years. RO was least expensive despite having highest energy demands (0.6 kWh/m^3 of DW), due to low cost of electricity. Lower operator demand than BD. RO produced better quality water (lower TDS, Ca, Mg, Na, K, Cl, nitrate, sulfate and bicarbonate), but the greatest percentage of waste.	Panglisch et al. (2005) Dördelmann et al. (2006 and ***N.D.***) Dördelmann (2009)
RO – Full-scale installations	Milan, Italy: Influent of 11 – 14 mg/L as N (50 – 60 mg/L as nitrate), target of 9 mg/L as N (40 mg/L as nitrate) with waste concentrate < 30 mg/L as N (132 mg/L as nitrate) (for sewer disposal). Series of 13 1-stage RO plants (7 to 58 m^3/hr permeate), with blending 77 – 88% water recovery (note the disposal limitation). Pretreatment: anti-scalant (2.0 – 3.5 mg/L). Cleaning only every year and a half (SDI<1) and requires only 3 techs.	Elyanow & Persechino, (2005)
RO – Alternative disposal options	Alternatives to conventional disposal measures of RO waste brine, including reuse for industrial processes, processing (e.g., for salt production), or use in energy generation (“solar brine pond”).	Howe (2004)
RO – Membrane scaling and fouling, colloidal interaction	Analysis of complex source water, membrane fouling and ways of anticipating changes in flux and rejection rates. Relationship between constituents and how RO treatment is affected by colloids, silica, concentration polarization (higher salt concentration near membrane surface), back diffusion, and scaling. Typical colloidal constituents: “sulfur, silica, and ferric and aluminum hydroxides.” Results indicate colloidal fouling may amplify other scaling/fouling factors. “By recognizing possible interferences between rejected salts and colloids deposited on the membrane surface, the work explored the phenomenon of coupling between colloidal fouling, concentration polarization, and scaling as main factors limiting the application of RO membranes” (Tarabara 2007).	Tarabara (2007) Wang & Tarabara (2007)
RO – Influence of upstream processes	Investigation of the interaction of upstream residuals on RO treatment (membrane and additive chemicals). Influence of coagulant residuals on colloidal fouling, and disinfectants on membrane oxidation.	Gabelich et al. (2004)

Table A.3. Selected research on the use of electrodialysis/electrodialysis reversal for nitrate removal.

ED – Pilot study and Full-scale installation	Austria: 2-year pilot starting in 1990 with 1 m^3/h capacity. Influent nitrate concentration 18 – 23 mg/L as N (80 – 100 mg/L as nitrate), design for 36 mg/L as N (160 mg/L as nitrate). Planning began in 1996 for full-scale installation with seasonal operation commencing in 1997. Disposal options: sewer or irrigation reuse. "Monovalent selective anion exchange membranes." 3 stacks in parallel – 3 stages offer capacity from 48 to 144 m^3/h. Complete automation. Effluent nitrate of 9 mg/L as N (40 mg/L as nitrate) (capable of product water nitrate of 5.7 mg/L as N, 25 mg/L as nitrate) and 23% hardness reduction. Nitrate selectivity: 66% nitrate removal with 25% desalination.	Hell et al. (1998)
ED – Pilot study – Optimization	Morocco: Pilot study using a nitrate selective membrane for nitrate removal from an influent level of 20 mg/L as N (90 mg/L as nitrate) to acceptable levels, in water with a TDS concentration of 800 mg/L. Analysis of operating parameters to minimize precipitation, scaling and associated chemical use. Variation of voltage, flow rate and temperature. Ion removal increased with increasing temperature.	Midaoui et al. (2002)
EDR – Full-scale installation	Delaware: EDR has been successfully employed for the removal of nitrate from potable water in Delaware. The nitrate concentration in treated potable water was 1 mg/L as N (4.4 mg/L as nitrate), just 7.5% of the 13.5 mg/L as N (60 mg/L as nitrate) influent nitrate concentration. 3 stage system, 88% demineralization (TDS reduction), 90% water recovery, and pH decrease from 6.2 to 5.4.	Prato & Parent (1993)
EDR – Full-scale installation	Barcelona, Spain: GE EDR plant with 50 MGD capacity (260,000 households), not specifically for nitrate. Limited to no anti-scalant use. "Compared to a typical RO treatment facility producing 3.8 million gallons of water per day, GE's EDR technology, operating at 83% efficiency, is designed to eliminate the need for over 28,000 pounds of anti-scalant, reducing operating costs by > $100,000 per year at typical 2008 chemical prices." *Previous EDR systems for nitrate removal were installed in Arizona, Delaware, Japan, Italy, Bermuda, and Israel.	GE (2010), GE (N.D.)
EDR – Full-scale installations	Europe: As of ~2005, Ameridia/Eurodia full-scale EDR plants in Italy, France, Switzerland, Netherlands, and Austria. Capacity ranging from 0.032 – 0.925 MGD and water recovery from 93 – 98%. Information on full-scale installations, pilot studies and costs.	Ameridia/Eurodia
ED – Multiple contaminant removal, role of organic matter	Research investigation using synthetic waters to determine the impact of organic matter in the removal of nitrate, fluoride and boron. Fouling led to decreased flux, although nitrate removal was the least affected, due to smaller "hydrated ionic radius." Removal of boron and fluoride was enhanced by the presence of organic matter, while nitrate removal was not enhanced and simply decreased over time with the decrease in membrane flux.	Banasiak & Schäfer (2009)
ED – Pilot study	Research investigation using synthetic waters to assess nitrate removal under different operating conditions. The impact of different voltage (from 40 to 50 V) was examined across several ionic species, including nitrate. Alternating anion and cation exchange membranes were used. Results indicate 94% nitrate removal, with a reduction in the removal rate at 50 V due to back diffusion and fouling.	Nataraj et al. (2006)
ED – Pilot study comparison with adsorption	Morocco: Nitrate removal from brackish water. Comparison of ED using a monovalent membrane and adsorption on chitosan. ED successfully removed nitrate. Adsorption can remove nitrate, but not likely feasible. Adsorption can be used to remove nitrate from ED waste concentrate. Highlights concerns regarding waste concentrate disposal from ED.	Sahli et al. (2008)
	Mashhad, Iran: Comparison of IX, RO, BD, and ED. ED pilot study was started in 2007.	Dördelmann (2009)

Table A.4. Selected research on the use of biological denitrification for nitrate removal from potable water.

Pilot Testing - Multiple Biological Configurations
The City of Glendale, AZ, has investigated three configurations of biological treatment to address high nitrate levels in groundwater wells (Meyer et al. 2010). An autotrophic MBfR using was compared with two heterotrophic fixed bed bioreactors each with different media (plastic versus granular activated carbon). Hydrogen gas and ethanol were used as electron donor for the autotrophic system and the heterotrophic systems, respectively. Post-treatment included filtration using biologically activated carbon and ozonation. The fixed bed bioreactor with plastic media and the MBfR performed well, with product water meeting or exceeding potable water standards. Multi-criteria analysis found the MBfR to be most favorable regarding benefits, but the least favorable regarding costs. Including comparison with IX, the fixed bed bioreactor with plastic media had the lowest life cycle cost. The MBfR costs were greatest.
An investigation of biological treatment options in the City of Thornton, CO, funded by the WaterRF, examined two packed bed bioreactors and a moving bed biofilm reactor (MBBR™) (City of Thornton 2010, Project # 4202) to address nitrate impacted source water. Nitrate levels were successfully decreased by each of the three pilot systems from an influent concentration of 10 mg/L nitrate as N to an effluent concentration of < 2 mg/L nitrate as N. Operation at high and low temperatures was tested with examination of seeding for low temperatures. The study highlights the need for nutrient and substrate dose optimization in biological treatment systems.
Substrates
Numerous alternative substrate options have been explored in the literature including newspapers, vegetable oil, cotton, and formate (Volokita et al. 1996; Hunter 2001; Killingstad et al. 2002; and Della Rocca et al. 2006.)
Fixed Bed
See Riverside, CA Case Study (Carollo Engineers 2008).
A fixed bed heterotrophic denitrification pilot study was implemented in Mashhad, Iran by Dördelmann et al. (2006) using two parallel fixed beds containing expanded clay media. Acetic acid and ferrous sulfate served as the electron donor and nutrient supply, respectively. Post treatment consisted of "aeration, dual media and activated carbon filtration" (Dördelmann et al. 2006). Influent nitrate levels of 26 mg/L as N (115 mg/L as nitrate) were decreased to < 9 mg/L as N (40 mg/L as nitrate) with a nitrate reduction rate of ~7 kg NO_3/m^3 d (0.43 lb NO_3/ft^3d) (Panglisch et al. 2005 and Dördelmann et al. 2006). Used for flushing and backwashing, 7% of influent volume was wasted. In practice, a final disinfection step would be required (Dördelmann et al. N.D).
An up-flow, fixed-bed, autotrophic, lab-scale system, using granular sulfur as both substrate and growth surface was explored by Soares (2002). Operated over a 5 month period, a maximum denitrification rate of 0.2 kg N/m^3d (0.012 lb N/ft^3d) was achieved with a one hour hydraulic retention time and a loading rate of 0.24 kg N/m^3d (0.015 lb N/ft^3d). Sulfur based autotrophic systems would not be appropriate for the treatment of feed waters high in sulfate.
Aslan (2005) examined a lab-scale packed sand bed system, with ethanol as substrate for the simultaneous removal of nitrate and several pesticides. After 3 days for biofilm development, 93 – 98% nitrate removal was achieved requiring at least a 2 hour residence time. Pesticide removal required longer residence times (up to 12 hours) for efficient removal.
Upadhyaya et al. (2010) investigated the use of a fixed-bed biological reactor with granular activated carbon media for the removal of nitrate and arsenic at the same time. The media was biologically activated from use in a separate bioreactor for the removal of nitrate and perchlorate. Reactors were thus biologically active carbon (BAC) reactors. With acetic acid as the substrate, two in series BAC reactors were used to treat synthetic groundwater. Arsenic levels were reduced from 200 µg/L arsenic in the influent to 20 µg/L in the effluent (still above the arsenic MCL of 10 µg/L) while nitrate levels were decreased from 11 mg/L as N (50 mg/L as nitrate) in the influent to less than 0.045 mg/L as N (0.2 mg/L as nitrate) in the effluent. Used as an electron acceptor by microbes in the oxidation of substrate, nitrate was reduced to nitrogen gas. Arsenic was removed from solution with the formation of arsenic sulfide (solids) and also with adsorption and "surface precipitation on iron sulfides (p. 4958)."

Fluidized Bed
See Rialto, CA Case Study (Webster & Togna 2009).
Kurt et al. (1987) investigated an autotrophic fluidized sand bed reactor using hydrogen gas as substrate. With an influent nitrate concentration of 25 mg/L nitrate-nitrogen, a maximum nitrate reduction rate of 5 mg/L per hour was attained using a mixed culture. The authors propose multiple stage reactors to address the problem of partial denitrification.
Using a mix of propionic acid and ethanol in a heterotrophic fluidized sand bed reactor, Holló & Czakó (1987) examined denitrification at the lab- and pilot-scale. Post-treatment consisted of cartridge filtration, gas exchange, sand filtration, carbon filtration and disinfection. "Nitrate removal capacity of the reactor was 50 – 60 kg NO_3^-/m^3/day (3.1 – 3.7 lb NO_3^-/gal/d), which could be maintained permanently at temperatures as low as 8 – 10^{o}C as well" (Holló & Czakó 1987).
MBR - Diffusive Extraction and Microporous Membranes (See also Pilot Testing above)
Mansell & Schroeder (2002) assessed hydrogenotrophic denitrification at the lab-scale using a membrane bioreactor (MBR) with a 0.02 micron microporous membrane through which nitrate diffuses to the biological compartment. The membrane prevents mixing of microbes with the water being treated and no carbon substrate was necessary because hydrogen gas was supplied as the electron donor for autotrophic denitrification. Previous issues regarding the transfer of hydrogen gas to the water and safety concerns due to explosive nature of hydrogen gas have been addressed with the development of "membrane dissolution systems" (Mansell & Schroeder 2002). Hydrogen gas was delivered to the biomass with silicone tubing. Results indicated reduction of nitrate levels from a maximum of 40 mg/L NO_3^--N in the feed water to 3.2 mg/L NO_3^--N in the treated water, with 92 – 96% removal. Measured HPC indicated minimal biomass transfer to the treated water compartment.
Ergas & Rheinheimer (2004) studied denitrification of potable water using a membrane bioreactor (MBR) in which feed water is passed through tubular acrylonitrile membranes, nitrate diffuses through the membrane and denitrification occurs on the exterior membrane surface. The mean transfer to the biofilm was 6.1 g NO_3-N/m^2 d (0.6 g NO_3-N/ft^2). The ultimate methanol (substrate) loading rate of 1.1 g/d resulted in a mean concentration of nitrate in the treated potable water of 5.2 mg NO_3^--N/L. A mathematical model of nitrate mass transfer was developed. A removal efficiency of 99% was achieved with a starting concentration of 200 mg NO_3^--N/L. Use of the MBR allows for denitrification with separation of the water to be treated and biological treatment, thereby avoiding post treatment removal of biomass and dissolved organics. The effluent would require additional treatment, because 8% (30 mg/L) of the methanol crossed the membrane; the authors suggest that further development of the biomass could minimize methanol transfer to the effluent stream.
MBR - Gaseous Substrate Delivery – Hollow Fiber Membranes
Chung et al. (2007) explored the use of autotrophic denitrification for the treatment of highly concentrated waste from nitrate removal via anion exchange. Using a hydrogen-based, hollow fiber membrane biofilm reactor, the impact of brine concentration (up to 15%) on nitrate reduction was found to be significant due to microbial inhibition. In the reduction of nitrate, use of hydrogen gas rather than an organic substrate offers an inexpensive alternative for potable water treatment systems. Biomass production is decreased and there is no need to remove substrate residual, as there would be with the use of carbon substrates.
Using nitrate as the primary electron acceptor, hollow fiber MBfRs with hydrogen gas as electron donor can effectively decrease the levels of multiple contaminants including perchlorate, chromate and arsenate (Nerenberg & Rittman 2004). Low levels of these oxidized species without a primary electron acceptor can limit biological reduction; however, in the presence of nitrate, reduction can occur. When the concentrations of nitrate and the oxidized species of interest were 1.1 mg/L as N (5 mg/L as nitrate) and 1 mg/L, respectively, 99% nitrate removal was achieved while removal of perchlorate, chromate and arsenate reached 36%, >75%, and >50%, respectively.

Bioelectrochemical Denitrification
Ghafari et al. (2008) provide a review of biological denitrification with a focus on bioelectrical reactors (BERs). In a BER, hydrogenotrophic denitrification occurs as hydrogen gas is produced at the cathode and utilized as the electron donor by denitrifiers, while nitrate is reduced to nitrogen gas. Previous BER research is discussed including autotrophic and heterotrophic examples across a range of nitrate levels and generally in synthetic waters. With additional research, BERs may become a feasible alternative for nitrate removal from drinking water.
In situ Denitrification
Permeable reactive barriers (PRBs) can be used to directly treat nitrate contaminated groundwater. Hunter (2001) examined the use of vegetable oil as an electron donor in biological denitrification. The use of an insoluble substrate minimized biomass blockage, a problem common with the use of soluble substrates like ethanol, methanol and acetate. The barrier was composed of soybean oil-coated sand and effectively decreased the nitrate levels from a starting concentration of 20 mg/L nitrate-N to below the MCL for a period of 15 weeks, with a flow rate 1100 L/week. After 15 weeks, insufficient oil remained for denitrification. High chemical oxygen demand, TSS and turbidity in the effluent of the reactor indicate a longer sand bed was needed; however, the author suggests that in situ application of this type of biological reactor would decrease these factors naturally. With a withdrawal point far enough from the barrier, subsequent potable water treatment requirements would be limited to disinfection. The most significant problem encountered in this study was the exhaustion of substrate. An effective means of substrate addition must be found (injection for example), but this was not explored. The estimated life of the PRB was 2.5 – 12.5 years depending on several key factors including flow, nitrate concentration and dissolved oxygen concentration.
With hydrogen gas as the substrate in autotrophic denitrification, Haugen et al. (2002) examined hydrogenotrophic denitrification in a lab scale experiment intended to imitate in situ treatment. Denitrification kinetics, the feasibility and longevity of substrate delivery via tubular membranes and post-treatment water quality were investigated. Delivery of hydrogen gas through tubular membranes minimized the risks associated with utilization of this flammable/explosive gas. The reactor was tested over 155 days. An initial influent nitrate concentration of 8.2 mg NO_3^- -N/L was doubled to 16.4 mg NO_3^- -N/L. After adjustment of parameters, complete nitrate removal was achieved using the tubular membrane bioreactor. A denitrification rate of 169 mg $N/h/m^2$ (membrane surface area) was attained with a hydrogen gas pressure of 1.44 atm (lower pressures resulted in incomplete reduction. The greatest hydrogen gas transfer across the membrane (flux) was 1.79×10^{-2} mg H_2/m^2s. The simulated groundwater velocity was 0.3 m/d resulting in 14 minutes of membrane contact time. Additional considerations for application of this treatment method include: the lower temperature of groundwater, the need for buffer in the current study, the depth limitation, nutrient requirements, and the difference between aquarium rocks and subsurface porous media. Intermediate denitrification products and end products (ammonia and nitrogen gas) were not measured in this study; however, the authors suggest the high nitrate to substrate ratio would result in reduction to nitrogen gas.
Schnobrich et al. (2007) simulated in situ nitrate removal via hydrogenotrophic Denitrification. With hydrogen gas delivery through a membrane module consisting of a fiberglass membrane wound in a spiral fashion and attached to polyethylene membranes. The study examined the influence of pH, nutrient requirements and the feasibility of appropriate levels of hydrogen gas delivery. To simulate in situ conditions, the porous media was extracted from an aquifer and the system was operated at 10^oC. Two up-flow columns were operated in series, only the first of which included a membrane fed with hydrogen gas. Including that required for the reduction of dissolved oxygen, the total concentration of hydrogen gas required for complete denitrification was 11.2 mg H_2/L. Overall, this study demonstrated effective substrate delivery and nitrate removal under conditions more similar to what would be expected naturally.

Table A.5. Selected research on the use of chemical denitrification (CD) for nitrate removal.

CD using ZVI	Reduction of nitrate to ammonia using ZVI powder was highly pH dependent with optimal kinetics below a pH of 4. The minimum ratio of iron to nitrate was 120 m^2/mol NO_3^- for complete reduction within 1 hour. 50 mg NO_3^-/L, 100% removal.	Huang et al. (1998)
CD – ZVI	Found that the end product of denitrification (nitrogen gas versus ammonium) could be controlled by the iron to nitrate ratio and the use of catalysts.	Xiong et al. (2009)
CD – ZVI	In the corrosion of ZVI, the formation of "green rusts" and "suspended green particles" is associated with stabilization of pH and steady decrease in nitrate.	Choe et al. (2004)
CD – ZVI	Examined the nitrate reduction rates of three types of iron. Findings indicate that rate increases with decreasing pH.	Alowitz & Scherer (2002)
CD – ZVI	Nitrate reduction by ZVI can be optimized through pretreatment of iron particles. High temperature exposure to hydrogen gas and deposition of copper were explored separately as options for pretreatment of the iron surface.	Liou et al. (2005)
CD – ZVI	Examined chloride as a potential competitor. Results indicate a minimal impact on nitrate removal; however, other competing ions could be important regarding both competition for adsorption sites and reduction.	Moore & Young (2005)
CD - SMI	"SMI-III® is a patented, iron-based granular media that has been commercially developed for the removal of nitrate, co-contaminants including uranium, vanadium and chromium, and other compounds from water. It is foreseen that the greatest benefit of this technology is that it does not produce a costly brine stream as do the currently accepted nitrate removal technologies of ion exchange and reverse osmosis."	DSWA and City of Ripon (2010)
CD – Catalytic Denitrification	Reddy & Lin 2000; Pintar et al. 2001; Gavagnin et al. 2002; Lemaignen et al. 2002; Pirkanniemi & Sillanpaa 2002; Chen et al. 2003; Palomares et al. 2003; Pintar 2003; Constantinou et al. 2007; and Sun et al. 2010.	

Table A.6. Advantages and disadvantages of the five major treatment options for nitrate removal.

	Advantages	Disadvantages
Ion Exchange	• Years of industry experience, • Multiple contaminant removal, • Selective nitrate removal, • Financial feasibility, • Use in small and large systems, and • The ability to automate.	• The disposal of waste brine, • The potential for nitrate dumping specifically for non-selective resin use for high sulfate waters, • The need to address resin susceptibility to hardness, iron, manganese, suspended solids, organic matter, and chlorine, and • The possible role of resin residuals in DBP formation.
Reverse Osmosis	• High quality product water, • Multiple contaminant removal, • Desalination (TDS removal), • Feasible automation, • Small footprint, and • Application for small and POU applications.	• The disposal of waste concentrate, • Typically high capital and O&M costs, • The need to address membrane susceptibility to hardness, iron, manganese, suspended solids, silica, organic matter, and chlorine, • High energy demands, and • The lack of control over target constituents (complete demineralization).
Electrodialysis/ Electrodialysis Reversal	• Limited to no chemical usage, • Long lasting membranes, • Selective removal of target species, • Flexibility in removal rate through voltage control, • Better water recovery (lower waster volume), • Feasible automation, and • Multiple contaminant removal.	• The disposal of waste concentrate, • The need to address membrane susceptibility to hardness, iron, manganese, and suspended solids, • High maintenance demands, • Costs (comparable to RO systems), • The need to vent gaseous byproducts, • The potential for precipitation with high recovery, • High system complexity, and • Dependence on conductivity.
Biological Denitrification	• High water recovery, • No brine or concentrate waste stream (nitrate reduction rather than removal to waste stream), • Low sludge waste, • Less expensive operation, • Limited chemical input, • Increased sustainability, and • Multiple contaminant removal.	• The need for substrate and nutrient addition, • High monitoring needs, • Significant post-treatment requirements, • High capital costs, • Sensitivity to environmental conditions (sometimes), • Large system footprint (sometimes), • High system complexity (sometimes, can be comparable to RO), • Lack of full-scale systems in the U.S., • The possibility of partial denitrification, • Permitting and piloting requirements, and • Slower initial start-up, which could cause challenges for wells with intermittent run time.

Chemical Denitrification	• Conversion of nitrate to other nitrogen species (no brine or concentrate waste stream),
	• The potential for more sustainable treatment,
	• High water recovery (higher than RO according to Cleanit®-LC), and
	• Multiple contaminant removal.
		• The potential reduction of nitrate beyond nitrogen gas to ammonia,
		• The possibility of partial denitrification,
		• The possible dependence of performance on pH and temperature,
		• The possible need for iron removal, and
		• The lack of full-scale chemical denitrification systems resulting in:
		  ○ Unknown reliability,
		  ○ Unknown costs, and
		  ○ Unknown operational complications.