Cost assessment of Produced Water Treatment

John Hackney and Mark R. Wiesner

May 1996

INTRODUCTION

This work investigates treatment of waste or "produced" waters generated during the exploration for and the production of crude oil and natural gas. This investigation includes produced water characterization with respect to quality and quantity, and an assessment of the technical and the economic feasibility of using a wide range of technologies to achieve various levels of treated water quality. The quality of the produced water is judged by quantifying the amounts of material present in categories of produced water contaminants. These categories are adsorbable organic materials, volatile organic compounds (VOC's), particulate and emulsified materials, and dissolved inorganic constituents. The chemical analysis of approximately 120 produced waters has been assembled into a database. The list of produced water constituents from the database that are considered in order to characterize the individual waters can be found in Table 1. This list of constituents was taken from the API study (API, 1987) of oil and gas exploration and production wastes. A basic statistical analysis of the produced water database was completed as part of this analysis.

Processes that potentially could be used to remove undesirable produced water constituents have been evaluated. Several processes were selected and further evaluated in order to determine produced water treatment efficiency. Treatment scenarios for the waters describing the necessity and order of these processes have been created, including the requisite pretreatment of process influent streams and treatment process effluent requirements.

1

Finally, the cost of treating the produced water with a given treatment train has been estimated using the combination of a variety of cost models, cost indexes, and strategies.

The methods used to accompish the three goals mentioned above will be discussed in Section 1. In Section 2 the estimated costs of produced water treatment over a wide range of produced and product water quality are reported.

SECTION 1:   METHODS

1.1 COSTING PROCEDURES

The minimum cost of treating produced water is the cost of simply disposing of the water. This is most frequently accomplished by deep well injection, ocean discharge, and/or hauling. Some pretreatment, particularly before deep well injection, is likely to be needed to maintain well injectability and minimize well maintenance costs. Typical values given for produced water disposal range from $0.63 to $3.15 / m^3 (Tomson, M.B., Oddo, J.E., Kan, A.T., 1992). When more extensive pretreatment is required before disposal or when the produced water is to be used, the cost of produced water treatment includes the capital and operating costs of unit processes applied to the waste stream. Irrigation is one potential use of produced water that has been adequately treated. Various levels of treated water quality will be investigated here and the potential for the use of the treated produced water is vast. The capital and operating costs vary over time in response to changing prices for anyt consumable product used during the produced water treatment. Cost functions must account for these time variable aspects of cost as well as relating costs to the design and operating variables for each unit process. The prices change over time for various reasons (e.g. inflation, market trends). Cost data from different times are updated using cost indexes. Indexes for broad categories of items or for

specific items are available and can be used to update a group of associated costs. The exact procedure for updating costs that was used in this work is described in (Qasim, S. et al., 1992). The cost components for construction costs and operation and maintenance costs that are described by Qasim, S. et al.(1992) are also used by Gumerman et al., (1979). These components are easily updated, using the corresponding indexes. Gumerman et al., (1979) developed cost curves for the U.S. Environmentasl Protection Agency (EPA) for processes commonly used to remove contaminants listed in the National Interim Primary Drinking Water Regulations from water. These curves describe the construction costs that were formulated from conceptual designs of unit processes and from projected operation and maintenance costs associated with labor, maintenance materials, and energy requirements for the processes. "Capital costs" refers to the investment required to construct and begin the operation of the plant, principally materials, labor, and interest. Operation and maintenance costs include the costs associated with the labor, material, and energy required to operate and maintain the treatment plant. A significant residual stream is not generated during all forms of water treatment, and therefore, in some instances, disposal costs may be small. Also, the residual waste stream generated during the excecution of a water treatment unit process will not always have a cost associated with its' disposal that is directly imputable to the disposer. Cost estimates in this work are limited to these items directly included in the construction and operation of the water treatment facility. Capital costs are amortized over the useful life of the facility. The capital recovery factor (CRF), used to spread out a capital cost over a given number of years at a specific interest rate, is defined as

$$CRF = \frac{I(1+I)^N}{[(I+1)^N - 1]}$$

where I is the interest rate and N is the number of years over which the cost will be spread. All capital costs considered in this work will be spread over a period of 20 years

at a 10 percent annual rate of intest (Adams, .J.Q. and Clark, R.M., 1989). All costs, in general will be presented per unit (m^3) of produced water.

1.2 LIQUID/SOLID SEPARATION

One of the four categories of treatment for the produced waters considered in this work is liquid/solid separation. Processes for liquid/solid separation remove particulate material and associated contaminants from produced water. In particular, heavy metals are often sorbed to the suspended solids in the produced waters and will thus be removed along with these solids. Also, emulsified hydrocarbons may be removed. Package or "off the shelf" technologies are likely to be most appropriate for low flow rates encountered in produced water treatment. These package plants are standard, commercial technologies, assumed in this work to consist of unit processes for chemical addition, initial mixing, flocculation, settling, and dual media (or diatomaceous earth) filters. The cost of these units typically scale as a function of the treated water flow. Package treatment plant costs are made up of three main components: capital costs, operation and maintenance costs, and the residual waste stream management costs of the plant (usually treated as additional operating costs). Additional capital investments for residual disposal may also be required. The costs provided by Gumerman et al., (1979) are used to estimate both the capital and operations and maintenance costs for the package plant treatment. Volume 1 of the 4 volumes written by Gumerman et al., (1979) contains the package plant treatment costs, based on the design rate of waste stream flow through the system. The costs of building and managing the package plants that would be neccesary to treat the flow regimes relevant for this work are listed in the tables of costs provided in this volume. The chosen rates of flow that are most representative of the flow regimes observed in oil fields were divided by a sizing factor of 0.7 in this instance. This sizing

4

factor provides extra capacity to deal with circumstances such as variations in the produced water flow rate. The produced water flows, after correction by the sizing factor, were approximately equal to 0.0144, 0.144, and 1.44 MGD (54.50, 545.04, and 5450.4 m^3/d). These flow rates are common rates of produced water flow for small, medium, and large oil fields. The cost of each of the plants that could treat produced waters flowing at the smallest two of these rates are listed in Tables 6 and 7 of the first volume of the Gumerman et al., (1979) study. The cost of the package plant that would be neccesary to treat the largest rate of produced water flow listed above was obtained by multiplying the cost for building and operating the maximum capacity plant evaluated within Tables 6 and 7 (4088 m^3/d) by a factor of 1.33. All of these costs were then calculated using the most detailed of the standard updating procedures that will be explained in a later section. The capital costs for this type of treatment plant as well as all other capital costs referred to in this work are assumed debt which is amortized by multiplying it by the CRF, discussed earlier, in section 1.1.

In liquid/solid seperation, the principal residual stream is that associated with sludges from settling, backwash waters, and powdered activated carbon (PAC) addition. The sludge handling and disposal costs typically include the costs of thickening, drying and land filling the sludge. In some cases, some or all of these handling and disposal processes may be combined or eliminated. Cost effectiveness (lowest average cost) is the criterion on which the decisions affecting the handling and disposal of the sludges are made. Sludge handling configurations and the estimated costs associated with them are used for sludges other than those generated by conventional treatment (e.g. spent PAC). The method of analysis used to evaluate the handling of sludge is described in section 1.3

The costs associated with the chemicals added for the purpose of particle destabalization and flocculation were calculated by multiplying a coagulant dosage of alum or ferric

sulfate at 200 and 60 mg/L, respectively.by the unit cost for each chemical. For systems smaller than 3785 m^3/d (1 MGD), the prices chosen for a ton of alum and for a ton of ferric sulfate were 500 and 200 dollars, respectively. These prices dropped to 250 and 155 dollars per ton for systems treating waste streams processing a rate of flow greater than 3785 m^3/d (1 MGD). The cost of feeding the chemicals into the waste stream, including the capital, and the operation and maintenance costs of the chemical feed systems, is incorporated in the estimate of the package treatment plant cost. Coagulant doses were selected based on the typical range used in waste water treatment. The assumption is that these average conditions will result in adequate removal of suspended solids and some of the associated contaminants found in produced waters.

Tables 2, 3, and 4 describe all of the components that contribute to the total average cost for the treatment of produced waters to remove particulates and emulsified materials. Capital, operating, and residuals disposal costs were added to the chemical costs and this sum was divided by the approprate rate of flow. This procedure yields an average cost per volume of treated produced water.

1.3 SLUDGE DISPOSAL

Sludge is generated by several of the unit processes evaluated in this work. PAC, coagulant addition, solids and oils in the produced water are all sources of sludge in liquid/solid seperation. The total suspended solids (TSS) removed from the waste streams and the coagulants added to the waste streams during package plant treatment will increase the amount of sludge that is produced for each stream as will the amount of PAC added (although PAC contribution is small).

Equations describing the amount of sludge that is produced are listed in Table 2. Amounts of sludge produced from the backwash water from the granular activated carbon (GAC) contactor are considered to be negligible. Post filter GAC contactors may not even have to be backwashed on a regular basis and the filter(s) of the package plant are assumed to be a polishing step. The sludge that is produced and considered here must be disposed of and should be as dense and dry as possible (Kos, P., 1977). The costs associated with sludge disposal are a function of the amount of sludge (volume) and the percentage of water remaining. This last variable used in the cost equations is a result of the fact that all sludges must pass a specific test for dryness before being allowed in a landfill, which is the assumed method of sludge disposal. The cost of the landfilling of sludge is based on the amount of sludge to be abated. This rate per volume is generally more than tripled should the sludge not pass the dryness test.

Gravity thickening is the first unit process used to reduce the sludge volume. The equations used to size the thickeners is also presented in Table 2. Only the waste streams with abnormally high TSS (>1000mg/l) will require sludge thickening below a flow rate of 3785 $m^{3/}d$ (1MGD). The factor which will dictate whether or not a thickener is needed is the required diameter of the thickener. It is not usually economically feasible to build a thickener with a diameter of less than 6 meters. Similarly the sand drying beds must have a minimum area. If the flux rate of sludge entering the thickeners or the drying beds is too great (and would make the diameter or area of the thickener too large), multiple units are constructed. If the sludge flow rate is too small, the processes are not built at all. In some cases, a process may be sized above that size dictated by a rate of sludge flux because the most cost effective method of disposing of the sludge calls for it. This occurs, for example, whenever the construction of the minimum sized sand drying bed would cost less per unit of treated water than the increased landfill cost for undried sludge, even though the drying bed is too large for the amount of sludge produced.

Gumerman et al., (1979) developed the equations used to estimate the costs associated with : gravity thickening and the sand drying bed processes. These components, added together are then combined with the various landfill costs which are a function of both sludge type and amount. PAC sludge, for instance, is regularly landfilled at a cost that is six times that of primary sludge and in some instances is much more expensive than that. The amount of sludge produced by the PAC process is relatively small compared to the other generators of sludge. The sum of these cost components is easily converted to a unit cost per volume of produced water treated.

1.4 VOLATILE ORGANIC CONTAMINANT REMOVAL by PACKED TOWER AERATION.

PTA

Packed tower aeration (PTA) or air stripping was evaluated for removal of volatile organic compounds. Organic contaminants that have Henry's constants greater than 1.93E-4 (atm/m^3/mol) are considered volatile and can therefore be stripped from the produced water if they are brought into contact with an adequate amount of air. This contaminant stripping occurs inside of packed towers. These towers are filled with randomly packed inert material such as Raschig rings. Costs for the packed towers are taken from the work of Gumerman et al., (1979) and consist of capital and operation and maintenance components. The equations describing cost functions used to estimate these costs are listed in Table 4. These costs are a function of tower volume. This volume is determined by a tower design strategy described by Kavanaugh and Trussell (1980). The diameter and height of these towers, as well as the air:water ratio necessary to strip the contaminants, are a function of the Henry's constant for the least strippable organic contaminant present in the waste stream. The contaminant matrix of the produced waters

will have no discernable effect on the removal of the different volatile organic contaminants. Neither the Henry's constants nor the mass transfer coefficients for the individual organic contaminants are affected by the other contaminants present in the waste streams to a significant degree. In this work, a computer program provided by Chellam (1990) incorporating the Kavanaugh and Trussell (1980) design procedure, was used to calculate the size of the packed towers. Once the tower volume was calculated in this manner, the costs of building, operating, and maintaining the packed towers were obtained from the cost curves Gumerman et al., (1979). These costs were updated by the cost indexes corresponding to the breakdown of the different cost components. A possible residual waste stream generated by this process is the off gas from the tower which will contain the organic contaminants that have been removed from the produced waters. Treatment of this stream is not considered in this work.

1.5 REMOVAL OF ORGANIC COMPOUNDS BY ADSORPTION

Adsorption is assumed as the treatment option for organic materials that cannot be easily stripped (low Henry's constant's). In many cases, "adsorbable" compounds are in fact easily adsorbed as evidenced by high Freundlich coefficients. However, in some cases, compounds of low volatility might not be considered particularly "adsorbable" based on determinations of their Freundlich coefficients. Indeed, there is only a weak negative correlation between Henry's and Freundlich constants.

For the purposes of this work, organic compounds that have Henry's constants higher than $1.93E-4$ (atm m^3/mol) are classified as volatile and those contaminants possessing Henrys' constants below this level will be refered to as "adsorbable". The values of the isotherm constant K, for the adsorbable organic constituents found in the produced waters are, on average, an order of magnitude higher than are those for the volatile

compounds. Also, the average value of 1/n is 0.51 and is 0.42 for the adsorbable organic

contaminants and for the volatile organic compounds, respectively.

The cost and performance of the powdered and granular activated carbon systems

used to remove adsorbable organic compounds are compared in this work. Reduction of

organic compounds by adsorption (and PTA) is evaluated in terms of the bulk reduction

in total organic carbon (TOC) achieved by these processes. This is an important

assumption since it is virtually insensitive to the specific compounds that comprise the

TOC. The TOC of the raw and treated water is calculated as the sum of the mass

concentrations of each of the species measured. Removal of these species and the

reduction in TOC proceedes in order of adsorbality as reflected by the Freundlich

constants. As activated carbon becomes saturated with organic compounds, the

adsorptive capacity of the carbon nears exhaustion. When the carbon capacity is

exhausted, the carbon must be disposed of or regenerated. This consumption of activated

carbon, termed the carbon usage rate, represents one of the principal operational costs for

the adsorption processes. In comparison with GAC, higher carbon usage rates and higher

costs are typically encountered when PAC is used as an adsorbant (Figure 9). GAC

contactors tend to utilize a greater percentage of the total adsorptive capacity of the

activated carbon. While GAC contactors may allow for more efficient adsorption, they

also entail a higher capital cost.
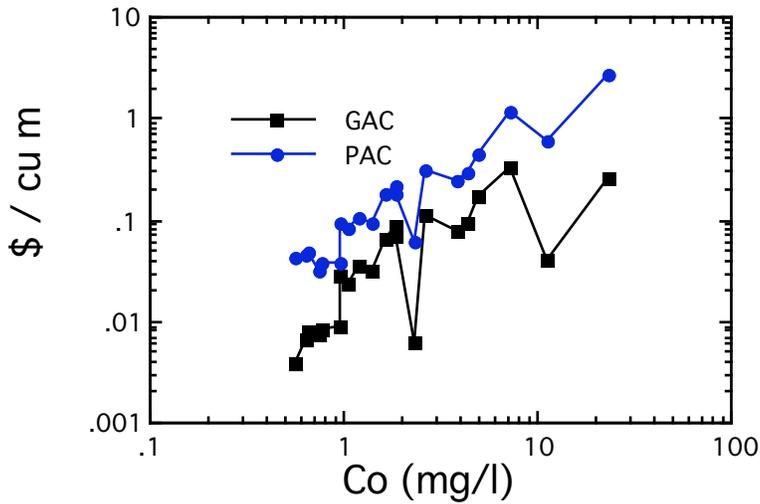
Figure 9. Carbon costs for powdered and granular activated carbon adsorption of organics out of produced waters. All streams treated to 0.5 mg/l TOC.

GRANULAR ACTIVATED CARBON

When treating produced waters with GAC, the cost components that are estimated and summed are the capital costs, the operation and maintenance costs and the cost of the carbon. The purposes of this type of treatment are best served by replacing the carbon when the bed has become exhausted. The unit cost of the virgin carbon is assumed to be $1.10 / lb*. "Spent" or exhausted carbon can also be replaced by regenerated carbon at a price set by the carbon supplier or regenerated on site. The latter option is assumed in this work.

The cost of spent carbon disposal is implicit in this price since the supplier is assumed to remove and regenerate the spent carbon. However, carbon handling losses are not included in this cost. Table 7 describes the procedure used to estimate the amount of

---

* This price is for carbon purchased in quantities equal to or greater than 2,000 lbs and was provided by the Calgon Corporation, Pittsburgh, PA , (7/1/94)

carbon that will be used during the treatment of produced water with GAC. The carbon

usage is determined by multiplying the amount of carbon in the contactor by the number

of times that this amount of carbon must be replaced. The bed is assumed to be

exhausted when the desired effluent concentration is exceeded. The contaminant

concentration profile leaving the bed is predicted using the homogeneous surface diffusion

model (HSDM) (Hand, et al., 1984). Following the procedures outlined in this

description of the the HSDM, the time to "break through" is calculated as the time untill

the TOC in the effluent from the contactor reaches a specified level. The amount of

organic contaminant that is adsorbed onto the activated carbon at equilibrium with the

local solution-phase concentration, must be known in order to use the HSDM. The

adsorption of individual compounds , when present in a mixture, must be considered for

the produced water contaminant matrix. The possible presence of at least 21 adsorbable

organic compounds requires the consideration of the competion for available space on the

activated carbon by each of the organic adsorbate compounds. The application (or

hydraulic loading) rate for a GAC adsorbtion system is the volume of water that can be

treated per unit area per time. This rate is constrained by the flow characteristics of the

waste stream through the carbon bed. In this work, the application rate is set at 5

gal/min/ft^2 (53 gal/min/m^2). A simplified model predicting the results of this competition

is presented in (Digiano, et al.1978). This model is based on the idea that if two

competing organic contaminants have identical Freundlich isotherms, they behave as if

there was only one contaminant in solution. This assumption is used to derive the

formula which uses average Freundlich isotherm constants to predict quantities of

compunds adsorbed to the activated carbon at equilibrium. Smaller degrees of removal are

achieved by blending treated and untreated portions of the waste streams. An empty bed

contact time (EBCT) of 10 minutes is assumed for the GAC contactors. This EBCT is

provided only for the portion of produced water that must necessarily be treated to reach

the final blended effluent concentration goal plus 30 % extra contactor volume to handle

variations of flow, etc.

The effect of blending is to reduce the capital costs for removing a smaller portion of the

contaminants from a specific waste stream at a specific rate of flow compared with a

higher level of treatment for the same scenario. The time to "break through" will remain

the same for the smaller contactor used for the less stringent effluent requirements

because of a constant EBCT, thus a smaller amount of carbon will be exhausted when the

level of treament is lowered for a given waste stream. Equations describing the costs of

the different components of the capital and operating costs are provided by (Adams,

J.Q., and Clark, R.M., 1989), based on the cost estimating techniques of (Gumerman,

J.C. et al, 1979). These particular cost equations are functions of the operating

parameter determined to be the most flexible and appropriate, whether it be total volume,

plan area, or design rate of flow. Table 7 explains these equations and describe the way in

which the values of the variables necessary to impliment the equations are calculated.

The carbon costs for an adsorption system are estimated by applying the current unit

carbon cost to the amount of carbon used and the capital and operation and maintenance

costs are updated in the standard fashion explained earlier. The summing of these three

components and subsequent divison of this sum by the waste stream flow rate will

produce an average unit cost for removing adsorbable organic compounds from the water.

POWDERED ACTIVATED CARBON

Powdered activated carbon (PAC), rather than GAC, will be used to affect the removal of

adsorbable organic compounds from the produced waters under certain curcumstances

when it has been proven to be the most cost effective treatment option. PAC treatment

costs include capital, operation and maintenance, and carbon costs as well as the

management and disposal cost for the residual, namely exhausted PAC. The capacity of

the PAC is exhausted at the point when no more organic contaminant matter can be adsorbed. The exhausted or "spent" activated carbon is not easily regenerated and must be disposed of. Therefore, the residuals management costs may be considerable. Pryor to disposal, the spent PAC is dried in sand drying beds. It is then sent to a landfill. A sum of $.005/m^3 is added to the treatment cost of produced waters treated with PAC as the cost of drying this sludge (PAC and associated water). This is the calculated cost for drying the spent PAC for the waste stream with the highest PAC dosage used in this work. The cost profiles for GAC and PAC (capital, operation and maintenance, and carbon) are significantly different. Since the PAC is added directly to the waste stream (prior to entering the package treatment plant), there is no need for a contactor and capital costs are lower. A small capital expenditure is required for the PAC feed system. Operating costs are primarily associated with carbon usage and disposal. These costs were estimated using equations that can be found in (Qasim, et al., 1992). The equations found here are based on the cost estimating techniques of (Gumerman, J.C. et al., 1979). These equations and the manner in which they are used are displayed in Table 5. These costs have been updated using standard procedures **.**

The amount of carbon that is used during PAC treatment of produced water is estimated, in part, with the same competitive adsorption model that is used to estimate GAC carbon usage. The capacity of the PAC is halved in this analysis to ensure that the usage rate is not underestimated. The amount of time that is necessary for the PAC to equilibrate with the waste stream varies with the molecular weight of the individual organic compounds. The range of molecular weights for these organic compounds suggests that equilibrium will not be reached and therefore the carbon usage rate must be adjusted.   A straight forward multiplication, as illustrated in Table 5. will produce a unit PAC cost for a given waste stream.

1.6 DESALINATION

Reverse osmosis and forced evaporation are the two methods of desalination considered in this analysis. The procedures used to estimate the costs related to these processes were developed by the Gas Research Institute and Remeation Technologies Inc.(GRI, 1993).   The costs associated with RO treatment of waste streams with low salinities (TDS < 5,000 mg/l), have been estimated by Clark (Clark, et. al., 1990). Cost estimations using GRI methods and the method described by Clark are compared in Figure 10**.** The GRI costs are significantly higher than the cost estimated using the method outlined by Clark. The costs are, however on the same order of magnitude. The GRI method of cost estimation is considered reasonable for this reason.
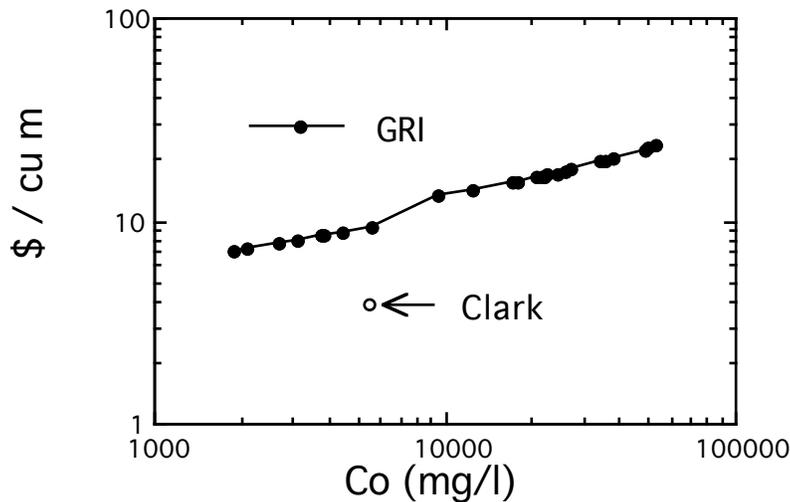
Figure 10. Cost estimating method comparison for treatment using reverse osmosis. Effluent is drinking water quality

Reverse osmosis (RO) is the most cost effective way to desalinate produced waters possessing total dissolved solids (TDS) concentrations levels at or below approximately 55,000 mg/l. This criterion distinguishes the waste streams to be desalinated by this unit process from those assumed to be desalinated using forced

evaporation. All produced waters containing more than 55,000 mg/l of TDS are assumed to be desalinated by evaporation when salt removal is required to meet the designated treatment goal The cost equations for RO systems used in this analysis are a function of the design rate of flow and the TDS concentrations. The forced evaporation costs are a function of flow rate only.   The cost components of desalination costs using RO are capital, operation and maintenance, and residual management (brine disposal). These same components comprise the costs for forced evaporation. The disposal costs for the salt resulting from forced evaporation are included in the evaporation costs. The resulting salt is assumed to be disposed of in a landfill.

REVERSE OSMOSIS

Fundamental relationships between the produced water quality and the precise characteristics of the membrane systems and the operating parameters as they affect membrane performance are not used in this analysis. While models of this complexity are available, the cost information from these models is considered to be less reliable than that obtained from the GRI cost equations. The GRI cost functions used to estimate the desalination costs in this work were provided for the GRI by an equipment vendor and are specifically applicable to the treatment of produced waters and include pretreatment costs (Table 8.). A standard pretreatment scenario, which includes a cartridge filter and pH adjustment is assumed for the RO systems. Costs calculated using this approach agree with treatment plant data for RO treatment of similar waters (GRI, 1993). The observed costs from twelve treatment scenarios were compared to costs predicted using these cost equations. These scenarios ranged in size from 25 to 1350 m^3/d and in TDS concentration from 1,000 to 35,000 mg/l. The observed costs and those predicted

by these costs funtions differed by less than 10 %, on average.   The "typical" RO system used here is assumed to maintain 94 % rejection of TDS across the membrane (GRI, 1993). The percent recovery of the influent waste stream is described by an equation also found in Table 8 (GRI,1993). As with adsorption, blending of the treated RO effluent with untreated produced water is assumed for lower removals (in this case, less than 94%). The necessary fraction of the waste stream will be treated and blended with the untreated remainder of the same stream in order to achieve a certain level of treatment. In the event that the 94 % rejection is inadequate, the permeate from one stage is treated by a second stage (i.e. the stream is treated twice). The equation describing percent recovery will also describe the amount of brine that must be disposed of. The portion of the produced water that is rejected by the RO membrane is assumed to be evaporated to dryness.

EVAPORATION

Desalination by forced evaporation is the most expensive treatment process used in the treatment of the produced waters. It is assumed to vary as a function of the design rate of flow only (Table 9). This process is used only on waters with extremely high TDS concentrations and is also assumed as the treatment scheme for the brines generated during RO treatment.

SECTION 2:   RESULTS

2.1 LIQUID/SOLID SEPARATION

The costs associated with package plant treatment vary with several parameters affecting the cost components. The size of a package plant will control the capital costs of the

17

plant while at the same time the residual management costs resulting from plant operation will vary with the initial contaminant (TSS) concentration.
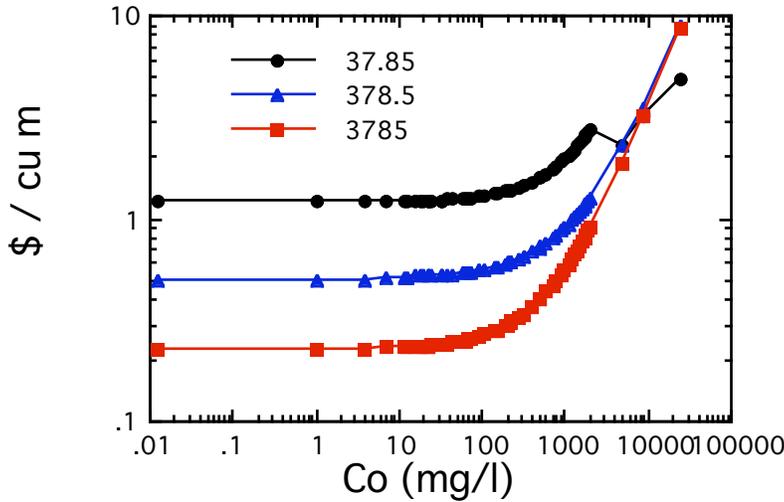
FIGURE 11. Package plant treatment costs as a function of influent TSS concentration.

The costs of package plant treatment appear to be constant with respect to flow rate and influent contaminant concentration at very low influent TSS concetraions. As the concentration of the contaminant in the influent increases, so does the level of treatment that must be performed on the residual stream before it is disposed of. The sharp rise in unit package plant cost can be directly attributed to these sludge handling costs. The capacities of the two sludge treatment processes have a dramatic affect on costs and are interrelated. Thickening of the sludge is economically feasable only for the largest produced water flow rate and several waste streams having a very high TSS concentrations at all rates of flow. Sludge is assumed, in this work to undergo a 50 percent volume reduction when thickened. This results in a 50 % reduction in landfill cost for a given waste stream. The costs associated with land filling undried sludge are 3.25 times greater than those associated with dry sludge. The residual waste streams generated during the treatment of the produced waters at the smaller rates of flow,(especially those with low TSS concentrations), are not likely to be dried for the same

reason that most of the sludge from the waste streams flowing at 37.85, and 378.5 m^3/d are not thickened. The significant drop in the unit cost of package plant treatment for the 37.85 m^3/d waste stream which occurs at approximately 3,000 mg/l of influent TSS corresponds to the point where the use of sand drying beds becomes an economically sound choice (Figure 11). The use of sand drying beds reduces the unit landfill price from $29.00/yd^3 to $9.00/yd^{3*} . All streams are treated to a "drinking water quality" of 8 mg/l TSS (assumed for this work), so there is no variable cost associated with the different levels of treatment. The cost of chemicals is constant at each rate of produced water flow. The lower chemical cost for the 3785 m3/d flow regime explains the difference in shape of the cost curve for the largest produced water flow rate (the steeper slope of increasing cost with rising TSS concentrations). A cost can be calculated for any waste stream for which the TSS content and the rate of flow is known if the calculations are performed as those in Tables 2, 3, and 4 are.

2.2 VOLATILE ORGANIC CONTAMINANT REMOVAL

Packed tower aeration is usually considered to be the most cost effective process for removing volatile organic contaminants from the water. Capital costs, including the packing material, support for the packing material and the actual tower dominate total average costs for the construction and operating of this process. The energy requirements are virtually independent of flow, and influent VOC concentration. This is due to the fact that the air:water ratio is the same for all of the towers (89.4). The model used to design the towers limits the variation in tower design and operation to tower size. A given Henry's constant will dictate the dameter of the tower and the required removal

---

* Landfill prices provided by Steve Gormily of The Fairbanks Landfill, Houston TX (1994)

determines the tower height.   Average costs increase with decreasing flow and with increasing concentrations of VOC's in the feed (Figure 12.).
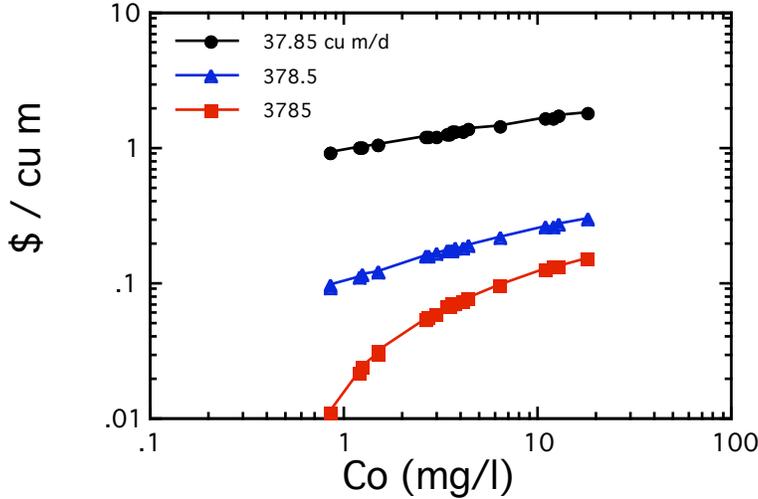

FIGURE 11. Packed tower aeration costs as a function of influent VOC concentration. Graph includes costs for produced water flow rates of 37.85, 378.5, and 3875 m^3/d. Effluent concentration is equal to 0.8 mg/l

The costs for the two smallest flow rates evaluated (37.85 and 378.5 m^3/d) are dominated by operation and maintenance costs. The stability of the curve describing the cost of stripping volatile organic chemicals from the produced water flowing at 37.85 m^3/d suggests that the energy costs might be the controlling component. The larger tower volumes require more air to be pumped in order to maintain the air:water ratio for a longer period of time. These energy requirements should vary directly with the concentration level of VOC's. Construction costs, however are more subject to economies of scale therefore the larger costs associated with the larger flow rates present curves that are not as linear as the functions representing the costs associated with smaller rates of flow. .

2.3 ADSORBABLE ORGANIC CONTAMINANT REMOVAL

The costs related to removing the adsorbable organic constituents from the produced

waters ranged from approximately $ 0.1 to $ 1.50 / m^3 of produced water treated for the

cases considered in this work. This broad range of prices can be attributed to the vast

difference between the waste stream contaminant concentrations, the different levels of

clean-up that were examined, and economies of scale. Costs at the higher end of this range

(> $ 0.50 / m^3) are the result of the relatively high capital costs incurred for the carbon

contactors and other equipment used to treat the 37.85 m^3/d waste streams. At smaller

flow fates, the investment in contactors is not compensated for by more efficient

activated carbon utilization. In these instances, PAC addition is more cost effective. The

difference in cost between the two types of activated carbon are compared in Figure 9.

There is a trade off between the capital cost savings realized by adsorbing these

contaminants with PAC and the increase in carbon costs that will occur if PAC is chosen

over GAC as an adsorbant. The costs associated with adsorbing these compounds onto

both types of carbon have been calculated and these costs were plotted versus the

concentration of adsorbable organic compounds found in the untreated waste stream.
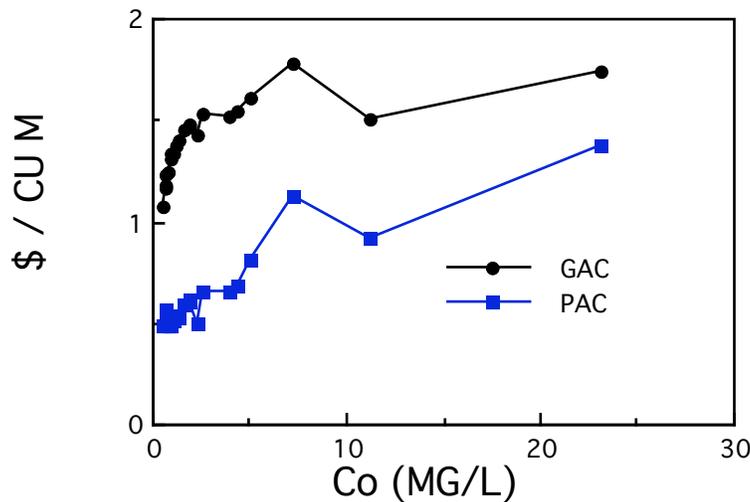

Figure 12. A comparison of total system costs for activated carbon adsorption using
PAC and GAC plotted as a function of Co [adsorbable organic compounds]. The
produced water flow rate is 37.85 m^3/d and the effluent concentration is 0.5 mg/l.

These types of GAC and PAC cost curves intersect at one point for some of the flow regimes. This intersection corresponds to the point where low capital costs of a powdered activated carbon systems are overshadowed by increased carbon costs (high carbon usage rates). There is no such intersection on the curve representing PAC and GAC costs for the 37.85 m^3/d produced water flow rate (Figure 12). This simply means that for these particular waters, the capital costs associated with GAC adsorption make PAC the preferred treatment option.
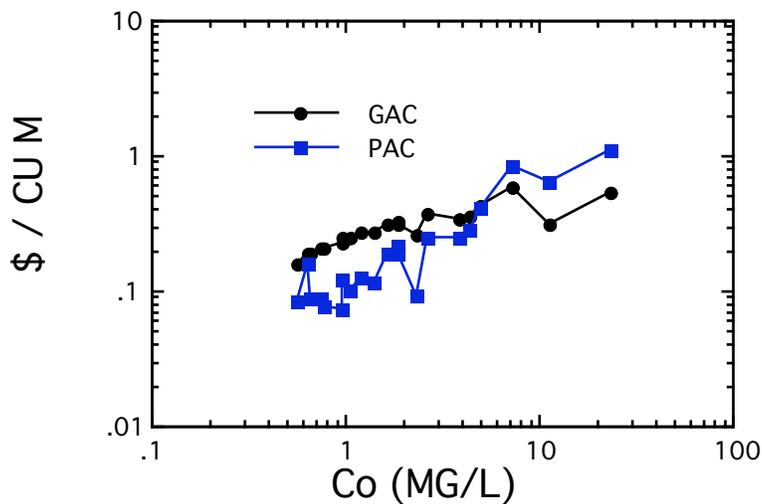

Figure 13. A comparison of total system costs for activated carbon adsorption using PAC and GAC plotted as a function of Co [adsorbable organic compounds]. The produced water flow rate is 378.5 m^3/d and the effluent concentration is 0.5 mg/l.

At high concentrations for the organic compounds in the influent waste stream these PAC systems use more carbon than GAC sytems do. This is because of the greater efficiency of carbon utilization on GAC . Examples of this are depicted in Figures 13 and 14, where the costs of both a PAC and GAC system are illustrated at flow rates of 37.85 and 378.5 m^3/d, respectively. At approximately 6 mg/l of contaminant concentration in the influent, GAC becomes the most cost effective method of carbon adsorption to use. Cost curves similar to Figures 13 and 14 have been plotted for the other relevant produced water flow regimes and treatment levels. Several of the curves show the level of adsorbable organic compound concentration in the waste stream at which treatment with

GAC becomes more cost effective than treatment with PAC. The influent concentration level of 5.75 mg/l for a flow rate of 378.5 m^3/d is shown to correspond to this point (Figure 13.) As larger quantities of produced water are treated, GAC becomes the prefered treatment option for adsorptive removal of organics at lower and lower concentrations. At a capacity of 3785 m^3/d, treatment of produced water using GAC is calculated to be comparable or cheaper than that using PAC over the entire range of influent concentrations that were investigated (Figure 14).
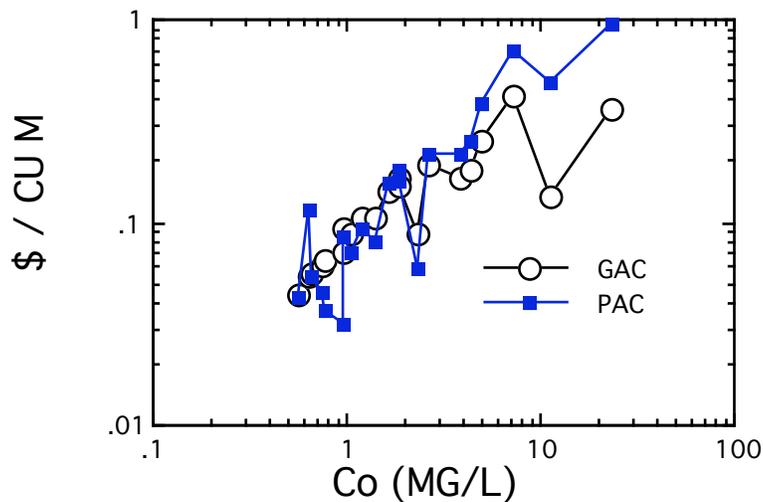
Figure 14. A comparison of total system costs for activated carbon adsorption using PAC and GAC plotted as a function of Co [adsorbable organic compounds]. The produced water flow rate is 3785 m^3/d and the effluent concentration is 0.5 mg/l.

The effluent concentration for all of the systems used in this comparison is 0.5 MG/L. Curves similar to these for different levels of treatment tell us which type of activated carbon to use in order to remove this particular type of waste stream constituent. Figures 15, 16, and 17 illustrate the costs of treating the produced waters to different levels of cleanliness using PAC at three seperate rates of flow.
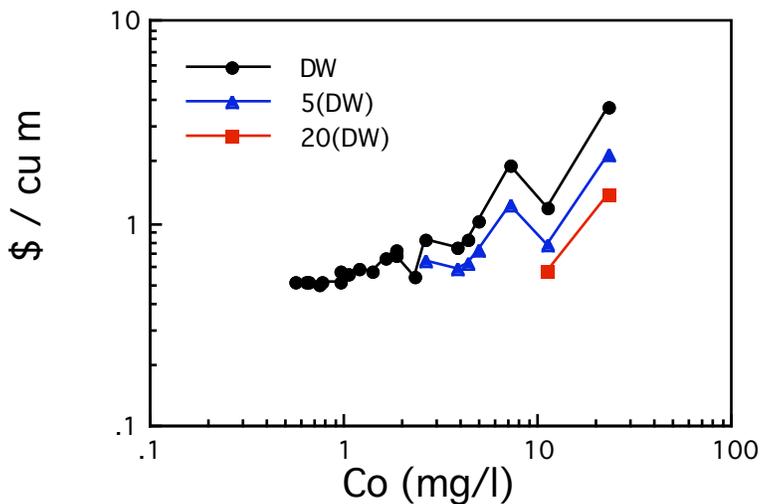
Figure 15. PAC costs presented as a function of as a function of the initial concentration of adsorbable organic compounds. Effluent concentration levels vary from the drinking water benchmark (0.5 mg/l) to twenty times that amount. Waste stream flow rate is 37.85 m^3/d. The decrease in the number of data points on the cost curves labeled (5DW) and (20DW) corresponds to a decrease in the number of waste streams requiring treatment to reach the effluent goals shown.
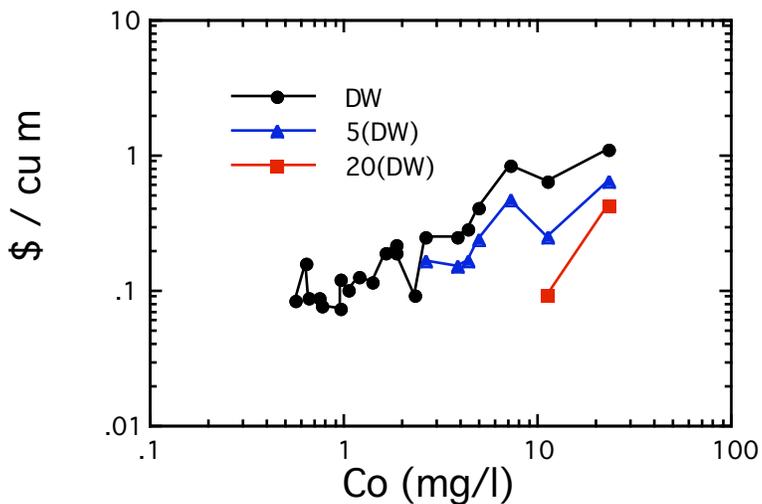
Figure 16. PAC costs presented as a function of as a function of the initial concentration of adsorbable organic compounds. Effluent concentration levels vary from the drinking water benchmark (0.5 mg/l) to twenty times that amount. Waste stream flow rate is 378.5 m^3/d. The decrease in the number of data points on the cost curves labeled (5DW) and (20DW) corresponds to a decrease in the number of waste streams requiring treatment to reach the effluent goals shown.

24
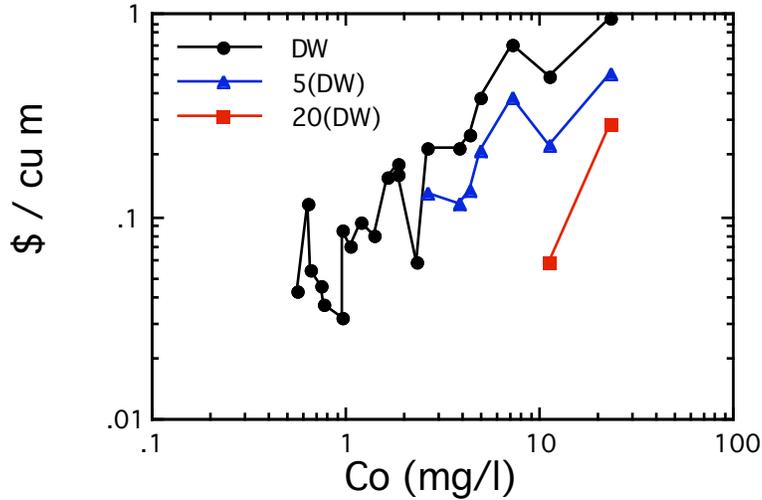
Figure 17 PAC costs presented as a function of as a function of the initial concentration of adsorbable organic compounds. Effluent concentration levels vary from the drinking water benchmark (0.5 mg/l) to twenty times that amount. Waste stream flow rate is 3785 m^3/d. The decrease in the number of data points on the cost curves labeled (5DW) and (20DW) corresponds to a decrease in the number of waste streams requiring treatment to reach the effluent goals shown.

The carbon costs for these three capacities, the levels of effluent quality evaluated, and both types of activated carbon vary with the influent adsorbable contaminant concentration ,as would be expected. The unexpected dips and sharp rises displayed on the cost curves are due to carbon costs. These costs are directly proportional to the activated carbons affinity for a specific waste streams organic contaminant matrix. This affinity can be thought of as the relative adsorbability of the different organic waste waters. All of the produced waters are treated in order to remove the same constituents during activated carbon treatment (PAC and GAC). Each point on the activated carbon cost curves corresponds to a separate produced water containing the initial concentration of some of the adsorbable organic materials expressed as TOC. The relative ratios of

contaminants in each produced water are different. Thus the cost of treatment at each level of TOC reflects changes in the compositon of the TOC as well as its concentration.
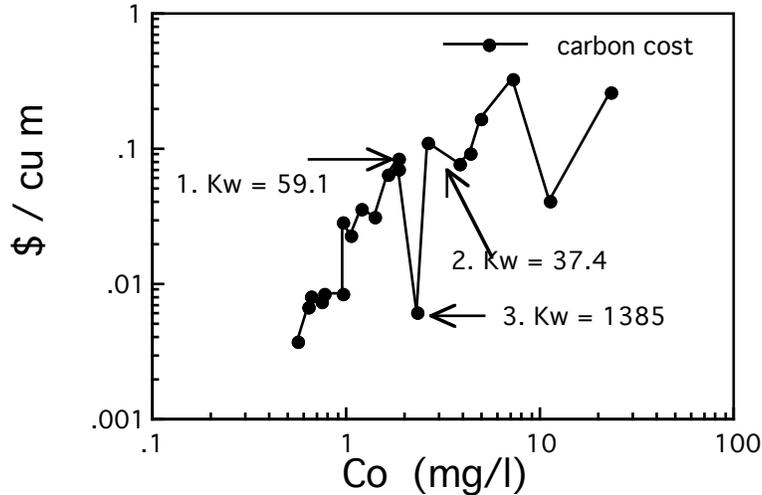

Figure 18 Carbon costs associated with the removal of adsorbable organic compounds onto GAC as a function of the initial concentration of adsorbable organic compounds in untreated produced waters. The value Kw is the weighted mean of the Freundlich coefficient K for the constituents of the waste stream corresponding to the designated graph points .

All points on the graph represent carbon costs associated with removing organic contaminants from distinct individual waste streams. The weighted mean value of the Freundlich isotherm constant K for all of the organic constituents in three of the waste streams are shown in Figure 18 as Kw. The mean values for this constant are weighted by the amounts of each in the waste streams

$$Kw = \frac{\sum C_i K_i}{\sum C_i}$$

The GAC has a large capacity for the contaminants found in the particular stream that has been evaluated at the graph point corresponding to a water possessing a weighted mean value of 1385 for Kw   The carbon does not have a large capacity for the organic contaminants found in the waste streams corresponding to the Kw values of 59.1 and

37.4. The Freundlich isotherm coefficient, K can be likened to a parameter describing the capacity of an activated carbon to adsorb a particular organic constituent or group of constituents. The weighted average of this isotherm coefficient corresponding to the contaminants found within the waste stream corresponding to the point Kw = 1385 is much larger than any of the averages corresponding to the other waters. The more favorable isotherms for the most principal contaminants found in this stream explain the dips in the amount of carbon used and consequently the lower cost. The cost of carbon for the treatment of the waste streams having weighted mean K values of 59.1 and 37.4 are more expensive than would be expected due to the low capacity of the carbon for the organic constituents most principal in these streams.
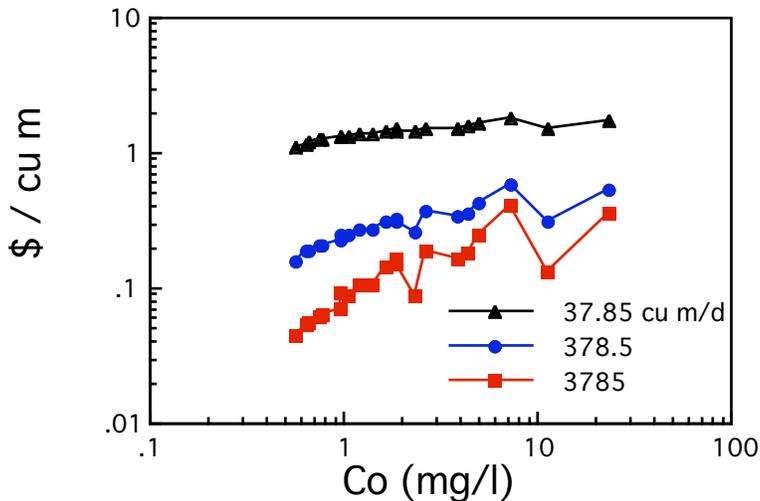
Figure 19 GAC adsorption costs for flow rates of 37.85, 378.5, and 3785 m^3/d plotted against the initial concentration of adsorbable organic compounds in the influent. The effluent concentration of each stream is 0.05 mg/l

In Figure 19, the peaks and valleys that exist within the costs represented in Figure 18 are present in each of the curves. The curves which represent the activated carbon treatment costs of produced water flowing at 37.85 and 378.5 m^3/d obscure this trend because the capital costs of the treatment dominate at lower capacities.

The difference in the costs incurred for the treatment of the produced waters to different levels of adsorbable contaminant removal represent a potential savings for the instances where a less stringent effluent level is necessary. The costs associated with the treatment scenarios described and shown in Figures 20, 21, and 22 are evidence of this fact.
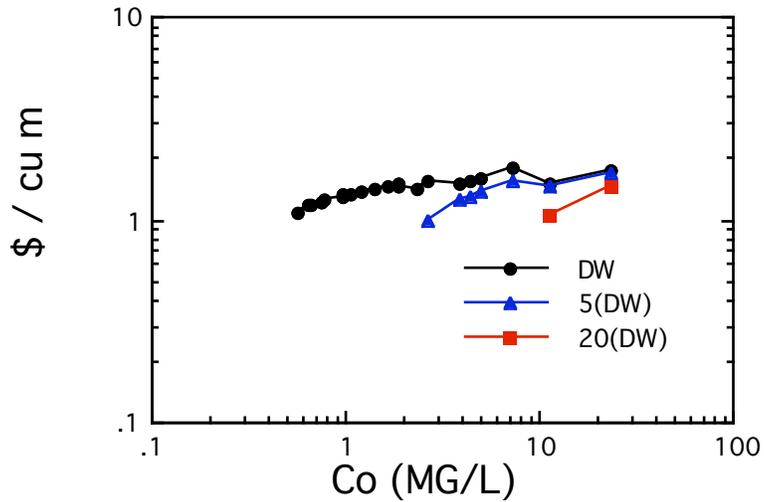

Figure 20. GAC adsorption costs presented as a function of the initial concentration of adsorbable organic compounds. Effluent concentration levels vary from the drinking water benchmark (0.5 mg/l) to twenty times that amount. Waste stream flow rate is 37.85 m^3/d. The decrease in the number of data points on the cost curves labeled (5DW) and (20DW) corresponds to a decrease in the number of waste streams requiring treatment to reach the effluent goals shown.
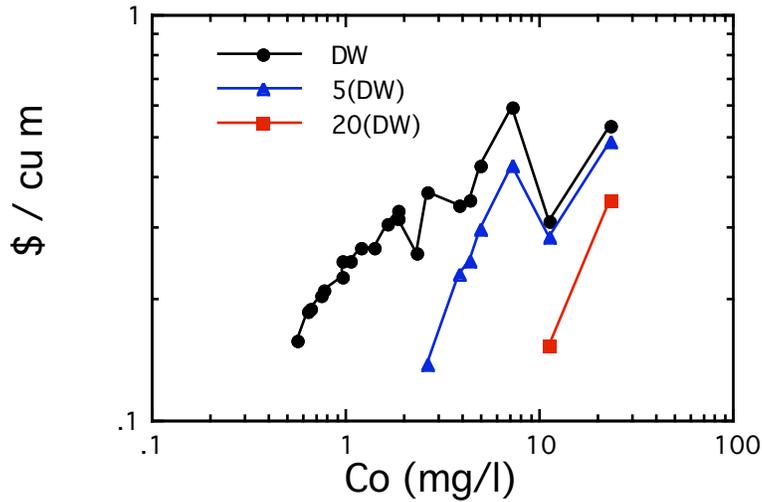
Figure 21. GAC adsorption costs presented as a function of the initial concentration of adsorbable organic compounds. Effluent concentration levels vary from the drinking water benchmark (0.5 mg/l) to twenty times that amount. Waste stream flow rate is 378.5 m^3/d. The decrease in the number of data points on the cost curves labeled (5DW) and (20DW) corresponds to a decrease in the number of waste streams requiring treatment to reach the effluent goals shown.
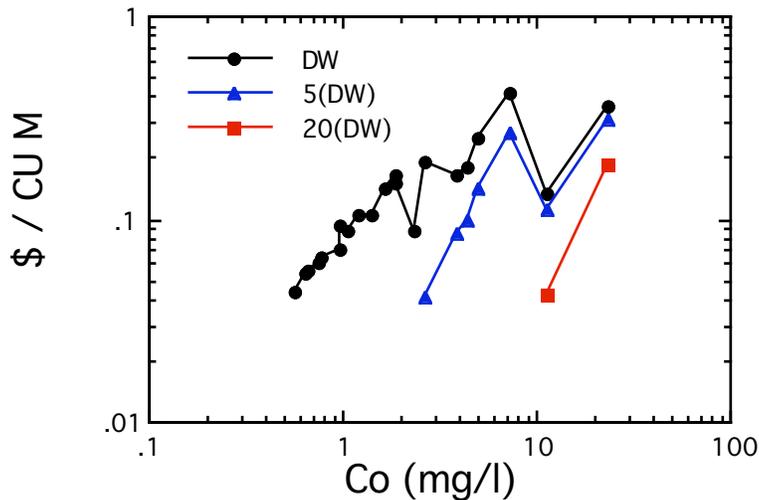
Figure 22. GAC adsorption costs presented as a function of the initial concentration of adsorbable organic compounds. Effluent concentration levels vary from the drinking water benchmark (0.5 mg/l) to twenty times that amount. Waste stream flow rate is 3785 m^3/d. The decrease in the number of data points on the cost curves labeled (5DW) and (20DW) corresponds to a decrease in the number of waste streams requiring treatment to reach the effluent goals shown.

2.4 DESALINATION

FORCED EVAPORATION

Forced evaporation is assumed as the treatment process for desalination when waste
streams have TDS levels above 55,000 mg/l, or about three times the salinity of sea water.
The cost of implementing this type of treatment is very high and probably prohibitive at
the smallest produced water flow rates. However, where alternative sources of water
may not be available, desalination of these very saline waters could, under unusual
circumstances, be desirable.
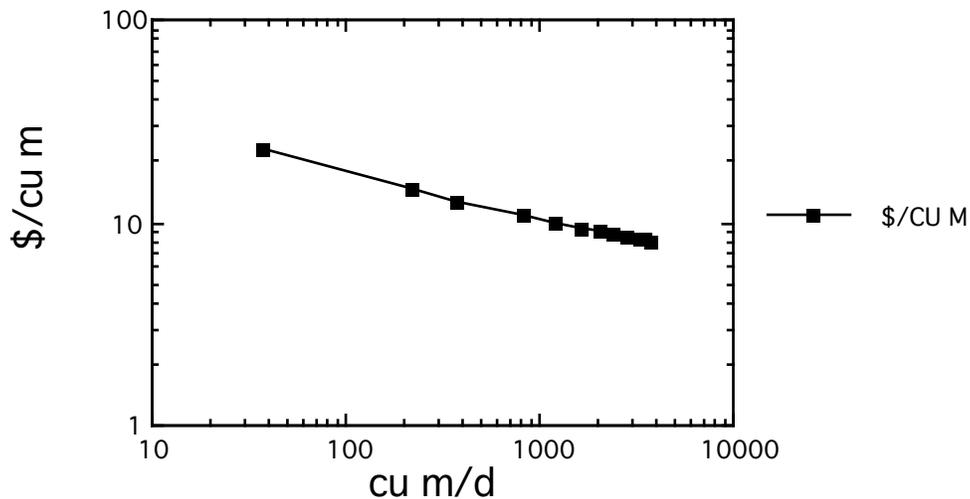
Figure 23. Forced evaporation costs. System capacity determines the system cost,
regardless of contaminant level.

The average cost for evaporation varies only slightly with system size (GRI, 1993).

Desalination of produced waters with lower TDS levels, using reverse osmosis (a process typically considered to be expensive), is significantly less expensive than forced evaporation. The costs associated with RO treatment range from a little less than $2.00 / m^3 up to over $25.00 / m^3
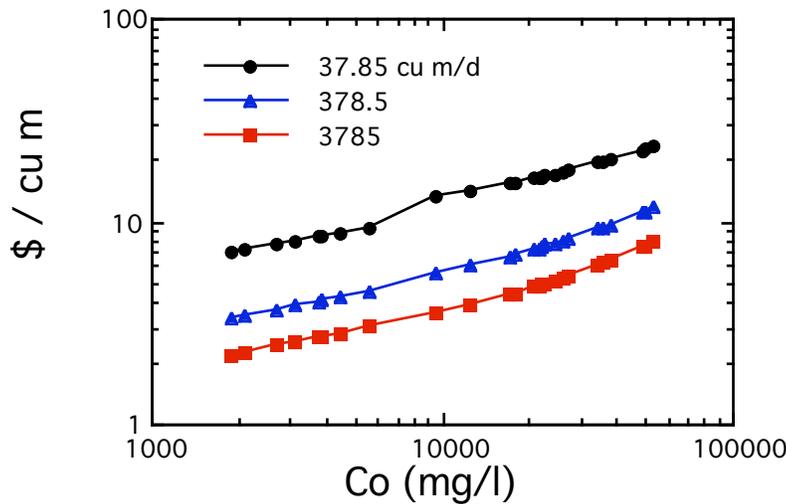

Figure 24. The estimated costs of RO desalination of produced waters for flow rates of 37.85, 378.5, and 3785 m^3/d as a function of initial waste stream TDS concentration. Effluent TDS concentration is 500 mg/l.

Economies of scale and severely increasing costs for the operation and maintenance of the systems at high TDS levels are the reasons for the difference between the higher and lower costs associated with RO treatment of the produced waters. The apparent rise in cost that can be seen in Figure 24 for all three rates of flow occurs at 8,333 mg/l of influent TDS concentration. This is the highest level of influent concentration that can be reduced to 500 mg/l in a single pass through the system. In order to meet this effluent criterion for the waters having TDS levels above 8,333 mg/l, permeate must be treated and the operation and maintenance requirements for the system are increased.
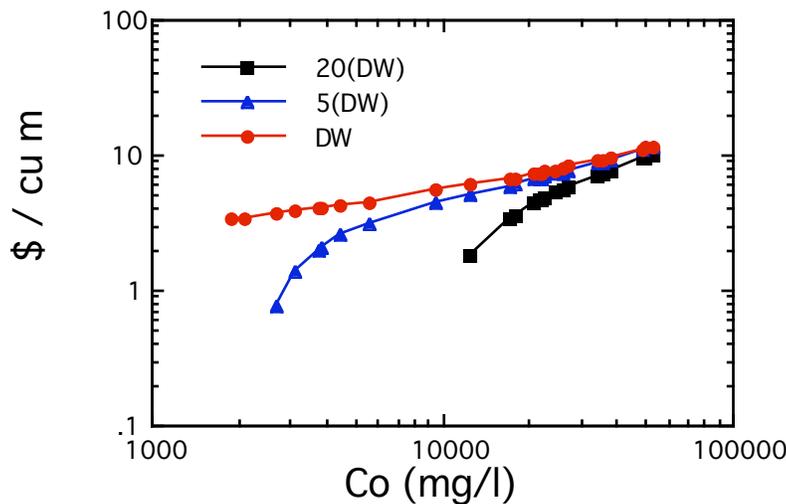
Figure 25 The estimated costs of RO desalination of produced waters at a flow rate of 378.5 m^3/d as a function of initial waste stream TDS concentration. Effluent TDS concentrations range from a drinking water benchmark (500 mg/l) to 20 times the standard. The decrease in the number of data points on the cost curves labeled (5DW) and (20DW) corresponds to a decrease in the number of waste streams requiring treatment to reach the effluent goals shown.

Waters that will not require the treatment level that would be accomplished by one pass through the RO system will undergo treatment proportionate to the level needed. A portion of the waste stream will be diverted from the RO treatment system and later blended with the treated portion. As the portion of the produced water being treated increases for the waste streams with less stringent effluent requirements, the unit costs for all streams converge. The operation of the treatment processes to achieve the different TDS concentration levels become similar to one another as the influent concentration increases. For example, the waste stream being treated to five times the drinking water stanard [5(DW)], will only pass a portion of its volume through the RO system at low levels of influent concentration. As the influent levels increase, a larger portion of the waste stream is actually treated until, for the very highest levels of influent concentration, a second pass through the system is necessary. The operating parameters

of the waste stream being treated to the drinking water standard (DW) are handled in much the same way except that the parameter changes occur at much lower influent concentration levels. Costs associated with the desalination of the produced waters using reverse osmosis behave similarly, regardless of flow rate. The shapes of the cost curves for reverse osmosis treatment of produced water at flow rates of 37.85 and 3785 m^3/d (Figures 26 and 27) resemble the shape of the cost curve shown in Figure 25. The actual costs decrease as the waste stream flow rate increases, due to economies of scale.
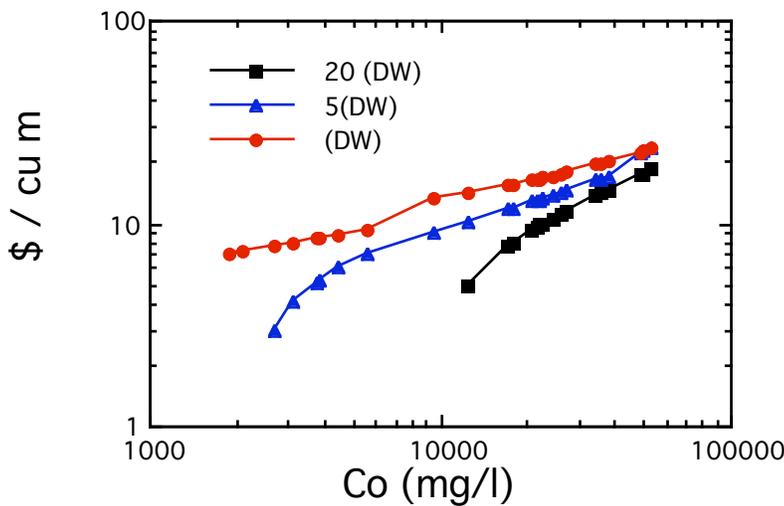

Figure 26. RO costs for produced water desalination as a function of influent TDS concentration and effluent requirements. Waste streams are treated to drinking water benchmark and multiples of drinking water benchmark. The flow rate for each treatment level is 37.85m^3/d. The decrease in the number of data points on the cost curves labeled (5DW) and (20DW) corresponds to a decrease in the number of waste streams requiring treatment to reach the effluent goals shown.
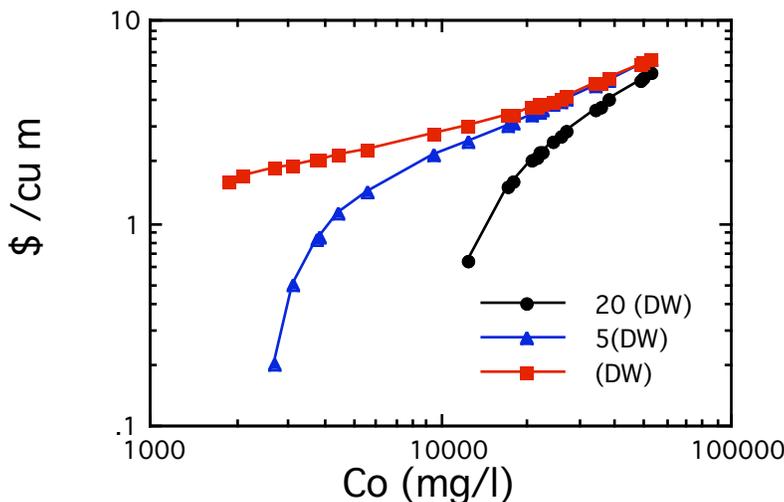
FIGURE 27. RO costs for produced water desalination as a function of influent TDS concentration and effluent requirements. Waste streams are treated to drinking water benchmark and multiples of drinking water benchmark. The flow rate for each treatment cost shown is 3785 m^3/d. The decrease in the number of data points on the cost curves labeled (5DW) and (20DW) corresponds to a decrease in the number of waste streams requiring treatment to reach the effluent goals shown.

Forced evaporation is the less expensive method of desalination in only a very few instances. These few cases are identified from the the cost curves. A comparison of the two processes in one figure would describe an extremely limited situation. The influent and effluent TDS concentrations , as well as the flow rate and level of treatment , all can have a profound effect on the cost of desalination by RO.

2.5 AGGREGATE COSTS

The total costs for the treatment of several representative waste streams are represented in Figure 28. The columns are labeled with the dominant waste stream characteristic. Representative waste stream number 14 (R 14) refers to a produced water that has a TDS concentration of 500,000 PPM. The desalination cost for this waste stream

dominates the costs associated with the removal of all other contaminants found in the water. R7 is described as "low", describing its level of contamination, because suspended solids is the only category of contaminant that requires treatment in order to bring the quality of the waste stream to the drinking water standard as it is defined in this work. The costs shown in Figure 28 illustrate which of the waste stream constituents are the most expensive to remove. The removal of dissolved solids from any waste stream which has a significant amount of TDS will usually dominante the cost of treatment. The costs associated with the treatment necessary to remove the highest levels of contaminants found in the other categories will be overshadowed by the cost of removing a significant amount of TDS
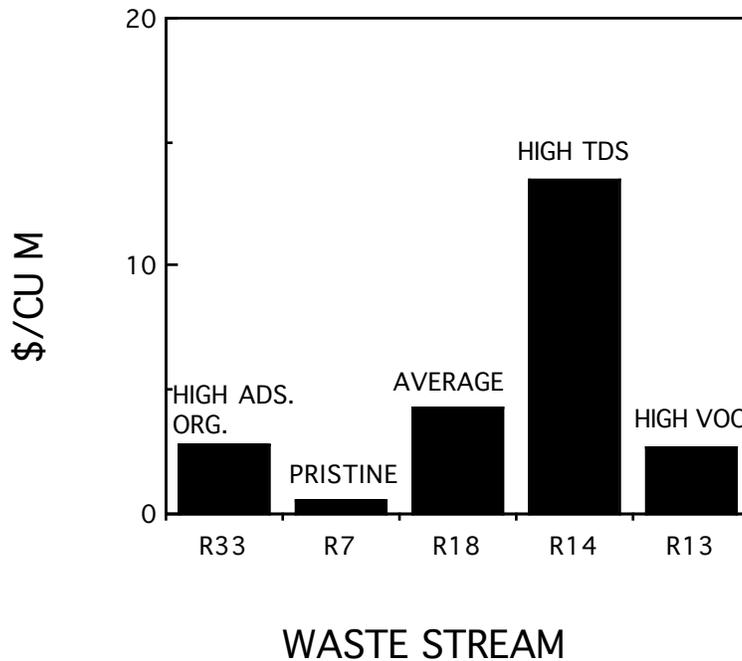

Figure 28 Total costs associated with the treatment of several representative waste streams. The flow rate of each stream is equal to 378.5 m^3/d.

These total treatment costs can be obtained from the cost curves that have been created for each of the different processes. The package treatment plant cost curve can be used

with only the knowledge of waste stream flow rate and TDS concentration. A packed

tower aeration cost will require a tower volume.
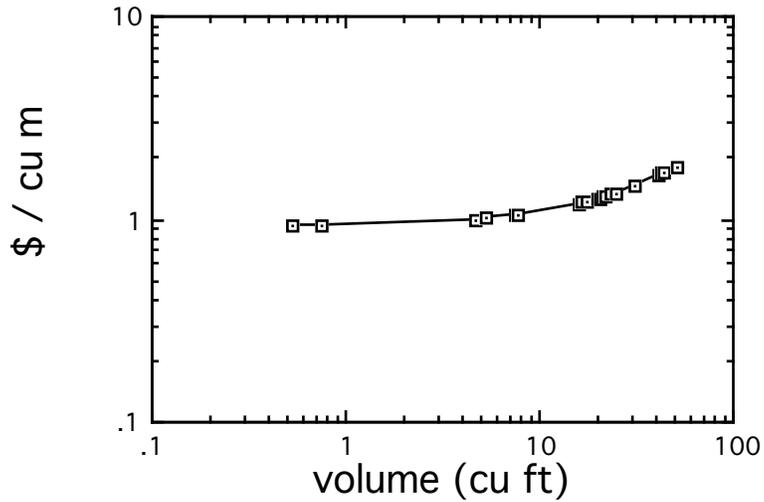
**Figure 29** Packed tower aeration costs as a function of tower volume at a flow rate of
37.85 m³/d
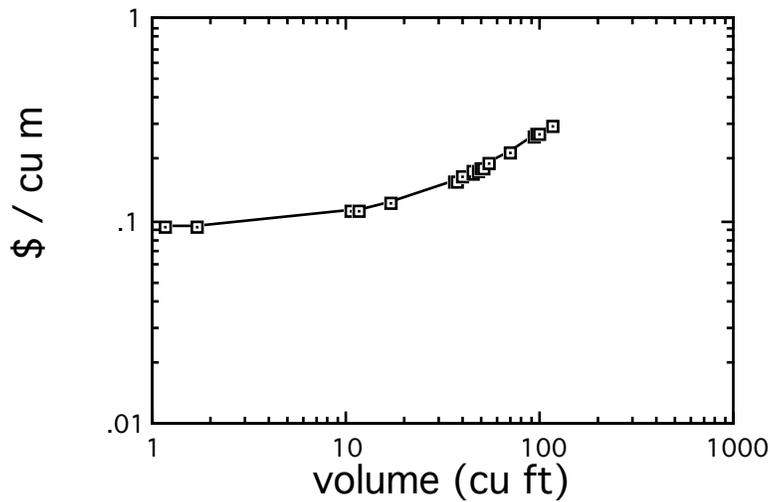
Figure30. Packed tower aeration costs as a function of tower volume at a flow rate of
378.5 m³/d
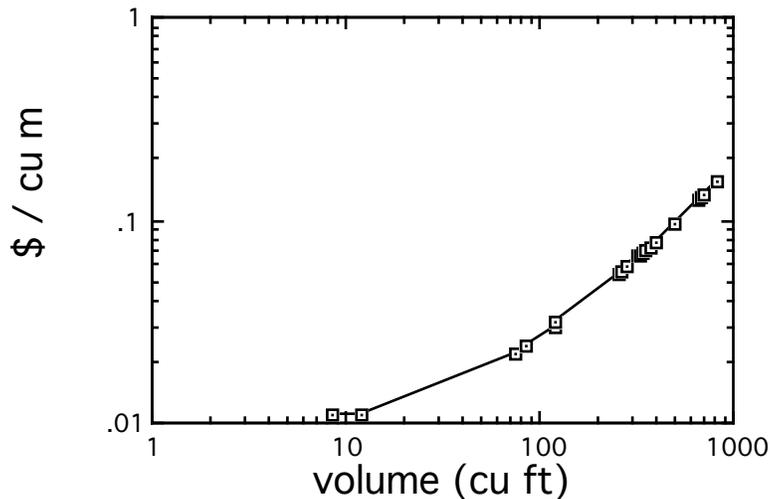
Figure 31. Packed tower aeration costs as a function of tower volume at a flow rate of 3785 m^3/d

The costs of packed tower aeration as a function of tower volume can be found in Figure 29 through 31 Knowledge of the relative adsorbability of the organic contaminant matrix of a given produced water is required to adjust the cost found on any of the cost curves pertaining to activated carbon adsorption. An estimate of cost can be obtained from the cost curves with some knowledge of organic contaminant levels in a given produced water. More accurate costs are taken from the curves that describe the RO and Evaporation costs. TDS concentration level and waste stream flow rate is the necessary information to estimate these components of produced water treatment cost.

The costs associated with the treatment of the representative waste streams shown in Figure 28. allow for an estimate of the costs associated with the treatment of all of the produced water that is generated in a typical year in the United States. The relative levels of contaminants in produced waters were identified from a simple statistical analysis of the database assembled in this work. The geometric mean of the amounts of the constituents that were examined in the database were calculated for all contaminants.

These values were compared to the values found in the analyses of existing waters. A particular produced water was chosen to represent the average quality of produced water, in general. This "average" water is labelled R18 in Figure 28 The estimated cost of treating this water, achieving the removal of most solids and undesirable inorganic contaminants, TDS levels at or below 500 mg/l, as well as adsorbable and strippable organic levels of 0.5 and 0.8 mg/l, respectively is almost $5.00 / m^3 . If the annual volume of produced water in a given year is taken to be 5 billion m^3, the cost of treating all of it to the level mentioned above would be 25 billion dollars per year.

References

Adams, J.Q., Clark, R.M., and Miltner, J.M. Controlling Organics with GAC: A Cost and Performance Analysis. Jour. AWWA, Vol. 81, No. 4, p. 132 (April, 1989)

Adams, J.Q. and Clark, R.M. Cost Estimates for GAC Treatment Systems. Jour. AWWA, Vol. 81, No. 1, p. 35(Jan., 1989)

Adams, J.Q. and Clark, R.M. Evaluating the Costs of Packed-Tower Aeration and GAC for Controlling Selected Organics. JAWWA Vol 83, No. 1, pp. 49-57 (Jan, 1989)

American Petroleum Institute. Waste Analysis Report, (1987)

Bauman, E.R. Granular-Media Deep-Bed Filtration. Water Treatment Plant Design for the Practicing Engineer, edited by R.L. Sanks, Ann Arbor Science Publishers, Ann Arbor Michigan, (1978)

Chellam, S. Design of Packed Towers to Remove Volatile Organic Compounds. (1990) (unpublished)

Digiano, F.A., Baldauf, G., Frick, B., and Sontheimer, H. A Simplified Competitive Adsorption Model. Chemical Engineering Science Vol. 33, No. 12, p. 1667, (April, 1978)

Gas Research Institute. Remediation Technologies Inc. Produced Water Management Options Model. User's Manual, Version 2.1., (1993) (unpublished)

Gumerman, R.C., Culp, R.L., and Hasen, S.P. Estimating Water Treatment Costs: Vol. 1-4., U.S. Environmental Protection Agency, EPA 600/2-79-162 a-d., (Aug., 1979)

Hand, D.W., Crittenden, J.C., Gehin, J.L., and Lykens, B.W. Design and Evaluation of an Air Stripping Tower for Removing VOC's from Groundwater. Jour. AWWA, Vol. 78, No. 9, p. 87, (Sept., 1986)

Hand, D.W. et al. Simplified Models for Design of Fixed Bed Adsorption Systems. Jour. Env. Eng.,Vol. 110, No 2, p. 440 (April, 1984)

Kavanaugh, M.C., and Trussel, R.R. Design of Aeration Towers to Strip Volatile Contaminants from Drinking Water. Jour. AWWA, Vol. 72., No. 12, p. 684, (Dec., 1980)

Kos, P. Gravity Thickening of Water Treatment Plant Sludges. Jour. AWWA, Vol. 69, No. 5,   p. 272, (May, 1977)

Lawrence, A.W., Miller, J.A., Miller, D.L., and Linz, D.G. An Evaluation of Produced Water Management Options in the Natural Gas Production Industry SPE/EPA Expolration and Production Environmental Conference. (Mar., 1993)

Mehrotra, A.K., and Banerjee, A. Evaluation of Reverse Osmosis for the Treatment of Oil Sands Produced Water. Water Pollution Research Journal, Vol. 21, No 2, p. 141, (1986)

Nielsen, H.L., Carnes, K.E., and DeBoice, J.N. Alum Sludge Thickening and Disposal. Jour. AWWA, Vol 65, No. 6, p 100(June, 1973)

Qasim, S.W., Siang, W.D.L., Motley, E.M., and Heung, K.G. Estimating Costs for Treatment Plant Construction. Jour. AWWA, Vol. 84, No. 8, p.56, (August, 1992)

Sontheimer, H. The Use of Powdered Activated Carbon. Translation of Repoorts of Special Problems of Water Technology, vol 9, Adsorption, U.S. Environmental Protection Agency, Report EPA-600/9-76-030., (Dec., 1976)

Tomson, M.B., Oddo, J.E., Kan, A.T. Brine Management Options to Gas Research Institute Report No.:92-253-0438