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Estimating costs for nitrate and perchlorate treatment for small drinking water systems

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Abstract

When choosing a treatment technology for nitrate or perchlorate removal, drinking water utilities overwhelmingly choose ion exchange. However, of late, biological treatment and point-of-use systems have received a great deal of attention. This article utilizes several new U.S. Environmental Protection Agency models to estimate the cost of nitrate and perchlorate treatment for small drinking water systems. The analysis here shows that, when comparing the three technologies for a typical set of design choices and drinking water quality conditions, the least-cost option varies among the three depending on system size. This relationship varies with changes to the water quality and design factors such as, but not restricted to, influent nitrate and perchlorate concentrations, the choice of residual management options, and the presence of co-contaminants and competing ions.

Keywords

cost; nitrate; perchlorate; treatment

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CONFLICT OF INTEREST
The authors declare there are no conflicts of interest.

DATA AVAILABILITY STATEMENT
The data that support the findings of this study are available from the lead author upon reasonable request.

1 | INTRODUCTION

The Safe Drinking Water Act (SDWA) categorizes small systems on the basis of population served. Pursuant to SDWA Section 1412(b)(4)(E)(ii), the US Environmental Protection Agency (USEPA) defines small systems as serving 10,000 people or fewer. Small drinking water systems have a number of challenges that their larger counterparts typically do not encounter. Perhaps the greatest issue is access to capital resources; however, other factors can be just as significant, such as lack of technical expertise and managerial capacity. Also, because small systems largely draw from groundwater sources whereas large systems often pull from surface waters, contaminants such as nitrate are more prevalent in small systems than large systems. Nitrate is a significant concern in small systems in the United States because it frequently occurs above the maximum contaminant limit (MCL) of 10 mg/L (as nitrogen). In 2018, more than 4000 small systems across the United States had at least one instance where they exceeded the MCL for nitrate (USEPA, 2019a). Nitrate's physical/chemical properties are such that it is not removed by conventional or simplistic treatment systems, making treatment a potentially complex and costly affair. Perchlorate has no MCL and is not detected as frequently as nitrate, and perchlorate concentrations in drinking water have declined over time (USEPA, 2020). However, perchlorate remains a contaminant of concern and the two contaminants' physical/chemical properties are similar, resulting in the same potential problems for small systems deciding on a treatment scheme. Both contaminants are oxyanions and can be removed by the same treatment processes, so this article presents treatment costs for each of the two contaminants for purposes of comparison.

There are three types of nitrate or perchlorate treatment approaches that hold promise for small systems. These include centralized anion exchange treatment, centralized biological treatment, and point-of-use (POU)/point-of-entry (POE) reverse osmosis (RO) treatment. Of the three, nitrate removal by anion exchange resins has been known to be a successful treatment for many decades (USEPA, 1983), and is the most commonly implemented treatment choice for small systems. In anion exchange treatment, water passes through a bed of synthetic resin. Nitrate ions in the water are exchanged with ions, typically chloride, that are preloaded onto the resin. When the capacity of the resin is exhausted, it is regenerated to restore it to its initial condition using a solution of sodium chloride (also known as brine). The spent regenerant brine is a concentrated solution of the removed nitrate and is commonly high in dissolved solids and excess regenerant ions (e.g., sodium, chloride). This waste stream requires disposal or discharge. In some instances, small systems may be challenged by the operation of the regeneration procedure and may not have practical options for disposing of the regeneration solution. For perchlorate, ion exchange is also an effective technology. However, in this application the process frequently uses single-use perchlorate-selective resins. These perchlorate-selective resins are usually disposed and replaced instead of being regenerated due to their high capacities and the difficulty in regenerating the resin. Single-use, as opposed to regenerable, resins also lend themselves to small system use because of the simpler operational requirements.

Biological treatment uses indigenous bacteria to remove contaminants. The treatment process passes water through a bioreactor that contains the bacteria attached to media in either a fluidized bed or a fixed bed. Because nitrate and perchlorate are fully oxidized, anoxic conditions are needed for contaminant removal. This adds an additional level of complexity compared with aerobic treatment. The anerobic bacteria, in combination with an electron donor (e.g., acetic acid) and nutrients (e.g., phosphorus), reduce the nitrate and perchlorate anions and produce biomass and other by products. Although the anoxic biotreatment of drinking waters for nitrate removal has been successfully implemented in Europe since at least the 1970's and has been discussed in the literature for decades (Bouwer & Crowe, 1988; USEPA, 1983), it has only recently been receiving serious attention as an option for drinking water systems in North America (Brown et al., 2015; Webster et al., 2009). Biological treatment of perchlorate has been successfully implemented at the full scale in recent years (Webster & Litchfield, 2017). Even with this success, the need to determine and maintain the optimal nutrient and electron donor feeds can entail operator attention or complex automation.

In addition, for anoxic biological treatment, post-treatment processes would likely be required to account for the addition of soluble microbial organic products and depletion of oxygen in the treated water (Gilbert et al., 2001; Harding ESE, 2001). The potential of sulfate reduction to hydrogen sulfide, which is often the next reduction reaction to be thermodynamically favorable after that for nitrate and perchlorate, will likely require a utility to install post-treatment safeguards. Because of these reasons and the limited full-scale track record, few utilities in the United States currently use anoxic biological treatment. However, there is interest in anoxic biological treatment because it could be a low-cost option that may not generate any waste streams. Additional benefits could be realized if the process could be operated similar to aerobic systems that do not require a great deal of operator attention.

POU/POE RO treatment is essentially a miniaturized version of centralized RO. It uses the same process of separating nitrate and perchlorate from water by forcing the water to flow through a membrane at pressure. RO has been shown to be effective for nitrate and perchlorate removal, given that the correct membrane is chosen (Bellona et al., 2008; Yoon et al., 2005). Given a membrane that effectively rejects nitrate or perchlorate, the process will generate a concentrated reject waste stream that requires discharge or disposal. However, because POU devices, in particular, treat a small fraction of the water delivered by a system, the absolute quantity of reject waste is significantly smaller than in centralized RO treatment and can be discharged down the drain. Also, the smaller quantity of water treated by POU/POE devices results in cost advantages for small systems. In centralized treatment, high-pressure membranes produce water that has corrosive characteristics that need to be addressed before the water leaves the plant. For POU devices, this is not a concern because the water does not enter any premise plumbing after treatment; however, POE devices that use high-pressure membranes may need to address this issue. Finally, another potential issue with the POU/POE approach, when used for compliance purposes, is that USEPA requires that a system install, control (i.e., own), and maintain POU/POE devices at all customer locations where water is consumed (e.g., residences) (USEPA, 2006). Engaging all households to achieve a full participation in a given system can be increasingly difficult as system size (i.e., the number of households served) increases.

Given the differing advantages and disadvantages of these treatment technologies, there is a need to compare the costs of these approaches on a consistent basis. The cost models developed by the USEPA provide a tool to conduct this evaluation (Khera et al., 2013). Although these models were primarily designed for estimating regulatory compliance costs as part of USEPA's rulemaking process, they can also be used to compare technology costs. The models use a work breakdown structure (WBS) approach for individual treatment technologies, linked to a central spreadsheet of component costs from which the discrete unit costs for all treatments can be obtained for the purpose of estimating total costs. This approach can provide a detailed assessment of both the capital and operating cost requirements of a treatment system. The models described below for anion exchange, biological treatment, and POU/POE RO treatment have flexible inputs and assumptions that can be varied easily, and the internal calculations are made in a visible and auditable manner to allow for cross comparisons.

The previous article (Khera et al., 2013) described the overall WBS approach and announced the public release of models for three technologies (granular activated carbon [GAC], packed tower aeration, and multistage bubble aeration). This article describes new WBS models for ion exchange, biological treatment, and POU/POE RO treatments, and presents results comparing the costs between the three technologies for removing nitrate and perchlorate at small drinking water systems.

2 | REVIEW OF MODEL FEATURES AND STRUCTURE

Khera et al. (2013) discussed the development history, design, and features of the WBS models in general. For conciseness, the present study includes only a brief review of the relevant issues specific to the three technologies evaluated herein. Table 1, updated from the previous publication, shows the status of all USEPA's WBS models. Several of these models are available to the public at: https://www.epa.gov/dwregdev/drinking-water-treatment-technology-unit-cost-models-and-overview-technologies.

2.1 | Contaminants

Although this article is reviewing nitrate and perchlorate treatment, each WBS model is capable of estimating treatment costs for different contaminants. For example, the anoxic biological treatment model allows the user to utilize standard input data for nitrate or perchlorate. The anion exchange model allows the user to select between nitrate and arsenic input data sets. The USEPA has developed a second, separate anion exchange model specifically for perchlorate. The POU/POE model's contaminant selection list includes a variety of contaminants, including nitrate, perchlorate, arsenic, and other inorganic and organic contaminants. Although the models contain standard input data sets that automatically populate the models for a few specific contaminants such as those mentioned above, the models also allow users to enter or change contaminant-specific inputs manually so that any contaminant or condition can be modeled. These data can be obtained from any appropriate source. For ease of use, the publicly available USEPA's Drinking Water Treatability Database (USEPA, 2019b) is a source of treatment data that are specifically matched to WBS model inputs.

2.2 | Treatment scenarios

The models also accommodate a range of contaminant removal scenarios. In both the nitrate and perchlorate anion exchange models, for example, the user specifies a resin bed life, in the form of number of bed volumes before regeneration or disposal, which corresponds to a given set of influent conditions (including both target contaminant and competing anion concentrations) and a target treated-water concentration. In the biological treatment model, the user specifies electron donor and nutrient requirements that correspond to influent conditions (including target contaminant concentrations). For nitrate, this article includes information for two hypothetical scenarios in which a small drinking water system (or systems) has an influent nitrate concentration exceeding the current MCL of 10 mg/L as nitrogen (44 mg/L as nitrate). The two scenarios consider two different influent nitrate concentrations: 20 mg/L as nitrogen (90 mg/L as nitrate) and 44 mg/L as nitrogen (195 mg/L as nitrate). For perchlorate, this article includes information for the scenario in which the influent perchlorate concentration is between 10 and 40 µg/L.

2.3 | Model design and structure

All of the WBS models share a common structure, as described by Khera et al. (2013) with the ability to generate cost estimates that include a consistent set of cost elements for treatment equipment, residuals management, building and add-on costs, operations and maintenance (O&M) costs, and indirect costs. The basis of the models' estimates is a central cost spreadsheet, shared by all of the models, that contains unit cost data for each of the cost components. For most of the capital equipment components, the spreadsheet includes a further breakdown for alternative construction materials, because the cost of materials can differ substantially. For example, pipe materials available in the marketplace include stainless steel, carbon steel, polyvinyl chloride (PVC), or chlorinated PVC; however, stainless-steel piping can cost twice as much as PVC. The choice of construction materials also determines each item's useful life, which affects the models' calculations for annualized cost of the treatment system. The unit costs are based primarily on price quotes from vendors, usually with a minimum of prices from at least three vendors, and are frequently updated as described in Khera et al. (2013).

For most of the capital equipment components, the spreadsheet's unit costs vary according to component size or capacity (e.g., 100-, 500-, or 1000-gal tanks). For these components, the spreadsheet includes best-fit cost equations (generated from the average prices using statistical regression analysis across the sizes) that estimate the unit cost of an item of equipment as a function of its size. To protect vendor confidentiality, the public release models do not include individual quotes, only the resulting average prices and component cost equations imported from the central cost spreadsheet.

2.4 | Common model inputs and assumptions

All of the WBS models require user input for certain design parameters. The input parameters are those that tend to vary by contaminant, regulatory scenario, system size, or other considerations. Several of these input parameters (e.g., design and average flow rates) are common across technologies. Khera et al. (2013) describes these common inputs in more detail. Each WBS model also includes a variety of critical design assumptions (e.g., vessel size constraints). User adjustment of these assumptions is optional. Some of the assumptions (e.g., access space between items of equipment) vary for small versus large systems, as discussed in greater detail below.

Table 2 lists the input parameters in the WBS models for the three technologies discussed in this article. It also identifies a few of the many critical design assumptions included in each model. As Table 2 shows, some inputs and critical design assumptions are common to all of the WBS models. Others are specific to the technology under consideration.

3 | MODELING CONSIDERATIONS SPECIFIC TO SMALL SYSTEMS

As discussed above, pursuant to SDWA Section 1412(b)(4) (E)(ii), the agency defines small systems as serving 10,000 people or fewer. In contrast, the WBS models define small systems as those with a capacity (design flow) less than 1 million gallons per day (mgd). The WBS models use this approach to define small systems because the engineering design requirements of the treatment system relate more directly to capacity than to population served.

In many of the WBS models, including those for ion exchange and biological treatment, the general design assumptions used and specific design calculations performed differ for small versus large systems. This is because small drinking water systems often use package plants to accomplish treatment goals. The primary difference between these package plants and the custom-designed plants used by larger systems is that the package treatment systems are pre-assembled in a factory, mounted on a skid, and transported to the site. Package plants can be either partially engineered to meet the treatment requirements of a specific system or available in fixed configurations and sizes. Therefore, the WBS models for technologies that are commonly deployed as package plants use different design parameters for small systems. In most cases, the differing design parameters used in the models for small systems are based on comparison of model outputs with as-built designs and costs for actual small treatment systems.

Both the anion exchange models and the biological treatment model handle package systems by costing all individual equipment line items (e.g., vessels, interconnecting piping and valves, instrumentation, and system controls) in the same manner as custom-engineered systems. This approach is based on vendor practices of partially engineering these types of package plants for specific systems (e.g., selecting vessel size to meet flow and treatment criteria). For small systems with less than 1 mgd, however, the models apply a separate set of design inputs and assumptions. These assumptions are intended to simulate the use of a small package plant. They also include assumptions that reflect the smaller capacity and reduced complexity of the treatment system. These design modifications typically reduce the size and cost of the treatment system. Khera et al. (2013) provide more detail on the specific assumptions that vary for small versus large systems.

4 | CONTAMINANT- AND TECHNOLOGY-SPECIFIC INPUTS AND ASSUMPTIONS

As shown in Table 2, many of the WBS model inputs and assumptions are specific to the contaminant and technology under consideration. Some of these are numeric values (e.g., design flow), whereas others are options that can be selected from a drop-down menu (e.g., resin type). For the purpose of estimating national costs, values for each of these inputs and assumptions would be based on a detailed review of the scientific literature, consultation with treatment technology experts, and consideration of the compliance scenario being evaluated. In many cases, a national estimate would vary these values to develop a range of costs representing the spectrum of compliance options or to reflect uncertainty or variability in the underlying data for a contaminant. Although the model can vary a variety of parameters, such an exercise for all parameters is beyond the scope of this article.

4.1 | Anion exchange

In both anion exchange models, the most crucial input parameters include resin type, empty bed contact time (EBCT), number of bed volumes before regeneration, vessel configuration, and residuals management options for spent regenerant brine or spent resin. For nitrate, options for resin type include nitrate-selective resin and various other standard strong base resins (e.g., polyacrylic gel-type Type I, polystyrenic macroporous Type II). Anion exchange removal of nitrate can be accomplished using standard strong base resins. However, because standard strong base resins prefer sulfate to nitrate, their nitrate capacity can be limited when sulfate is high. Furthermore, if the resins are run past a certain breakthrough point, sulfate can begin to displace the nitrate on the resin, resulting in chromatographic peaking of nitrate (Dimotsis & McGarvey, 1995).

Because of these limitations, resin manufacturers have developed specialized nitrate-selective resins with an increased affinity for nitrate over sulfate. In addition to their greater nitrate capacity in the presence of high sulfate, the literature suggests that these resins are less sensitive to the presence of dissolved organic matter (Song et al., 2013). They also remain effective in the presence of other competing species such as chloride and bicarbonate (Samatya et al., 2006). These advantages can make up for the relatively higher cost of the nitrate-selective resin. Therefore, the most cost-effective choice of resin will depend on both resin cost and water quality parameters.

Standard strong base resins, along with the nitrate-selective resins, discussed above can also remove perchlorate. Resin manufacturers also have developed specialized perchlorate-selective resins. There are significant differences among the different types of resins in terms of their relative affinity for perchlorate (Batista et al., 2003; Boodoo, 2003; Darracq et al., 2014; Tripp et al., 2003). The differences in affinity translate directly to differences in each resin type’s long-term capacity, particularly in the presence of competing anions, and ease of regeneration. The perchlorate-selective resins have the highest affinity and capacity for perchlorate, but are also difficult to regenerate using conventional methods. Therefore, most anion exchange facilities targeting perchlorate have found it cost-effective to take advantage of the high capacity of perchlorate-selective resins by using them on a “throwaway” basis, disposing and replacing the spent resin, instead of regenerating it. This mode of operation is advantageous to small systems because of its simplicity.

The resin type, influent and target contaminant concentrations, and other water quality parameters will influence the choice of vessel size and configuration, and number of bed volumes fed before regeneration (i.e., resin bed life). Vessel configuration options include vessels in parallel or in series. For parallel operation, the model assumes phased operation and therefore separate regeneration. For the purpose of estimating unit treatment costs for a specific contaminant, the choice of resin and values for EBCT, resin bed life, and vessel configuration would be based on a review of the scientific literature and pilot study results representing a range of typical water quality data.

In the anion exchange models, residuals management options for spent regenerant brine include direct discharge to surface water (under an appropriate permit), discharge to a publicly owned treatment works, discharge to a septic system, and discharge to evaporation ponds (including dredging and solids disposal). The model currently does not have the option for off-site non-hazardous disposal, and therefore it could not be considered herein. For spent resin, residuals management options include incineration or landfill disposal. In most cases, the resin can be handled as a non-hazardous waste. The choice of a residuals management option will depend on cost, system size and location, and environmental regulations.

4.2 | Biological treatment

In the biological treatment model, the most crucial input parameters include: design type, electron donor and nutrient requirements, biomass generation, EBCT or hydraulic residence time (HRT), post-treatment options, and residuals management options for spent backwash. Options for design type include gravity-fed open concrete basins with a fixed media bed, pressure vessels with a fixed media bed, and pressure vessels with a fluidized media bed. For fixed bed reactors, influent water passes through a static media bed located in a vessel or concrete basin. Fluidized bed bioreactor designs use vessels where recycled water is pumped at high rates in an up-flow design to fluidize the media bed, allowing for increased mass transport of contaminants, electron donors, and nutrients to the attached biomass, but losing the ability to create specific zones within the fixed bed that may be necessary depending on the contaminant(s) present (oxic versus anoxic).

Electron donor and nutrient requirements and biomass generation during treatment depend on site-specific conditions including raw water characteristics. EBCT (for fixed bed reactors) or HRT (for fluidized bed reactors) must be sufficient to provide the necessary reduction of the target contaminant. Electron donor and nutrient addition must be sufficient to support the biological process. Biomass generation affects the cost of residuals disposal. The model assumes all of the biomass ends up in the residuals, partly in the form of settled holding tank solids and partly as suspended solids in spent backwash. Determining these values typically requires pilot study tests and source water sampling, along with stoichiometric and thermodynamic calculations. The model includes optional inputs for water quality that the model can use to provide theoretical results for electron donor dose and biomass generation. These calculations are based on example site-specific relationships found in the literature

(Harding ESE, 2001) or provided by model peer reviewers. For the purpose of estimating unit treatment costs for a specific contaminant, the choice of input values for electron donor and nutrient requirements, biomass generation, and EBCT or HRT would be based on a review of the scientific literature and pilot study results representing a range of typical water quality data.

Biological treatment will result in the depletion of oxygen and the formation of soluble microbial organic products in the treated water. Therefore, post-treatment in the form of re-oxygenation, followed by mixed media filtration for removal of turbidity, hydrogen sulfide, and/or dissolved organic content typically will be required. To estimate post-treatment cost, the biological treatment model allows the user to select either aeration or hydrogen peroxide addition for re-oxygenation. It also allows the user to select whether or not to include post-treatment (polishing) filtration.

The biological treatment model does not include post-treatment disinfection, because existing facilities may already be present that provide sufficient disinfection. As shown in Table 1, USEPA has separate WBS models under development (e.g., chlorine gas disinfection, hypochlorite disinfection) that will be able to generate costs for this post-treatment step in cases where existing disinfection is insufficient.

In the biological treatment model, residuals management options for spent backwash, from both fixed bed bioreactors and post-treatment filters, include direct discharge to surface water (under an appropriate permit), discharge to a publicly owned treatment works, discharge to evaporation ponds (including dredging and solids disposal), and recycling (including treatment and solids disposal). The choice of a residuals management option depends on system size and location, environmental regulations, and cost-effectiveness.

4.3 | POU/POE treatment

In the POU/POE treatment model, the crucial input parameters include: the specific POU/POE technology used, the water source (i.e., groundwater or surface water), and whether to install ultraviolet (UV) disinfection along with the base POU/POE technology. The POU/POE technologies available in the model depend on the contaminant selected, but can include GAC, cation exchange, and RO. For nitrate and perchlorate treatment, the model allows the user to select POU RO, choosing only between equipment purchase and rental. POU RO devices certified under NSF/ANSI Standard 58 must reduce nitrate/nitrite from challenge level of 30 mg/L as nitrogen (133 mg/L as nitrate) to a target level of 10 mg/L as nitrogen (44 mg/L as nitrate). The same standard requires that certified devices reduce perchlorate from a challenge level of 130 μg/L to a target level of 4 μg/L.

The POU/POE treatment model converts the input design flow into a service population and then into a number of households served, which in turn determines the number of treatment devices required by a system. For instance, over the range of flows evaluated for the POU evaluated herein, 0.03 mgd corresponds to 25 households, and 1 mgd corresponds to 993 households. These calculations use statistical relationships between population and flow that USEPA has developed from survey data collected through the Community Water System Survey (USEPA, 2000). These relationships vary by water source (i.e., groundwater versus

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surface water) and, therefore, the model includes a crucial user input for the source water type.

The POU/POE treatment model includes the option to add UV disinfection devices along with the base POU/POE technology. Under 40 CFR Section 142.62(h) (5), it may be necessary to use such devices and/or conduct heterotrophic plate count monitoring following treatment with POU/POE GAC filters (USEPA, 2006). Because POU RO devices include pre- and post-treatment GAC, disinfection might also be necessary with that technology.

For the example case of nitrate removal, Table 3 shows the values selected for each crucial input parameter each of the WBS models for the two hypothetical influent concentration scenarios examined here. Table 4 shows the values selected for each model for the example scenario where perchlorate is the target contaminant, instead of nitrate. The example cost outputs described in the following sections use these input values.

5 | WBS MODEL OUTPUTS

The output sheet of each WBS model lists the size and quantity required for each item of equipment included in the design along with the corresponding unit cost from the central WBS spreadsheet. The output sheet multiplies unit cost by quantity to determine the total cost for each WBS component.

Many of components are available in optional materials, all of which are illustrated on the output worksheet. For example, pressure vessels can be constructed with different types of body material (stainless steel or carbon steel) and different types of internal material (stainless steel or plastic). When optional materials are available, the model selects from among them based on the chosen inputs for component level and system size. The direct capital cost is the sum of these selected component costs. The output sheet also contains sections that report addon costs, indirect capital costs, annual O&M costs, and total annualized cost. Table 5 identifies the components included in each of these cost elements.

As described in Khera et al. (2013), the model derives annualized cost from the useful life estimates for the individual components, taken from the WBS cost spreadsheet, and it uses these estimates to calculate an average useful life for the entire system. The calculation uses a reciprocal weighted average approach. The output sheet uses this average useful life calculation for the system, along with a discount rate, to annualize total capital cost, resulting in capital cost expressed in dollars per year. The models use a default discount rate of 7%, which users can adjust directly on the output sheet. The output sheet adds annual O&M cost to the annualized capital cost to arrive at a total annual cost in dollars per year.

5.1 | Accuracy of cost estimates

During peer review of the anion exchange model, one peer reviewer responded that resulting cost estimates were in the range of budget estimates (+30% to −15%). The other two reviewers thought anion exchange estimates were order of magnitude estimates (+50% to −30%), with an emphasis on the estimates being high.

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During peer review of the biological treatment model, one reviewer thought the model underestimated O&M costs by 20% to 30% (which would be in the range of an order of magnitude estimate), but overestimated capital costs by about 25% (which would be in the range of a budget estimate). A second reviewer responded that direct capital costs were at the fringes of a budget estimate (+30% to −15%), while total capital costs were in the order of magnitude range (+50% to −30%) or possibly even better, in the budget estimate range. This reviewer's conclusions about total capital costs were based on a comparison with preliminary costs for a plant currently under construction, for which the model underestimated costs. The final reviewer responded that costs were budget estimates (+30% to −15%).

During peer review of the POU/POE model, reviewers felt that the default assumptions may tend to overstate "out-of-pocket" costs to systems because very small systems could use volunteers to perform some tasks. While this may be true, USEPA's model is designed to estimate the opportunity costs for a successful POU/POE program that is consistent with USEPA guidance (USEPA, 2006), which does not include volunteers.

6 | NITRATE TREATMENT COSTS

6.1 | Example cost results

Each WBS model generates the detailed cost breakdown, total capital cost, annual O&M cost, and total annualized cost for a single set of inputs. That is, the results correspond to a single system size. Figures 1 and 2 show example costs across varying system sizes for the two input sets in Table 3 (influent concentrations of 90 and 195 mg/L as nitrate in Figures 1 and 2, respectively). The figures show the annualized costs for the three technologies (anion exchange, biological treatment, and POU RO) used for nitrate removal, generated using the low-cost setting in each model's input for component level. The figures also reflect the other assumptions inherent in the inputs selected in Table 3. For example, they assume UV disinfection is not added along with the POU RO devices, and that post-treatment disinfection is not added along with biological treatment.

The results shown in the figures represent multiple runs of the WBS models across a range of system sizes that correspond to the WBS definition of a small system—specifically, 17 flow rates ranging from 0.03 to 0.999 mgd. The figures also show results for systems that the WBS models categorize as medium-sized systems—specifically, 15 more flow rates ranging from 1 to 9.999 mgd. Some systems in this size range would meet certain SDWA definitions of a small system. For anion exchange and biological treatment, the sudden increase in estimated costs at a design flow of 1 mgd reflects the change in model design assumptions and calculations shown in Khera et al. (2013) corresponding to the shift from packaged to custom-design systems. The figures do not show results for POU RO treatment beyond a design flow of 1 mgd because it is assumed that only very small systems would use POU programs.

6.2 | Comparing technology costs

For the hypothetical scenarios modeled, the results in Figures 1 and 2 show that POU RO is the lowest cost option for the very smallest systems. Because the POU technology is relatively insensitive to changes in influent nitrate, the costs for POU treatment do not change between the two influent concentration scenarios. In comparison, centralized anion exchange costs are highly sensitive to regeneration frequency and, therefore, influent nitrate concentration. Thus, POU treatment remains cost-competitive with centralized anion exchange for slightly larger sizes in the higher influent nitrate scenario. Specifically, for the lower influent nitrate scenario (Figure 1), centralized anion exchange becomes more cost-effective than POU treatment at a design flow of approximately 0.12 mgd (about 112 households). For the higher influent nitrate scenario (Figure 2), centralized anion exchange becomes more cost-effective at a design flow of approximately 0.16 mgd (about 145 households).

In the lower influent nitrate scenario (Figure 1), anion exchange remains the most cost-effective option from a design flow of about 0.12 mgd up to the maximum size considered here (design flow of 10 mgd). Anoxic biological treatment begins to “close the gap” for the largest sizes, but has higher costs throughout the range in the lower influent nitrate scenario. Although biological treatment costs are somewhat sensitive to influent nitrate (because of increasing electron donor requirements and increasing biomass generation), this sensitivity is not nearly as dramatic as for anion exchange. Therefore, in the higher influent nitrate scenario (Figure 2), biological treatment becomes the most cost-effective option at a design flow of approximately 3.5 mgd and above.

A key difference between the two centralized treatment technologies is that anoxic biological treatment is more capital cost intensive than anion exchange because of the larger reactor vessels (10 min HRT for fluidized bed biological treatment versus 2 to 3 min EBCT for anion exchange) and additional post-treatment equipment requirements. Anion exchange, conversely, is more operating cost intensive, with regenerant brine consumption and discharge requirements that increase linearly with flow rate. This difference explains why biological treatment becomes more cost-competitive with increasing system size. It also means that, in comparing costs on an annualized basis, the discount rate (i.e., cost of capital) becomes a significant factor. To illustrate this point, Figure 3 shows results for the higher influent nitrate scenario assuming a 3% discount rate instead of the 7% WBS model default. With the lower discount rate, biological treatment becomes more cost-effective than anion exchange above a design flow of approximately 2 mgd instead of 3.5 mgd. Note that, for either discount rate, these results are specific to the inputs used in this analysis (Table 3). Differing water quality conditions, such as the presence of co-contaminants or competing ions, could change the comparative costs of the technologies. Also, in the case of nitrate, the availability of a low-cost option, such as discharge to a publicly owned treatment works (as assumed in this analysis), for brine management is key to the cost competitiveness of ion exchange.

Figures 1 through 3 show biological treatment costs assuming a fluidized bed design. However, fixed bed biological treatment is potentially lower cost because it does not require recycle pumping to fluidize the media bed. For comparison sake, Figure 3 also shows model results for fixed bed treatment using pressure vessels. Fixed bed costs are, as expected, somewhat lower than fluidized bed costs, due to avoiding the capital and operating costs of recycle pumping. The cost of recycle pumping, however, increases more slowly with system size than other costs. Thus, as size increases, the cost saving becomes less significant relative to total cost. As a result, the point at which biological treatment becomes more cost-effective than anion exchange is not substantially different between fixed and fluidized bed designs. Given this, either of the two biological approaches could be a practical solution for a larger utility considering such a system.

7 | PERCHLORATE TREATMENT COSTS

7.1 | Example cost results

Figure 4 shows example costs for perchlorate across varying system sizes for the inputs in Table 4. The figure shows the annualized costs, assuming a 7% discount rate, for the three technologies (anion exchange, biological treatment, and POU RO) used for perchlorate removal. As with the nitrate results, the results for perchlorate shown in Figure 4 represent multiple runs of the WBS models across a range of system sizes, generated using the low-cost setting in each model's input for component level. Also similar to the nitrate results, there is an increase in estimated costs at a design flow of 1 mgd for anion exchange and biological treatment reflecting the shift from packaged to custom-design systems. Figure 4 does not show results for POU RO treatment beyond a design flow of 1 mgd because it is assumed that only very small systems would use POU programs. These results are specific to the inputs identified in Table 4.

Available data suggest that perchlorate-selective resin retains a high capacity across the range of influent concentrations typically observed in drinking water sources (Blute et al., 2006; Drago & Leserman, 2011; Russell et al., 2008; Wu & Blute, 2010). Like other resins, however, the bed life of perchlorate-selective resin can be sensitive to influent concentrations of competing anions such as nitrate and sulfate. Therefore, Figure 4 shows anion exchange costs given two different assumptions about resin bed life: 170,000 bed volumes and 250,000 bed volumes.

7.2 | Comparing technology costs

Similar to the results for nitrate, for the hypothetical scenario modeled for perchlorate, the results in Figure 4 show that POU RO is the lowest cost option for the very smallest systems. In the case of perchlorate treatment, centralized anion exchange becomes more cost-effective than POU treatment at a design flow of approximately 0.08 mgd (about 73 households). The differing assumptions about anion exchange bed life examined here do not significantly change this result.

For perchlorate, anion exchange remains the most cost-effective option from a design flow of about 0.08 mgd up to the maximum size considered here (design flow of 10 mgd). Anoxic biological treatment begins to "close the gap" for the largest sizes, but does so more slowly than in the case of nitrate. This is because anion exchange has lower annualized costs in the perchlorate scenario examined here than in the nitrate scenarios discussed above. Capital costs are reduced because equipment to accomplish regeneration (e.g., brine storage tanks, piping, valves) is not required. Although disposal and replacement of the single-use resin is expensive, the long resin bed life means that, on an annual basis, O&M costs are also lower for perchlorate anion exchange treatment than for nitrate anion exchange treatment.

Anoxic biological treatment, on the other hand, exhibits higher capital costs for perchlorate than for nitrate due to the longer HRT (12 min instead of 10 min) with slightly lower operating costs due to the lower electron donor and nutrient dosages and lower biomass generation. Overall, annualized costs are slightly higher for biological treatment in the perchlorate scenario examined here compared to the nitrate scenarios discussed above. Therefore, for perchlorate, anion exchange remains the least cost option of the two centralized treatment technologies throughout the range of system sizes examined here. As shown in Figure 4, this result does not change regardless of whether fixed or fluidized bed biological treatment is considered. It also does not change if a discount rate of 3% is considered (results not shown here). It should be noted that these results are specific to the inputs used in this analysis (Table 4). The comparative costs of the technologies would differ under different water quality conditions, such as high-concentration site cleanup situations. Furthermore, anoxic biotreatment is relatively new in the United States and certain cost savings may be realized if the process becomes more commonly used.

8 | CONCLUSIONS

The USEPA has continued its efforts to develop a suite of WBS models for estimating the costs of drinking water treatment and make the models publicly available. These models include specific design assumptions and calculations that have specific applicability to costing treatment for small water utilities. These WBS models were helpful in evaluating three technologies frequently considered for nitrate and perchlorate removal: anion exchange treatment, anoxic biological treatment, and POU RO treatment.

For nitrate, the analysis here shows that anoxic biological treatment can be cost-effective compared to anion exchange treatment, at least for certain combinations of influent nitrate concentration and system flow rates. Anion exchange treatment is shown to be the least-cost option for a broad range of influent concentrations and flow rates for both nitrate and perchlorate. Finally, POU treatment is the least-cost option for both nitrate and perchlorate for very low design flow rates.

As noted above, the results presented here are specific to the inputs used in this effort. USEPA’s WBS models, however, provide a tool that can be used to compare technology costs for additional inputs such as different technology options, water qualities, and other site-specific considerations. For anion exchange, these parameters include regeneration or disposal frequency and the availability of discharge options for spent regenerant, if regeneration is used. For anoxic biological treatment, they include electron donor and nutrient requirements, along with post-treatment needs.

In addition to costs, community-specific needs and downstream impacts of the treatment processes on distribution systems will also need to be evaluated and can be a decision point for a utility. Finally, it is acknowledged that anoxic biological treatment is a relatively new process to be implemented in the drinking water field, compared to anion exchange resins and high-pressure POU membranes. Therefore, as additional research on, and implementation of, anoxic biological treatment processes is completed, there will likely be cost savings that are realized, thereby lowering the total cost of treatment for various influent scenarios and design considerations while providing a system that can be robust, reliable, and sustainable. The finding here that anoxic biological treatment can be cost-competitive even absent these advances supports consideration of biological treatment as a viable process for perchlorate removal.

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Article Impact Statement

Small systems that struggle with nitrate and perchlorate contamination will benefit from this evaluation of treatment costs.

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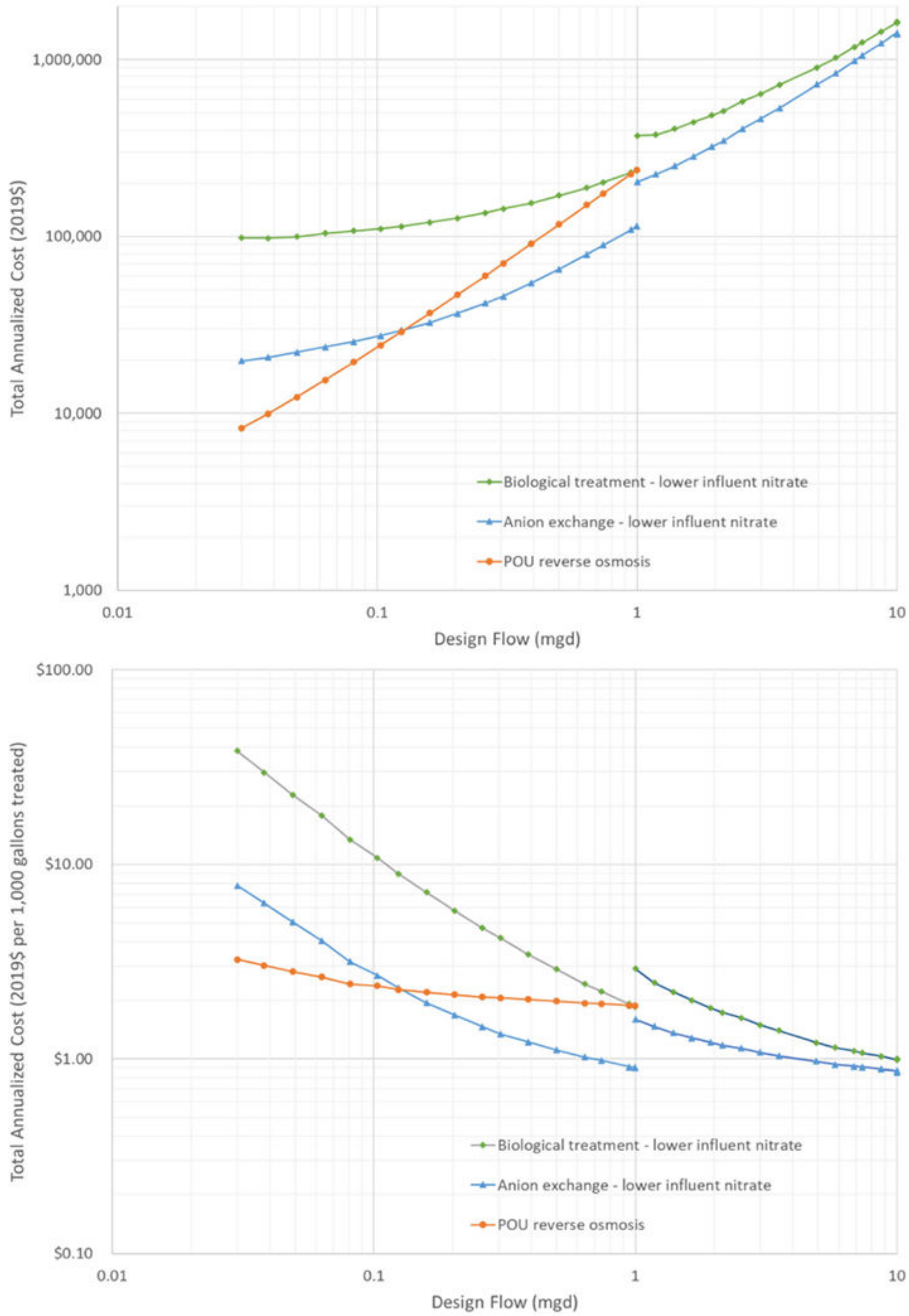

FIGURE 1.
Nitrate treatment costs at 90 mg/L influent nitrate (20 mg/L as nitrogen), 7% discount rate. Abbreviation: POU, point-of-use

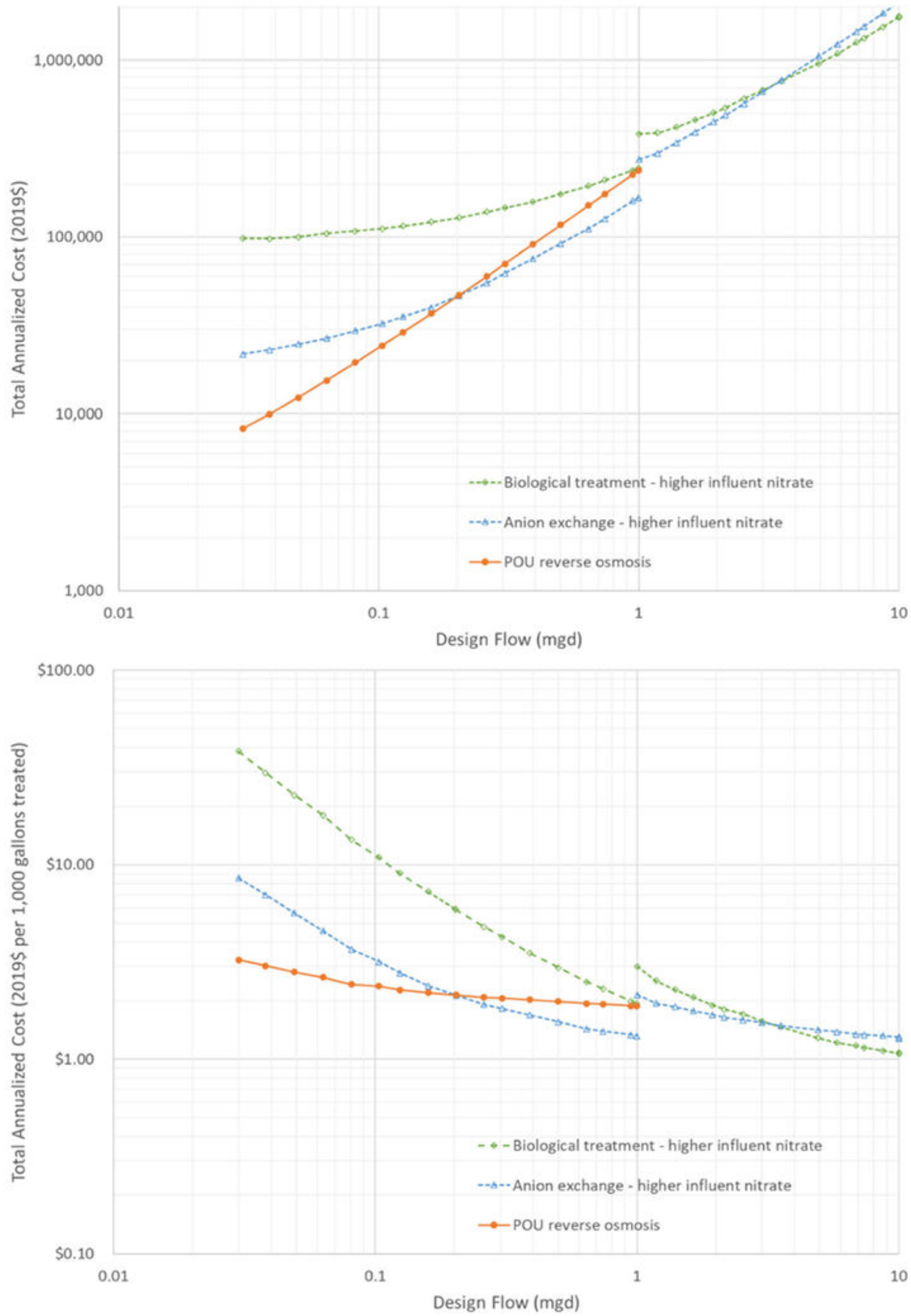

FIGURE 2.

Nitrate treatment costs at 195 mg/L influent nitrate (44 mg/L as nitrogen), 7% discount rate. Abbreviation: POU, point-of-use

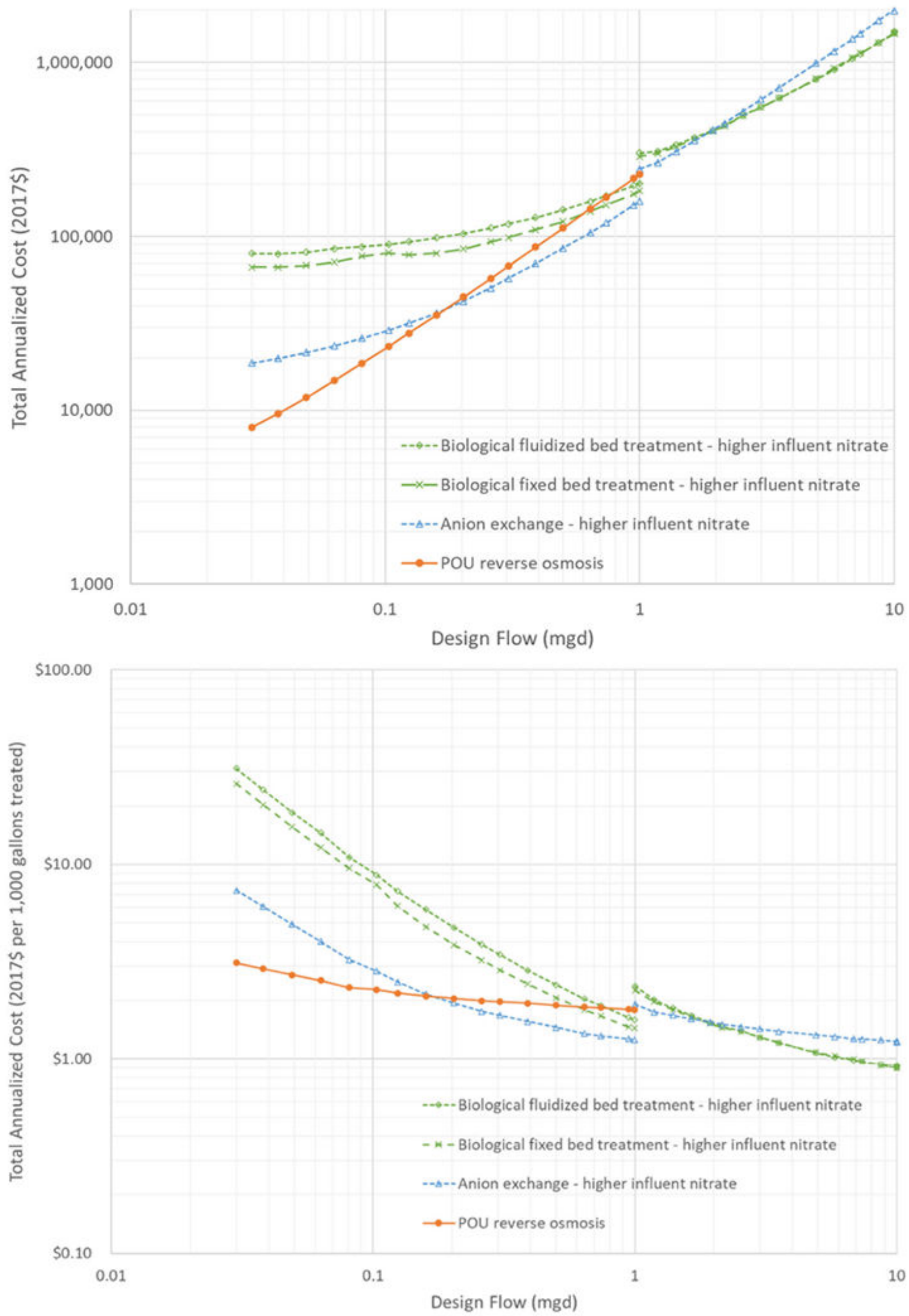

FIGURE 3.

Nitrate treatment costs at 195 mg/L influent nitrate including fixed bed biological treatment, 3% discount rate. Abbreviation: POU, point-of-use

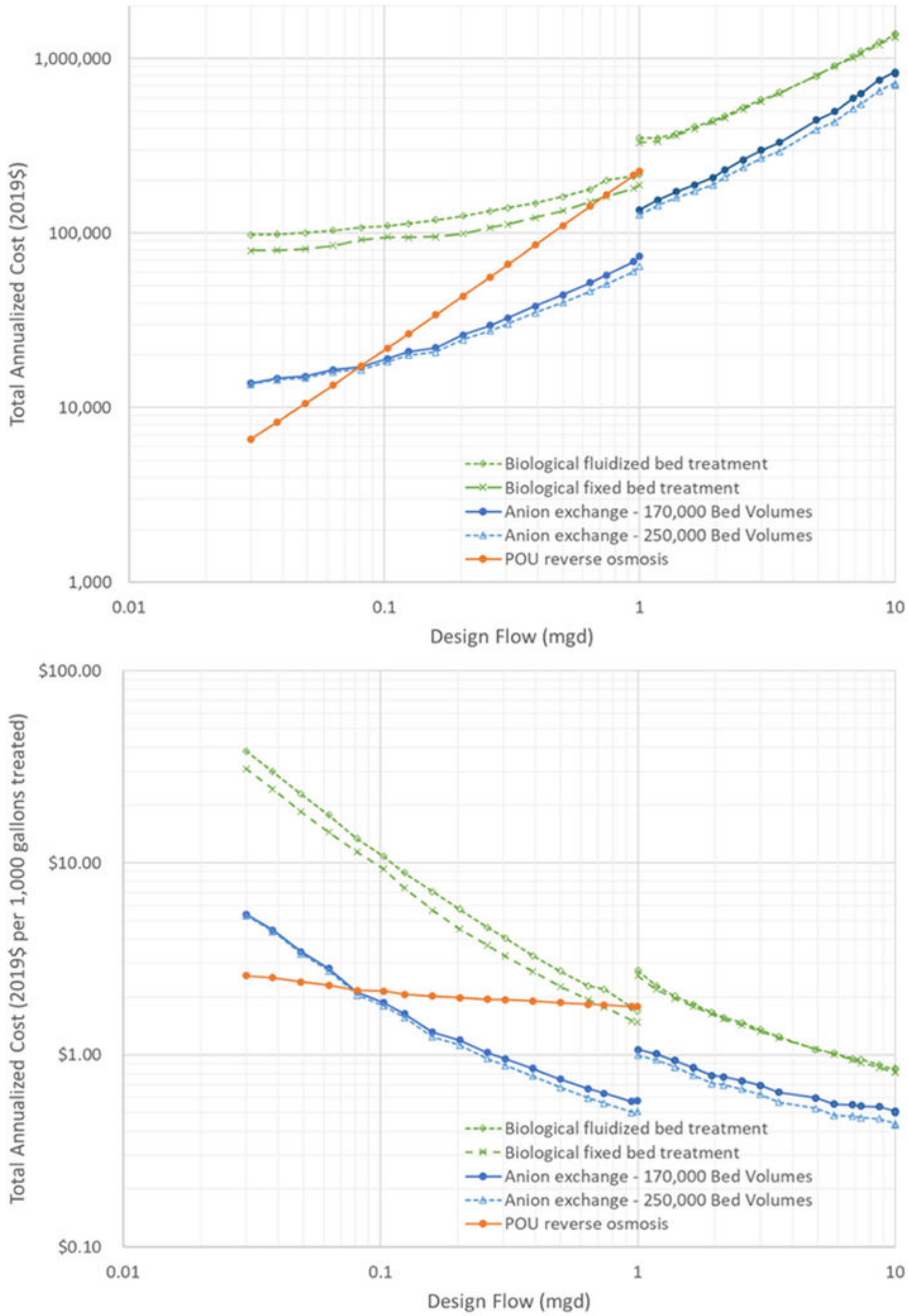

FIGURE 4.
Perchlorate treatment costs, 7% discount rate. Abbreviation: POU, point-of-use

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TABLE 1

Technologies included in the work breakdown structure model suite

Primary technologies	**Add-on technologies**
Available to the public	Available to the public
• Granular activated carbon[a]	• Caustic addition[c]
• Packed tower aeration[a]	• Orthophosphate addition[c]
• Multi-stage bubble aeration[a]	
• Anion exchange[b]	Under development
• Anion exchange for perchlorate[b]	• Powdered activated carbon addition
• Biological treatment (oxic and anoxic)[b]	• Chlorine gas pre-oxidation
• Point-of-use/point-of-entry (POU/POE) technologies[b,d]	• Chlorine dioxide pre-oxidation
• Cation exchange[c]	• Hypochlorite pre-oxidation
• Non-treatment option[c,e]	
Under development	• Ozone pre-oxidation
• Adsorptive media[f]	• Ammonia addition for chloramination
• Greensand filtration	
• Low-pressure membrane filtration[g]	• Permanganate addition
• Nanofiltration and reverse osmosis	• Lime addition
• Electrodialysis reversal	• Coagulant addition
• Diffuse aeration	• Clearwell storage
• Tray aeration	• Acid addition
• Spray aeration	
• Chlorine gas disinfection	
• Chlorine dioxide disinfection	
• Hypochlorite disinfection	
• Ozone disinfection	
• Ultraviolet (UV) disinfection	
• Advanced oxidation with UV	
• Conventional filtration and direct filtration	
• Lime softening	
• Dissolved air flotation	
• Magnetic ion exchange	

[a] See Khera et al. (2013).

[b] Described in this article.

[c] Also newly available, but not described here.

[d] Including granular activated carbon, cation exchange, and reverse osmosis POU/POE devices.

[e] Including new wells and interconnection.

[f]Including activated alumina and several proprietary arsenic removal media.

[g]Including pressure-driven microfiltration, pressure-driven ultrafiltration, and vacuum-driven immersed membrane systems.

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TABLE 2

Work breakdown structure model inputs and critical design assumptions

Technology	Inputs	Example critical design assumptions
Common across technologies	• Contaminant • Design and average flow • Component level (not a required input for POU/POE treatment) • System automation (not a required input for POU/POE treatment)	• Indirect cost percentages • Average household size
Anion exchange	• Resin type • Influent contaminant concentration • Bed volumes before regeneration or disposal • Number of vessels in series (i.e., parallel or series operation) • Empty bed contact time • Bed depth and vessel dimensions • Brine delivery method • Residuals management options (for spent brine and spent resin) • Number of booster pumps • Number of redundant vessels • Treated water corrosion control • Backwash pumping design • Backwash storage design	• Vessel design constraints (e.g., minimum and maximum surface loading rate, maximum vessel diameter) • Bed expansion during backwash and regeneration • Backwash assumptions (e.g., backwash loading rate, backwash duration) • Regeneration assumptions (e.g., brine concentration and loading rate, slow rinse and fast rinse volumes) • Annual resin loss • Number of valves and instruments per vessel • …more than 100 additional critical design assumptions
Biological treatment	• Design type (fixed bed pressure, fixed bed gravity, or fluidized bed) • Electron donor type and dose • Biomass generation • Influent concentrations • Nutrient requirements • Empty bed contact time or hydraulic residence time • Bed depth and bioreactor dimensions • Post-treatment options • Interval between backwashes and inclusion of air scour with backwash • Residuals management options (for spent backwash) • Number of booster pumps • Number of redundant units • Backwash pumping design • Backwash storage design	• Surface loading rate constraints • Fluidized bed expansion and freeboard • Fixed bed bioreactor backwash assumptions (e.g., backwash loading rate, backwash duration) • Post-treatment filter backwash assumptions • Media attrition • …more than 100 additional critical design assumptions
POU/POE treatment	• Treatment technology • Water source (groundwater or surface water) • Inclusion of UV disinfection	• Installation labor (including scheduling) • Public education requirements (materials and labor) • Equipment replacement schedules • … more than 50 additional critical design assumptions

Abbreviations: O&M, operations and maintenance; POU/POE, point-of-use/point-of entry; UV, ultraviolet.

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TABLE 3

Example values for key work breakdown structure model inputs for nitrate removal

		Value selected to generate example costs	
Technology	**Input**	**Influent concentration 90 mg/L as nitrate**	**Influent concentration 195 mg/L as nitrate**
Anion exchange	Resin type	Nitrate-selective	Nitrate-selective
	Number of bed volumes before regeneration	420	260
	Vessel configuration	Parallel	Parallel
	Empty bed contact time	2 min	3 min
	Discharge option for spent brine	Publicly owned treatment works	Publicly owned treatment works
Biological treatment	Design type	Fluidized bed	Fluidized bed
	Electron donor requirements	37.4 mg/L acetic acid	49.6 mg/L acetic acid
	Biomass generation	15.3 mg/L	19.0 mg/L
	Nutrient requirements	2 mg/L phosphoric acid (as phosphorus)	2 mg/L phosphoric acid (as phosphorus)
	Hydraulic residence time	10 min	10 min
	Post-treatment options	Aeration and polishing filter	Aeration and polishing filter
	Discharge option for spent backwash	Recycle	Recycle
POU/POE treatment	Technology	POU reverse osmosis (purchased)	POU reverse osmosis (purchased)
	Water source	Groundwater	Groundwater
	Include UV disinfection	No	No

Abbreviations: POU, point-of-use; POU/POE, point-of-use/point-of entry; UV, ultraviolet.

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TABLE 4

Example values for key work breakdown structure model inputs for perchlorate removal

Technology	Input	Value selected to generate example costs
Anion exchange	Resin type	Perchlorate-selective
	Number of bed volumes before disposal	170,000 to 250,000
	Vessel configuration	2 vessels in series
	Empty bed contact time	3 min total for both vessels
	Disposal option for spent resin	Non-hazardous waste disposal by incineration
Biological treatment	Design type	Fluidized bed
	Electron donor requirements	10 mg/L acetic acid
	Biomass generation	8 mg/L
	Nutrient requirements	1 mg/L phosphoric acid (as phosphorus)
	Hydraulic residence time	12 min
	Post-treatment options	Aeration and polishing filter
	Discharge option for spent backwash	Recycle
POU/POE treatment	Technology	POU reverse osmosis (purchased)
	Water source	Groundwater
	Include UV disinfection	No

Abbreviations: POU, point-of-use; POU/POE, point-of-use/point-of entry; UV, ultraviolet.

TABLE 5

Cost elements included in all work breakdown structure models

Cost category	Components included	
Total capital costs		
Direct capital costs	• Technology-specific equipment (e.g., vessels, basins, pumps, blowers, treatment media, piping, valves)	• Instrumentation and system controls • Buildings • Residuals management equipment
Add-on costs	• Land • Permits	• Pilot testing
Indirect capital costs	• Mobilization and demobilization • Architectural fees for treatment building • Equipment delivery, equipment installation, and contractor's overhead and profit • Sitework • Yard piping • Geotechnical • Standby power	• Electrical infrastructure • Process engineering • Contingency • Miscellaneous allowance • Legal, fiscal, and administrative • Sales tax • Financing during construction • Construction management and general contractor overhead
• Operations and maintenance costs		
Labor	• Operator labor for operation and maintenance of process equipment • Operator labor for building maintenance • Managerial and clerical labor	• Operator labor for other technology-specific tasks (e.g., managing regeneration, backwash, or media replacement)
Materials	• Materials for maintenance of booster or influent pumps • Materials for building maintenance • Materials for maintenance and operation of technology-specific equipment	• Replacement of technology-specific consumables (e.g., chemicals) or other frequently replaced items (e.g., treatment media)
Energy	• Energy for operation of booster or influent pumps • Energy for lighting, ventilation, cooling, and heating	• Energy for operation of technology-specific items of equipment (e.g., blowers, mixers)
Residuals	• Residuals management operator labor, materials, and energy	• Residuals discharge and disposal costs (including transportation)