

**OPTIMIZATION OF COMBINED AIR STRIPPING AND
ACTIVATED CARBON ADSORPTION FOR VOC REMOVAL FROM GROUNDWATER**

by
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ABSTRACT

This study examines the combined treatment processes of air stripping and activated carbon adsorption for removing carbon tetrachloride, trichloroethylene and 1,4 dichlorobenzene from groundwater. Air stripping was used as a pretreatment with activated carbon as a polishing step. Optimization is achieved by determining the minimum operating and amortization costs for the variables of packing media depth versus carbon usage rate. A simplified design example illustrates the method of determining the optimum combination of treatment processes for the compounds and concentrations examined.

The study included laboratory experiments which examined the air stripping performance of two different packing media as well as developing activated carbon isotherms. The transfer unit model was utilized to estimate the overall liquid mass transfer coefficient ($K_L a$) for all three compounds at all experimental points.

INTRODUCTION

Numerous chemical contaminants have been occurring in drinking water supplies with increasing frequency during the past several years. Federal, State, local and private studies have sampled and identified many hundreds of different chemical compounds in groundwater. Most are the result of accidental or deliberate discharges of industrial or commercial chemicals and waste products. Among these are volatile organic compounds (VOCs) and synthetic organic compounds (SOCs). Many of these substances have been found to cause serious adverse health effects in animals and humans and have prompted the implementation of regulations to control their presence in drinking water.

Presently, hundreds of drinking water suppliers are faced with contamination problems which must be addressed. Over recent years, numerous groundwater pollutants were treated with granular activated carbon (GAC). However, GAC often proved to be somewhat ineffective as a reliable and cost effective means of treatment. Frequently VOCs would achieve early breakthrough, thereby requiring frequent and expensive carbon replacement or regeneration.

Recently, many researchers and engineers are recognizing the advantages of using air stripping and activated carbon in a two-stage treatment process. This system has the ability to effectively treat groundwaters containing multiple contaminants having diverse

volatilities and adsorption characteristics. Using air stripping as a pretreatment step will remove most or all of the highly volatile compounds as well as varying amounts of the lesser volatile compounds thus reducing the organic load on the activated carbon. There is also a wide range of design and operation flexibility with the combined systems.

This study attempts to demonstrate the applicability of this system for a selected combination of chemical compounds. Laboratory experiments were conducted to establish performance of a typical air stripper and adsorption isotherms were developed to determine the capacity of a specific activated carbon for the multi-solute solutions.

Current research and implementation of combined air stripping and carbon adsorption is proving to be an effective technology. The means by which an optimum combination is determined will remain a challenge to the engineer. This study suggests one basic approach together with the development of the necessary preparatory information.

OBJECTIVES

The objective of this study was to demonstrate the advantages of combining air stripping with activated carbon adsorption for removing VOCs from groundwater. Optimization is achieved by performing a comparison of the treatment costs associated with accomplishing a given level of VOC removal. Laboratory experiments were utilized to establish air stripping performance and activated carbon capacity for treating three selected organic contaminants. The compounds were examined in multi-solute solutions in order to realistically reflect any competitive effects.

LITERATURE REVIEW

The process of air stripping and activated carbon adsorption have been widely used in recent years, but generally as separate systems. The water treatment and chemical engineering fields have generated a vast number of publications and references relating the both of these processes. Only recently have researchers and industry begun to report on water treatment systems which combine air stripping with activated carbon adsorption.

A rather extensive literature search resulted in many reference articles and bibliographical sources related to VOCs, air stripping and activated carbon adsorption. Most information is from the American Water Works Association Journal, Research Foundation reports and conference papers. Many additional articles were found in other engineering journals, periodicals and conference proceedings.

Approximately 80 percent of the references were published within the last five years and the remaining 20 percent were published between 1975 and 1980. Clearly the period since 1980 has been the most intensive for scientific research and engineering activities in this field.

It is interesting to note the chronological sequence of events as reflected in the topics reported in periodical literature.

1975 - Discovery of THMs, SOCs and VOCs in drinking water.

1978 - EPA proposes use of granular activated carbon for treatment to meet MCLs.

1980 - Initial publication of research and studies of air stripping and aeration techniques for treatment of THMs, SOCs, and VOCs. Most research involved comparisons between GAC and air stripping in an attempt to demonstrate that air stripping was a better technology for VOCs, THMs and some SOCs.

1981 - Many case studies are reported which demonstrate effective use of air stripping for VOC removal from drinking water. Most researchers do indicate that a series combination of air stripping and GAC would be effective for removal of multiple contaminants involving volatile and non-volatile organics.

1983 to 1987 - Research and plant studies present quantitative data regarding the effectiveness of series operation of air stripping and GAC treatment. Improvements in air stripper design, packing media and application are evident.

Air Stripping; Theory, Models and Studies

The basis of air stripping theory is found in the two-film model for mass transfer between gases and liquids [31]. There are four steps in the transfer process; 1) diffusion within the bulk liquid, 2) transfer through the liquid film, 3) transfer through the gas film and, 4) diffusion within the bulk gas. When sufficient mixing is provided within the bulk gas and bulk liquid phases, the bulk-diffusion factors are relatively negligible and the rate controlling factor becomes the transfer through the gas-liquid interface. The degree of solubility of

the gas will then dictate whether the gas or liquid film will be rate controlling. VOCs have relatively low solubilities, thus the mass transfer across the liquid film is generally rate controlling. The term used to represent this is the overall liquid mass transfer coefficient, K_L . The driving force for mass transfer is the concentration gradient that exists between the bulk gas and liquid. The overall rate of mass transfer from bulk liquid to bulk gas is mostly dependent upon the solubility of the VOC, usually expressed in terms of a Henry's Law constant, and the magnitude of the concentration gradient produced by the air stripper. The effects of relative diffusivity of VOCs generally is not important because there is significant turbulence within the bulk solution and the diffusivity coefficients, dependent upon molecular size and weight, often do not differ significantly among VOCs.

Several different design models for air stripping of volatile organic compounds (VOCs) from water have been developed. The various equations generally determine the overall mass transfer coefficients based upon variables such as, compound characteristics, temperature, packing media, liquid loading rate, air to water ratio, and compound concentration. Subsequent calculations and characteristic curves are then utilized to determine packing height, pressure drops, tower diameter and final air to water ratios.

The chemical engineering field has long used a transfer unit model for gas absorption system [1]. The efficiency of a given packed column is represented by the Height of a Transfer Unit (HTU), and the

number of transfer units are chosen to achieve a desired transfer ratio within a certain column height. Chemical engineers generally design systems that handle concentrated solutions whereas water treatment concentrations are much more dilute. Consequently, the procedures developed for the design of gas absorption systems can be simplified and adapted to water treatment applications.

The kinetic theory of gases states that molecules of dissolved gases can readily move between the liquid and gas phases. Consequently, if water contains a volatile compound in excess of its equilibrium concentration, then the compound will move from the liquid phase (water) to the gas phase (air) until equilibrium is reached. The driving force for air stripping is derived by maintaining a gradient between the equilibrium conditions and the operating condition within the column.

Different compounds will exhibit different degrees of volatility based upon their Henry's Law constant. Henry's Law constant is the ratio of the partial pressure of a compound in air to the mole fraction of the compound in water at equilibrium. Compounds with a high Henry's Law constant are easier to strip from water than those with a low Henry's Law constant. Determination of an appropriate Henry's Law constant for the design model equations is an important step. For most compounds of concern in domestic water treatment these are generally accepted values which are widely published. Some researchers have recently conducted the necessary laboratory experiments to establish or verify Henry's Law constants [2, 3, 4, 5].

The Henry's Law constant (H) is strongly influenced by temperature. The relationship of H to temperature can be modeled by a van't Hoff-type relation. If the enthalpy change caused by dissolution of the compound in water is considered independent of temperature, the relation takes the following form:

$$\log H = \frac{-\Delta H^\circ}{RT} + K$$

where R = the universal gas constant, 1.987 kcal/kmol, °K; T = absolute temperature (°K); ΔH° = change in enthalpy due to dissolution of the compound in water (kcal/kmol); and K = constant. Henry's Law constants can thus be adjusted to account for temperature differences that may exist between literature values, pilot studies or scale-up design.

The transfer unit model (adapted from chemical engineering design methods) has received extensive use in recent years. Use of the transfer unit model is found in references [6 - 13]. Kavanaugh and Trussell [6] present a comprehensive design methodology and quantitative framework for evaluation of pilot plant studies using the transfer unit model. The general form of the equation is:

$$Z = \text{HTU} \times \text{NTU}$$

where Z = height of packing (m); HTU = height of a transfer unit (m); NTU = number of transfer units (dimensionless). HTU characterizes the efficiency of mass transfer from water to air:

$$\text{HTU} = \frac{L}{K_L a C_0}$$

where L = liquid loading rate ($\text{kg} \cdot \text{mole}/\text{m}^2 \cdot \text{hour}$); $K_L a$ = overall liquid phase mass transfer coefficient (hr^{-1}) and C_0 = molar density of water ($\text{kg} \cdot \text{mole}/\text{m}^3$).

The overall liquid phase mass transfer coefficient, $K_L a$ is a function of the specific compounds being removed, packing media, and liquid loading rate. Recent studies have also found that the air loading rate may also influence $K_L a$ [14]. Approximations of $K_L a$ can be obtained from empirical relationships such as the Sherwood-Holloway [15] or Onda-Takeuchi-Okumoto [16] correlations. However, pilot and lab scale testing proves to be the most reliable means of determining $K_L a$. Bishop and Cornwell [11] suggest that the transfer unit model be solved for $K_L a$ utilizing lab or pilot scale data and a reasonable estimation of Henry's Law constants.

The number of transfer units, NTU, is a characterization of the difficulty of removing a solute from the liquid phase. For dilute solutions and solutes which obey Henry's Law, the expression for NTU can be solved analytically. The equation for NTU is:

$$\text{NTU} = \frac{R}{R - 1} \ln \frac{(C_{in}/C_{out})(R - 1) + 1}{R}$$

where R = stripping factor (dimensionless); C_{in}/C_{out} = the ratio of solute influent to effluent concentration (dimensionless). The stripping factor, R , is defined as:

$$R = \frac{HG}{P_t L}$$

where H = Henry's Law constant (atm); P_t = the total pressure (atm);

G = air loading rate (kg moles/m²h); and L = the water loading rate (kg moles/m²h).

The relationship between stripping factor, R , and NTU for various solute removal percentages is illustrated on Figure 1. Note that R must be greater than 1 to get much removal, but increasing R beyond 4 or 5 will yield diminishing improvements. The ratio of solute influent to effluent concentrations, C_{in}/C_{out} , may be equal to actual stripper sample concentrations if the objective is to determine $K_L a$ or may represent proposed removal concentrations if the objective is to design a full-scale stripping column for a given removal efficiency.

Kavanaugh and Trussell [6] also present a design example which illustrates the calculations needed to evaluate allowable gas flow pressure drop and tower diameter.

Various other design models have been developed by other researchers which are not directly derived from the transfer unit model [2, 3, 11, 17]. Singley et al. [2] present a comprehensive design approach whereby compound equilibrium constants, H , are derived experimentally and the mass transfer coefficients are a function of Reynolds Number, Re . They conclude that the judgement as to whether a compound can be removed by air stripping must be based upon the Henry's Law constant and the mass transfer coefficient. Mass transfer coefficients that were developed were independent of the packing material and thus could be used with any packing provided the surface area per unit volume of packing is known. Scale up procedures and extensive cost analysis information are presented.

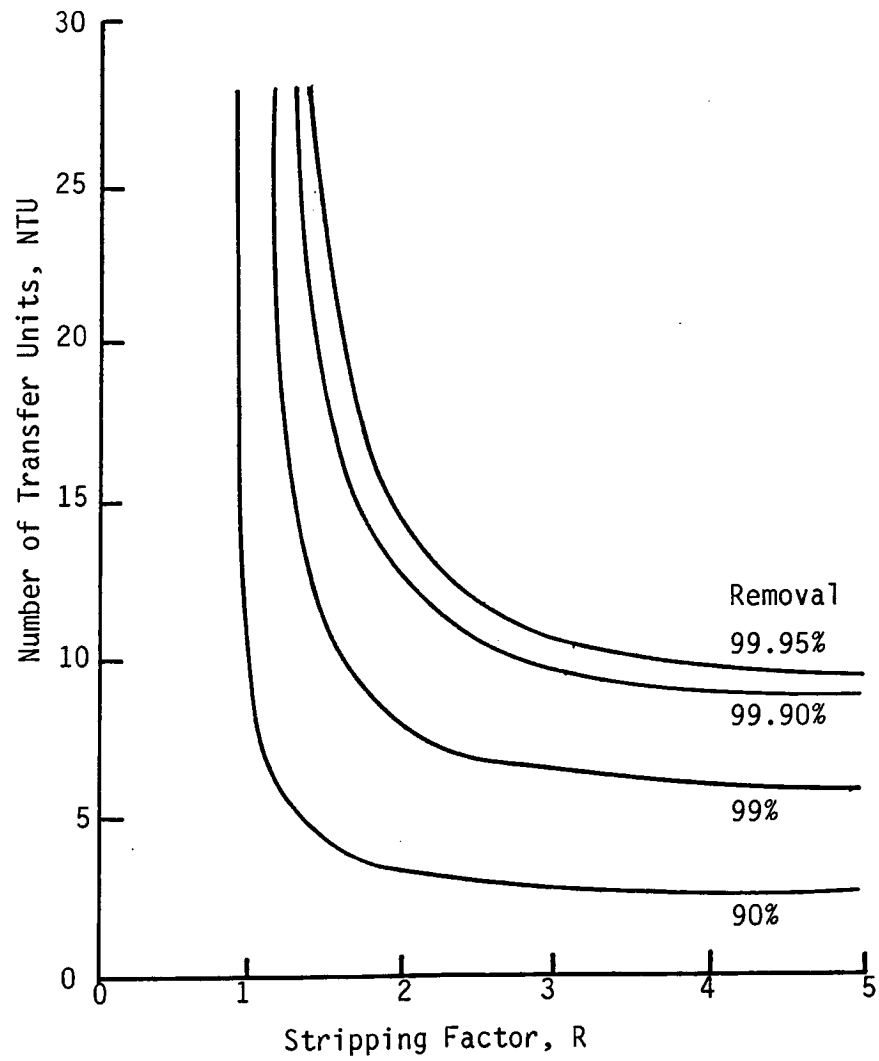


Figure 1. Relationship Between Stripping Factor and NTU.
from ref. [2]

McCarty et al., [17, 18] present a simple equilibrium model which suggests that for a given air to water ratio and Henry's Law constant there will be a corresponding lower concentration of solute in the effluent at equilibrium. A wide range of compounds were examined on a full-scale stripping tower.

Reference [3] presents a unique design approach which was used to plan and operate a full scale air stripping system to remove TCE. The design equation somewhat resembles the transfer unit model:

$$Z_t A = \frac{L}{K_L a} \frac{\ln \left[\frac{C_1}{C_2} - \frac{RT}{A_w H_c} \left(\frac{C_1}{C_2} - 1 \right) \right]}{1 - \frac{RT}{A_w H_c}}$$

in which Z_t = height of packing, m; A = cross-sectional area of tower, m^2 ; L = volumetric liquid flow rate, m^3/min ; C_1 = the inlet concentration, ppb; C_2 = the outlet concentration, ppb; R = the universal gas constant; T = the temperature, $^{\circ}\text{K}$; A_w = the volumetric air to water ratio and H_c = Henry's Law constant, $m^3 \text{ atm/mol}$. Using a generalized flooding and pressure drop diagrams (from ref 77), the volume of tower packing is calculated:

$$X = \frac{1}{A_w} (\rho_l / \rho_g)^{0.5}$$

in which X = packing volume, m^3 ; ρ_l = density of water, lb/ft^3 ; ρ_g = density of air, lb/ft^3 . The air loading rate, G , is determined:

$$G = \frac{(\gamma \rho_g \rho_l g_c)^{0.5}}{F \mu^{0.2}}$$

in which Y = gas pressure drop, inches water per foot; g_c = gravitational acceleration, 32.2 ft/sec^2 ; F = packing factor, ft^{-1} ; and μ = water viscosity, centipoise. Final calculations determine the tower diameter and packing depth.

Dyksen et al., [19] presents a summary of packed column pilot test data from 22 wells with nine different VOCs. Some pertinent conclusions were;

- 1) The VOC concentration affected the percent removal rate with higher VOC concentrations achieving slightly higher removal rates.

- 2) Higher removal efficiency was obtained if only one VOC was present in the water.

- 3) At lower liquid loading rates, the mass transfer characteristics generally correlated with the Henry's Law constant. The influence of multiple solutes on stripping rate has been assumed by most researchers to be negligible; however, no definitive studies have been conducted. Dyksen also concludes that pilot studies have proved to be necessary for developing design criteria for full scale treatment facilities.

At the time when air stripping towers were first being introduced for use in treating contaminated groundwaters, questions were often raised concerning the secondary effects of the VOC-laden air that exits the top of the stripping tower. Umphres et al., [20] briefly review the use of GAC adsorption for air pollution control. They recommend pilot studies and steam regeneration of the GAC. Some additional secondary effects which Umphres also discusses are the problems associated with

iron precipitation and removal of free and combined chlorine. It appears that for typical air stripping facilities significant losses of either free or combined chlorine will not occur.

Improving stripping tower performance can be approached by a number of ways other than increasing air to water ratios. Johnson et al., [21] conducted field studies that focused on raising the temperature of the water stream in order to remove methyl ethyl ketone, MEK. A series of batch runs at progressively higher temperatures produced favorable results. Zanetti et al., [22] elaborate on efficient liquid distributors and redistributors for improved performance. A particularly informative paper is presented by Chen [23], although directed toward the chemical engineering field, does provide the water treatment design engineer with an overview of some of the materials and equipment that are available for full-scale stripping towers. Subjects include support plates, liquid distributors, liquid redistributors, collectors, packing types and materials, troubleshooting problems and upgrading existing towers.

Studies investigating the mass transfer of volatile organic compounds by means of diffused aeration was done by Roberts and Dandliker [24] and Matter-Muller et al., [25]. Wood et al., [26] and O'Brien and Stenzel [27] show respectable performance of spray head aeration for a wide range of VOCs although the operating costs due to increased pumping pressures are a drawback. Dyksen et al., [28, 29] and Goers and Hubbs [30] present general comparisons of the various aeration techniques with a conclusion that packed columns are the most efficient and economical.

GAC Theory Models and Studies

Activated carbon adsorption was deemed the best available technology for VOC removal in the period immediately following the discovery of toxic organics in water supplies, late 1970s - early 1980s. Although massive volumes of literature exist on the applications and characteristics of activated carbon and other synthetic adsorbents some of the most relevant appearing in periodicals and research reports are presented here. In the experimental work done for this paper, activated carbon adsorption isotherms were prepared primarily as a basis of comparison with air stripping and to establish relative adsorbability of multi-solute solutions on one particular carbon type. Greater emphasis was placed on analyzing the variables present with air stripping systems. For this reason the literature review, results, discussions and conclusion regarding activated carbon are somewhat general.

The theories associated with activated carbon adsorption are found in many reference texts [1, 31] and general discussions are also found in periodical literature [32] and reports [33]. Many different models have been developed for characterizing single and multi-solute adsorption processes. McGuire and Suffet [32] present a brief overview of the following: mass transfer zone model (MTZ), Numeric model (MADAM, Michigan Adsorption Design and Application Model), Ideal Adsorbed Solution Theory (IAST), Polanyi Adsorption Potential Theory, Solvophobic Theory, Brunauer, Emmett, Teller (BET) and net adsorption energy concept. Weber [34] applied the MADAM model to seven toxic or

carcinogenic compounds found commonly in U.S. water supplies and found the model suitably predicted breakthrough profiles for fixed bed adsorbers. Crittenden et al., [35] applied the IAST theory to multi-solute mixtures of six VOCs and used it to predict competitive effects in waters having unknown background constituents. Crittenden et al., [36] also found that a simplified version of the MTZ model can be used to optimize design of fixed bed adsorber systems. Pilot and full-scale data are presented to demonstrate its validity.

The most widely used models for characterizing single solute adsorption are the Langmuir and Freundlich equations. The Langmuir model is based upon the hypothesis of single-layer adsorption. It assumes that maximum adsorption corresponds to a saturated layer of solute upon a single-layered adsorbent surface, that the energy of adsorption is constant, and that there is no transmigration of sorbates on the sorbent surface. The Freundlich model is an empirically based curve-fitting approach derived from the Langmuir equation. It is widely accepted for intermediate solute concentrations where it takes on the form:

$$X/M = KC_f^{1/n}$$

The logarithmic form of the equation gives a straight line plot with slope $1/n$ and intercept of $\log K$ for $C=1$.

$$\log X/M = \log K + 1/n \log C_f$$

in which $X = C_0 - C_f$ = the amount of solute adsorbed, C_0 = the initial solute concentration, C_f = the final equilibrium solute concentration,

M = the mass of sorbate in the reactor, K and $1/n$ are empirical constants. The isotherms presented in this report are plotted from the Freundlich model equation.

Several researchers have examined the removal of organics with adsorbents other than activated carbon [37-40]. In all cases, however, GAC was used as a reference for comparison. Snoeyink [37] concluded that resins are not applicable as a general adsorbent at water treatment plants because they generally too selective. Snoeyink [41] does however suggest that two different adsorbents in series may be used to optimize removal rates, compensate for competitive adsorption and improve regeneration economy.

Powdered activated carbon, PAC, is also used to remove VOCs, THMs, and SOCs; however, the method of application is entirely different from the use of GAC. Since PAC is commonly used for taste, odor and color removal at water treatment plants its usefulness is easily extended to include organic contaminants. Research into such applications are presented by Singley et al., [42], Miller [43] and in the American Water Works Association Research Foundation, Cooperative Research Report [33]. McGuire et al., [44, 45] examine the interactions among different water treatment unit processes in removing organic contaminants. Several process unit arrangements are presented which incorporate either PAC or GAC into the treatment scheme.

Combining Air Stripping with GAC

Since early 1980s researchers have recognized the benefits that could be gained from utilizing a series combination of air stripping

with activated carbon adsorption. However, it has only been recent studies and applications which have proven it to be an effective and economical system.

The principal advantages of air stripping followed by GAC adsorption are found in the ability of the system to treat waters containing multiple contaminants having different volatilities and adsorption characteristics. The two-stage process will provide added operational flexibility which can be useful in meeting treatment goals when influent concentrations change and for optimizing operating costs such as power and carbon regeneration or replacement.

As a general rule, air stripping is most effective with low-molecular weight, highly volatile compounds, while activated carbon adsorption is best with high-molecular weight compounds having low solubility. Therefore, pretreatment by air stripping will reduce the organic load on the carbon and may remove compounds competing for adsorption sites and reduce the problems associated with desorption.

Several references present general discussions of management and treatment alternatives for dealing with the presence of VOCs in groundwater supplies [46 - 48, 12, 62]. These references conclude that some groundwater contamination problems may be adequately treated by either GAC or air stripping alone; however, the combined processes usually offer the best solution when multiple contaminants are involved. The importance of pilot testing is also emphasized.

O'Brien and Stenzel [27, 49] compared the performance and costs of two different types of air stripping equipment, a conventional

packed tower and an induced draft spray head stripper. They point out that the packed tower stripper is most frequently coupled to pressure carbon adsorbers whereas the induced draft stripper easily adapts to a gravity flow carbon filter unit. The results of pilot testing indicate that the induced draft stripper did not achieve as high removal rates as the packed tower but the construction and operating costs were lower.

Amy et al., [50] determined that air stripping followed by activated carbon adsorption is a viable treatment method for groundwaters containing certain VOCs and dissolved organic matter. The study examined the removal of carbon tetrachloride (CT) and trichloroethylene (TCE) in the presence of dissolved organic carbon (DOC). They found that the DOC significantly reduced the adsorptive capacity of the VOCs onto activated carbon. The study clearly demonstrated that when air stripping is employed as a pretreatment process, there is a substantial reduction in carbon usage (as compared to "carbon only" treatment).

They also demonstrated that competitive adsorption is an important consideration with respect to the activated carbon step. Most of the competitive adsorption isotherms were based on CT or TCE concentrations of 40 to 50 ug/L. Slightly higher adsorption capacities were observed for high as opposed to low initial CT or TCE concentrations.

The results of the study [50] indicate that, at the concentrations examined, the removal of TCE will be the controlling

factor in the air stripping process while CT removal will control the carbon adsorption process. They suggest that possible advantages of coupled air stripping and activated carbon adsorption are: (1) reduced media depth at a given A:W ratio and liquid rate or (2) the use of a lower A:W ratio at a given media depth and liquid rate or (3) reduced liquid rate at a given A:W ratio and media depth.

McKinnon and Dyksen [51] report on the use of GAC and air stripping for removal of ethers and TCE from a municipal water well in Rockaway Township, New Jersey. The initial treatment system consisted of GAC adsorption; however, the ether compounds were achieving breakthrough within 4 to 6 weeks. A countercurrent packed column air stripper was added as a pretreatment step to the GAC. In July 1983, the GAC system was taken offline because of the excellent performance of the aeration system alone and because of diminishing contaminant concentrations in the influent.

EPA Regulations

On February 9, 1978, the USEPA published proposed regulations for the control of organic chemical contaminants in drinking water. Maximum Contaminant Levels (MCLs) for THMs and VOCs and proposed monitoring requirements for 59 organic compounds, of which more than half were VOCs. On March 4, 1982, the EPA published the "National Revised Primary Drinking Water Regulations, Volatile Synthetic Organic Chemicals in Drinking Water" which set MCLs for many water contaminants of concern. On November 13, 1985, EPA promulgated final Recommended Maximum Contaminant Levels (RMCLs) and proposed MCLs for 8 VOCs. Also

proposed in the announcement were monitoring requirements for 51 other unregulated compounds and proposed MCLs for 26 SOCs which include some pesticides. The 1986 amendments to the Safe Drinking Water Act (SDWA) have redefined RMCLs so that they are now known as Maximum Contaminant Level Goals (MCLGs). In the future MCLs and MCLGs must be proposed simultaneously and promulgated simultaneously.

July, 1987 the USEPA adopted final drinking water standards for eight VOCs which will be enforced beginning December 31, 1988 (see Table 1). The adopted standards are identical to the proposed MCLs set on November 13, 1985 with the exception of vinyl chloride and 1,4 dichlorobenzene. In the case of 1,4 dichlorobenzene, the USEPA had originally adopted an MCLG of 750 ug/l and proposed an MCL of 750 ug/l based on a determination that evidence of human carcinogenicity was inadequate. However, more recent health effects data in April 1987 prompted a change to an MCLG of zero and an MCL of 5 ug/l based on the chemical being classified as a probable human carcinogen. The final MCLG of 75 ug/l and MCL of 75 ug/l are based on a determination that 1,4 dichlorobenzene is a possible human carcinogen. Granular activated carbon and aeration are designated as the Best Available Technology (BAT) for meeting the VOC standards.

These eight new VOC standards and all future standards will apply to non-community water suppliers as well as community systems, and enforcement fines will be imposed for violations whether willful or not [52].

Table 1. USEPA Standards for VOCs in Drinking Water

Chemical	Final MCL (ug/l)
Trichloroethylene	5
Carbon tetrachloride	5
Vinyl Chloride	2
1,2 Dichloroethane	5
Benzene	5
1,4 Dichlorobenzene	75
1,1 Dichloroethylene	7
1,1,1 Trichloroethane	200

The USEPA will establish initial MCLs for 83 contaminants over the upcoming three-year period and thereafter MCLs for 25 new contaminants are to be added to the list every three years.

It is apparent from the above synopsis that there has been much progress in the regulatory arena; however, a certain amount of confusion and uncertainty has accompanied it. Water providers and engineers have been faced with contamination problems which demanded immediate action; however, the regulatory standards and BAT may not have been established. Thus many treatment systems have been designed and implemented to achieve removals based on rather arbitrary target effluent concentrations.

In 1974-75 the EPA sponsored an 80-city National Organics Reconnaissance Survey (NORS) to determine the presence of trihalomethanes (THMs), carbon tetrachloride and 1,2 dichloroethane in U.S. drinking water supplies. Widespread occurrence of THMs were found in water treatment plants following the chlorination stage. Very little evidence of carbon tetrachloride and 1,2 dichloroethane were found. Subsequently the EPA conducted the National Organics Monitoring Survey (NOMS) on 113 community water supplies to determine the frequency of occurrence of 23 selected VOCs, THMs and SOCs. The NOMS survey also detected the presence of at least 63 additional organic compounds [53, 54].

On February 9, 1978, the EPA published proposed amendments to the National Primary Drinking Water Regulations. The regulations included MCLs for THMs and prescribed granular activated carbon (GAC)

treatment for control of SOCs. Significant controversy and deliberation surrounded the issue for the next few years [55 - 61]. Issues included skepticism of the necessity of GAC treatment, validation of the toxicological health effects, cost of compliance, MCLs, and other treatment or management alternatives. By the early 1980s, air stripping had emerged as an effective alternative treatment method and GAC was gaining acceptance through research and use.

Analysis and Monitoring Techniques

Although the majority of literature regarding VOCs focuses on treatment system design, modeling and cost effectiveness, etc., laboratory sample analysis is also of great significance. Trace organics analysis has made tremendous advances in recent years in terms of equipment and procedures. It has become increasingly easier to classify, isolate, resolve, identify and quantify an ever-widening array of compounds. Minimum detection limits are gradually improving with development of new equipment and techniques. As the detection limit moves lower it will become increasingly difficult to meet MCLs of zero.

Trussell and Umphres [63], presented a very informative paper in 1978 which summarized the state of the art at that time. In 1981, the American Water Works Association compiled a collection of 19 papers dealing with analysis of organics in drinking water [64]. Stevens et al. [65], utilized GC-FID organics profiles for evaluating treatment processes designed for removal of VOCs and SOCs. Davenport et al. [66], suggest the use of a continuous monitoring instrument for general

organics. Such a device may be particularly useful for treatment systems utilizing activated carbon adsorption of highly-colored waters containing SOCs or VOCs.

Case Studies

A small collection of articles concerning the occurrence and treatment of groundwater contamination due to VOCs is represented by references [67 - 73]. Although the majority of these articles refer to sites located in the Eastern U.S., it is well known that significant groundwater contamination problems also exist in numerous other places throughout the country.

It is interesting to note the general trend of treatment approaches which occurred. Early treatment methods typically used GAC contactors; however, frequent and expensive carbon regeneration or replacement soon made GAC an unaffordable method. The solution in some instances was to add an air stripping tower for pretreatment [70]. Often the GAC could be taken offline completely because the aeration system was so efficient. Other more recent VOC treatment systems have simply used air stripping towers alone [68, 69].

EXPERIMENTAL PROCEDURES

The laboratory work that was required for this study consisted of; air stripping experiments, powdered activated carbon experiments (isotherms) and analytical testing. The compounds examined included Carbon tetrachloride (CT), Trichloroethylene (TCE) and para (1,4) dichlorobenzene (DCB).

Compounds Examined

Carbon tetrachloride (CCl_4) is an aliphatic hydrocarbon, nonpolar, hydrophobic and quite volatile. It has been used primarily as a solvent, dry-cleaning agent, fire extinguisher agent, grain fumigant and machinery degreaser in the early 1900s. Current industrial uses are in the manufacture of refrigerants (chloro-fluorocarbons-freons). Virtually all CCl_4 involved in dispersive uses is eventually released to the environment. It has been estimated that approximately 3.04 million tons of CCl_4 had been emitted to the environment as of 1975 [21]. In 1975, the National Organic Reconnaissance Survey (NORS) found 12.5 percent of finished drinking waters and 4.1 percent of raw waters had CCl_4 present. The occurrence of CCl_4 in raw waters is generally attributable to covert waste discharges or spills. Its occurrence in finished waters has also been traced to CCl_4 contamination of chlorine used in the water treatment process. Since 1970, carbon tetrachloride has been banned for all

use in consumer goods in the United States, and in 1978 it was banned as an aerosol propellant.

Principal toxic effects include central nervous system depression and cellular necrosis of the liver and kidneys. It is a suspected mutagen and carcinogen.

Trichloroethylene (C_2HCl_3) (TCE) is an aliphatic hydrocarbon, hydrophobic and moderately volatile. It is commonly considered to be an "average" compound with respect to removal from water by air stripping. It has been used primarily as a degreasing solvent in metal industries and a common ingredient in household products such as spot removers, rug cleaners, air fresheners, dry cleaning fluids, refrigerants and inhalation anesthetics. Annual production in 1980 was reported at 234,000 metric tons; however, stringent regulations have resulted in its declining use. TCE is probably the most common contaminant found in groundwater supplies due to leaking storage tanks, industrial spills, illegal dumping or improper disposal methods. Groundwater contamination in Tucson, Arizona has been traced to TCE disposal practices used by aircraft manufacturers in the 1950s. The disposal method used was considered acceptable at the time but has proven to be environmentally unsound nearly 30 years later.

TCE is readily absorbed into the bloodstream when ingested and its metabolites appear to have some moderate bioaccumulative properties. The National Cancer Institute (NCI) has concluded that TCE is a liver carcinogen in mice. Cases have been documented of liver and kidney damage in humans due to acute and chronic exposure to high doses of TCE.

Para-Dichlorobenzene ($C_6H_4Cl_2$) (1,4 DCB) is an aromatic hydrocarbon, hydrophobic and only slightly volatile. DCB is directly incorporated in a variety of consumer products and widely used as a reactive intermediate for production of a number of other chemicals. Its principal uses are as a pesticide in moth repellants, fumicide-bactericide, odor-suppressor, high pressure lubricant, as a component in insecticides and germicides and as a precursor in dye manufacture. Domestic sewage, industrial effluents, spills and wastes from insecticides, germicides and dye manufacturing are common sources of 1,4 DCB in the environment. 1,4 DCB has been detected in various water supplies of cities in the United States and overseas. It has been determined that 1,4 DCB is degradable by microorganisms, non-biological factors and sunlight.

The toxicological effects of 1,4 DCB are not well defined; however, recent USEPA regulatory actions were based on the conclusion that it is a possible human carcinogen.

Table 2 lists the common properties of the three compounds studied.

Air Stripping Experiments

The air stripping experiments examined the VOC removal capability of two different polypropylene packing media; 5/8" pall rings and 1" tripacks. A lab scale air stripping column, including the associated control and metering devices, was constructed. The air stripper was a conventional countercurrent packed column with devices to measure air and water flow rates as well as air pressure at the base

Table 2. Properties of Compounds Studied

Parameter	Carbon-tetrachloride	Trichloroethylene	p-Dichlorobenzene
Molecular Weight g/mol	153.8	131.5	147
Density, g/ml	1.59	1.46	1.46
Boiling Point, °C	77	87	173.4
Solubility, mg/L (@ 25°C)	800	1,100	49
Vapor Pressure mm Hg	91	74	1.0
Henry's Law Constant	1.2	0.49	0.1
Log K _{ow}	2.72	2.29	3.39
Diffusivity (cm ² /sec)	0.667	0.681	0.671

of the column. Figure 2 shows the equipment arrangement and critical dimensions.

The basic column consisted of a plexiglass tube of five inches inside diameter, and 72 inches length. At the base of the column was an internal wire screen which served as a support for the packing media and provided an inlet well for the air and an outlet sump for the effluent water. In order to prevent channeling of the water along the inside of the column walls, four redistribution rings were provided. The rings were made of plexiglass, five inches outside diameter and 3-3/4 inches inside diameter. They each have a vinyl wiper on the outside diameter to provide a tight seal against the column and a PVC skirt on the inside diameter to prevent water from channeling along the bottom surface of the ring and out to the column wall. The column has four sampling ports which allow removal of water samples from the column at the top, bottom and two intermediate depths of the packing. The influent water was distributed on top of the packing by a PVC nozzle that produces five small streams of liquid at an approximately even spacial distribution.

The source of water was from the domestic supply tap in the lab and the source of air was from the compressed air outlets also available in the lab. Influent water flow rates were regulated with a 5/8" ball valve and measured with a Wallace and Tiernan rotameter having a range of 0 to 3.4 gpm. Air flow was regulated with a 1/2" corporation stop valve and measured with a specially fabricated orifice meter having a range of 0 to 100 cfm.

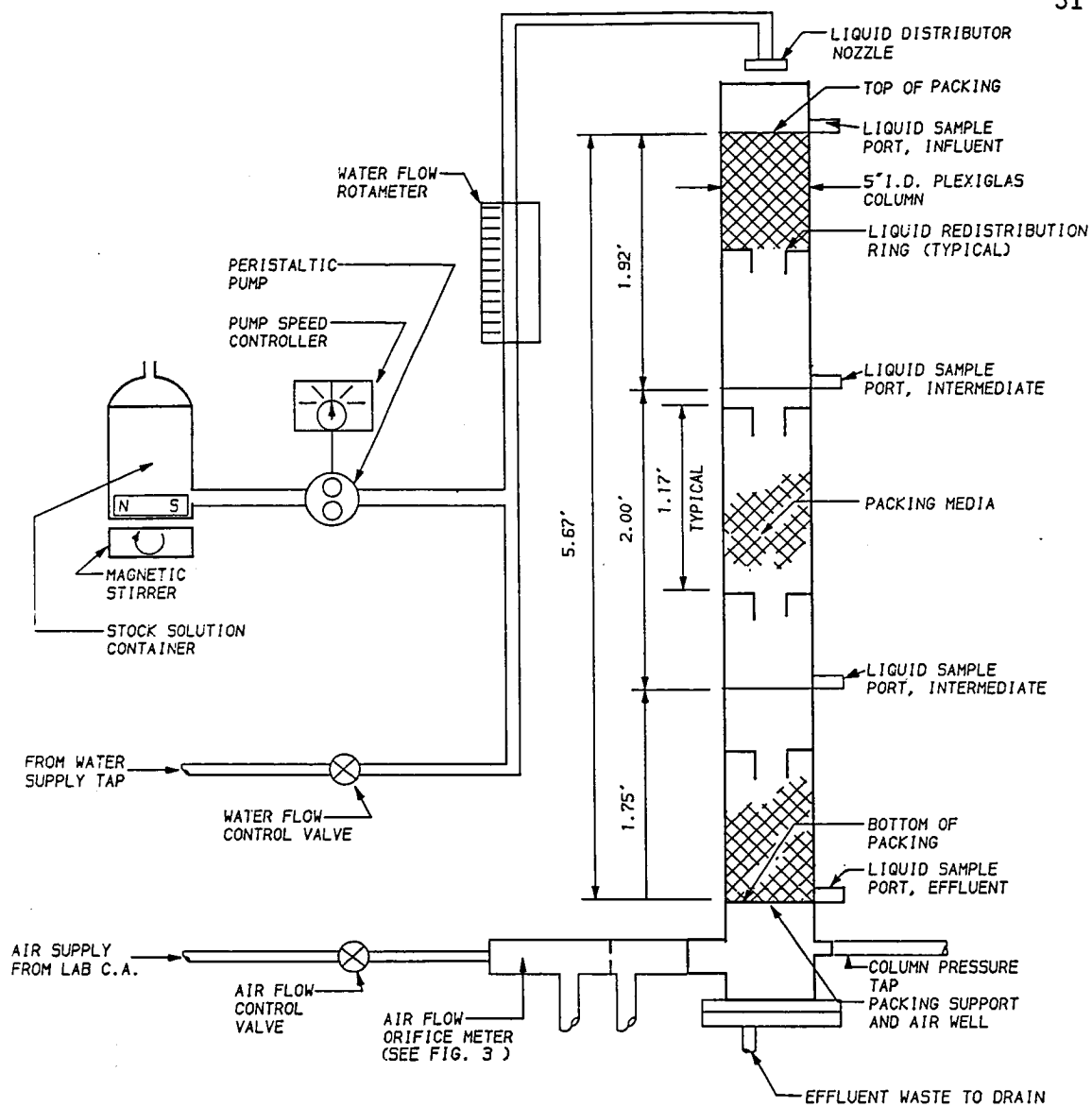


Figure 2. Diagram of Air Stripping Column and Appurtenances

The orifice meter (for air flow measurement) consisted of a 1-3/4 inch diameter PVC tube with a flat metal orifice plate having a 7/8" diameter orifice (see Figure 3). In order to measure the rate of air flow, the pressure upstream and downstream (p_1 and p_2) of the orifice plate are measured and applied to the following equation.

$$w = qd = CYA \sqrt{\frac{2g (p_1 - p_2)d}{1 - B^4}} \quad [\text{ref. 77}]$$

where:

- w = weight of discharge (pounds/second)
- q = volume of discharge (cubic feet/second)
- d = density of air (0.075 pounds/cubic foot)
- B = ratio of throat diameter to pipe diameter
- A = area of orifice opening (square feet)
- C = coefficient of discharge (0.60)
- Y = expansion factor (0.97)
- g = gravitational acceleration (32.2 feet/second²)
- p_1 = pressure upstream of orifice (pounds/square feet)
- p_2 = pressure downstream of orifice (pounds/square feet)

The terms C and Y were determined by additional equations in reference [77]. To facilitate measurement of p_1 and p_2 a differential pressure manometer gauge was used (Dwyer Model 2002) for the range of 0 to 2 inches of water column. Beyond 2 inches water column it was necessary to utilize a common water column manometer which would allow measurement to more than 8 inches of water column. Figure 4 is a plot of the relationship between $p_1 - p_2$ and CFM utilizing the above-described orifice meter and orifice flow equation.

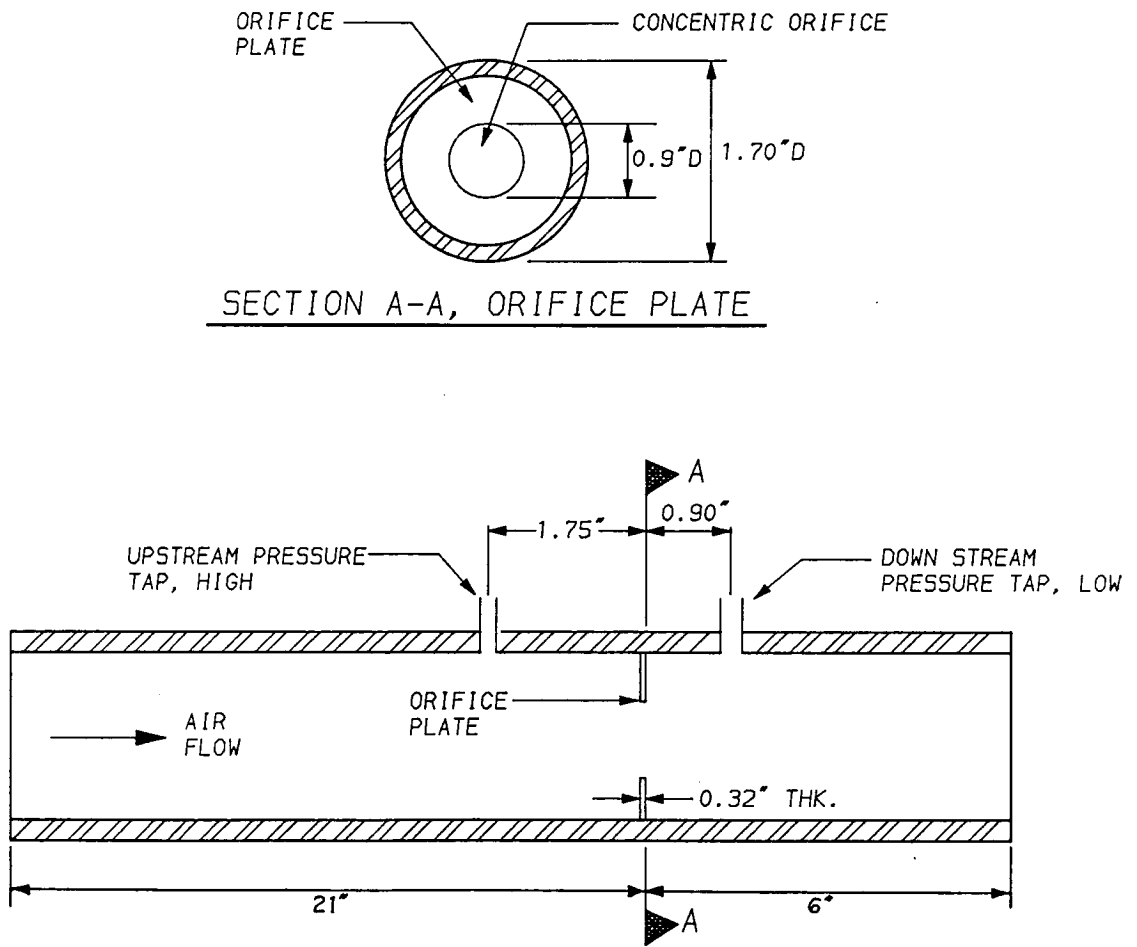


Figure 3. Orifice Meter Details and Dimensions

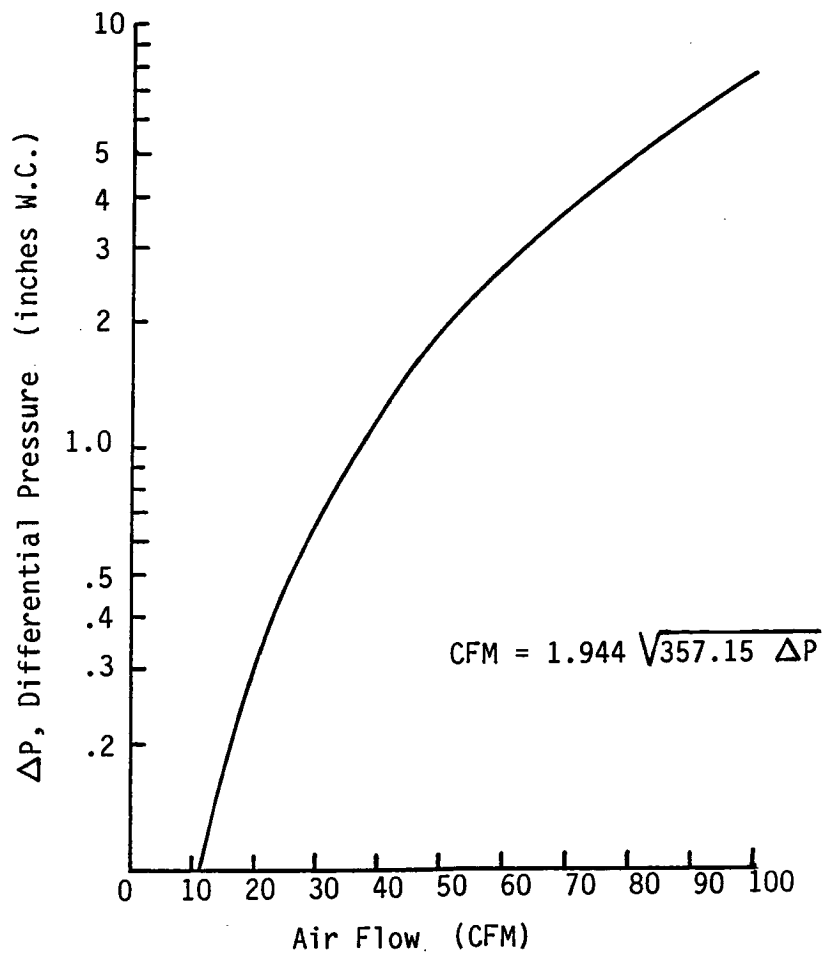


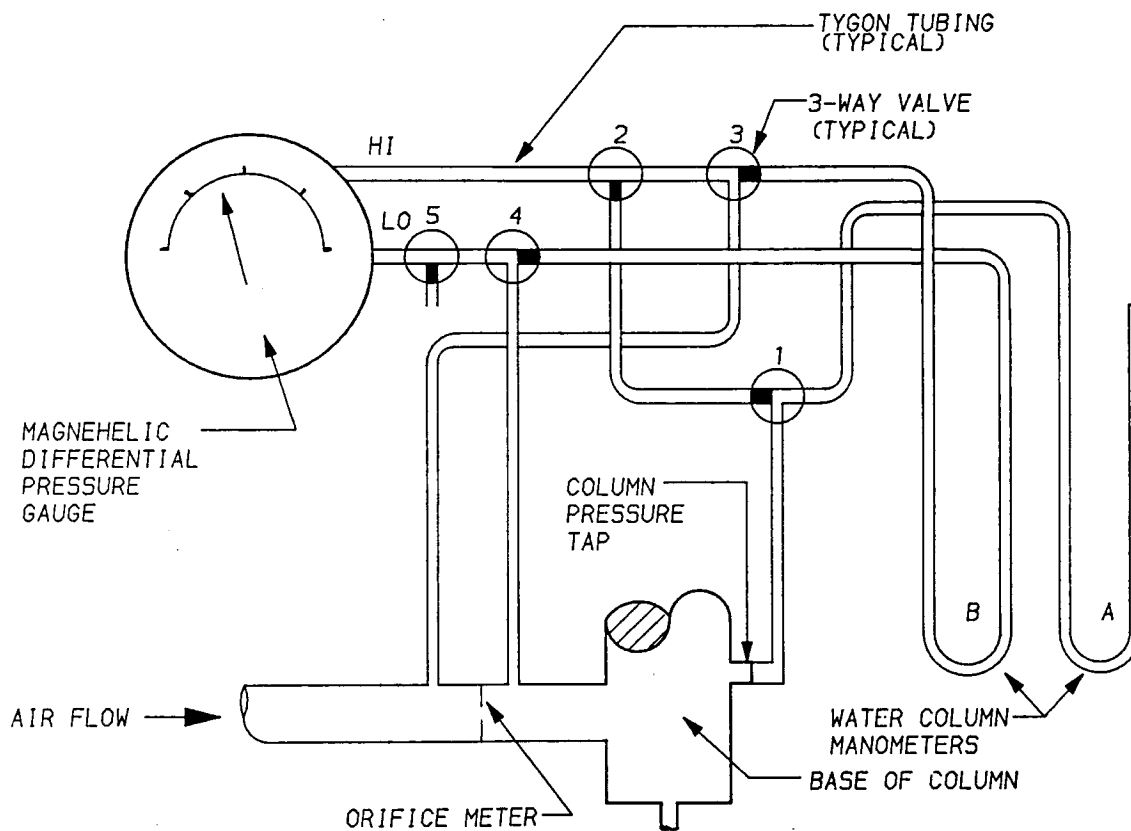
Figure 4. Orifice Flow Meter, Differential Pressure vs. CFM.

The air pressure within the air well at the bottom of the packing was measured by utilizing a small diameter tap through the column wall. The pressure was measured by the differential manometer gauge for pressures up to 2 inches water column or by the water column manometer for pressures greater than 2 inches water column. The measurement of column inlet pressure was coordinated with measurement of air flow rate by means of a network of pressure tubing and five control valves. Figures 5 and 6 show the piping schematics and the valve positions required in order to achieve the appropriate connection between the primary element and the respective manometer or gauge.

The solutes of concern (CT, TCE and DCB) were introduced into the influent water stream by means of a concentrated stock feed solution and a peristaltic pump. The stock feed solution was contained in a 2 liter aspirator bottle with a rubber membrane interior bladder, such that it prevented the air from contacting the surface of the stock solution as it was being withdrawn from the bottom outlet. This helped to minimize volatilization of the stock solution which could affect the influent solute concentration. The peristaltic pump (Cole Parmer 7553-10) was equipped with a variable speed controller and was calibrated at flow rates from 0.2 to 1.0 liters per hour.

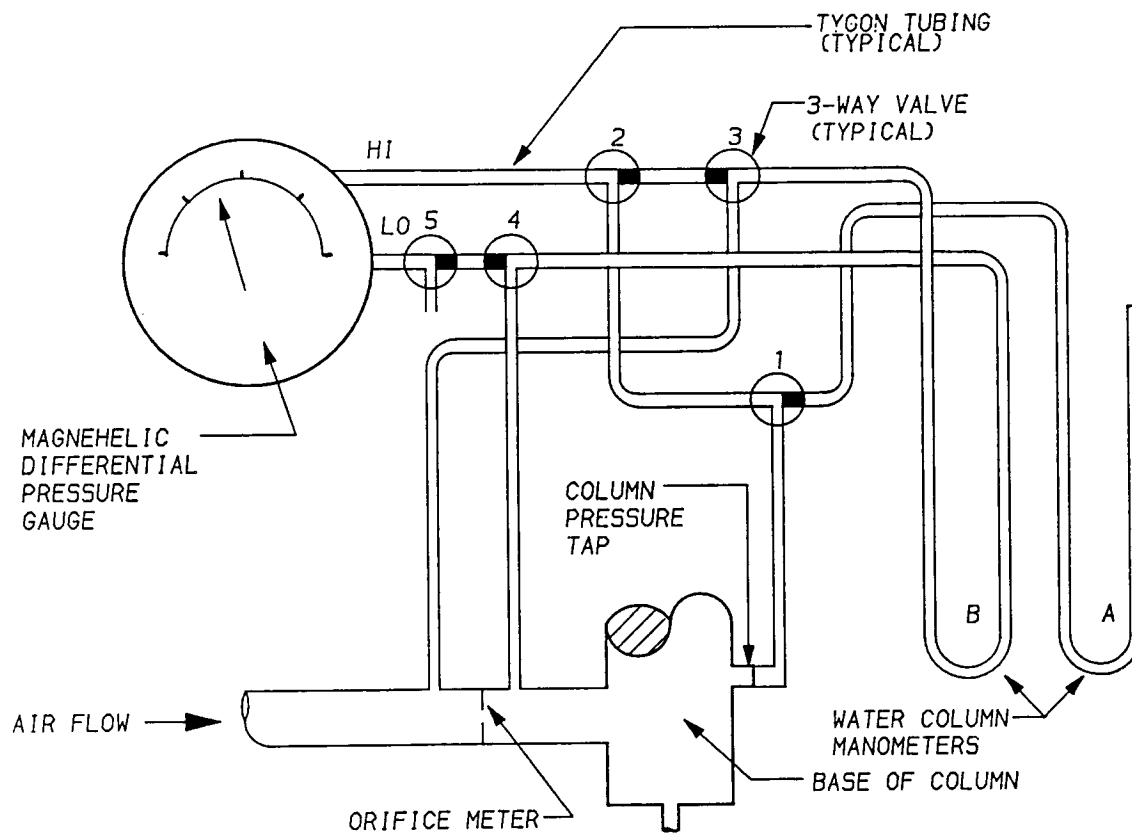
Operational flexibility of the air stripper included;

- 1) Liquid loading rate at 0 to 3.4 gpm
 - 2) Air loading rate 0 to 100 cfm
- and 3) Stock solution feed rate 0.2 to 1.0 liters/hour.



Orifice Meter = Magnehelic Gauge
 Column Pressure = Water Column Manometer A

Figure 5. Air Flow Meter and Column Pressure Gauges,
 Valve Positions A.



Orifice Meter = Water Column Manometer B
 Column Pressure = Magnehelic Gauge

Figure 6. Air Flow Meter and Column Pressure Gauges
 Valve Positions B.

Each of the two packing media were tested for the hydraulic characteristics of air flow rate vs. liquid loading rate vs. column pressure rise. Figures 7 and 8 illustrate the results. The pall rings exhibited "flooding" at various loading rates; however, the tripacks would not flood within the operational limits of the air stripper. After determining the packing hydraulics, a specific range of loading/flow rates was selected for the test stripping runs. A constant liquid loading rate of 7.3 gpm/square foot and air to water ratios from 0:1 to 314:1 (cfm:cfm) were used for the pall rings. The tripacks were tested at a liquid loading rate of 13.9 gpm/square foot and air to water ratios from 0:1 to 205:1. The rates and ratios were selected within the optimum hydraulic performance range of each packing media type.

Operation of the air stripping column involved the following general procedures;

- 1) Connect all piping for air, water, stock feed, and metering apparatus.
- 2) Prepare stock feed solution.
- 3) Set constant water flow rate as desired.
- 4) Set stock feed rate (peristaltic pump) as desired.
- 5) Set air flow rate as desired.
- 6) Allow system to operate for several minutes to reach equilibrium.
- 7) Collect samples at each sample port.
- 8) Adjust air flow rate to next setpoint and collect additional samples.

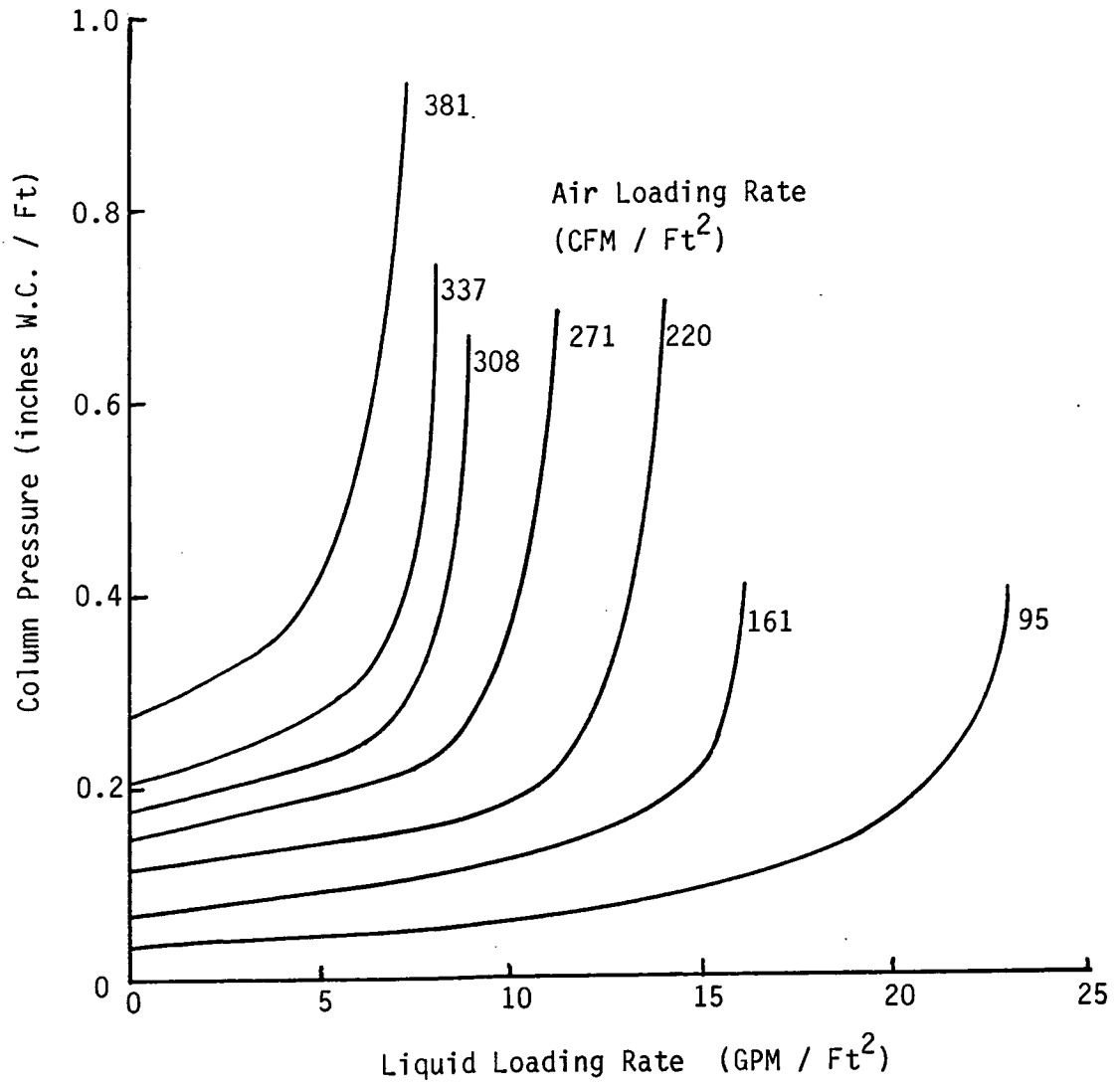


Figure 7. Column Hydraulics for Pall Rings Media.

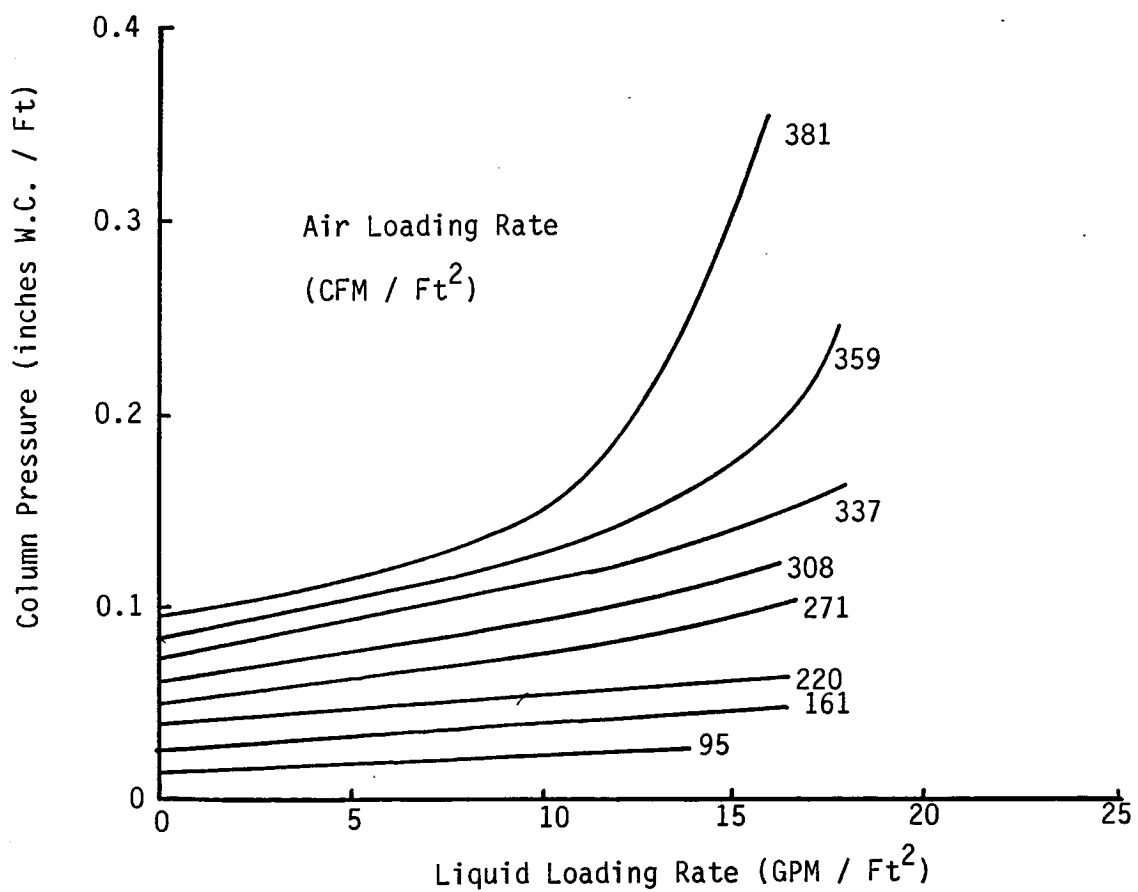


Figure 8. Column Hydraulics for Tripacks Media.

Powdered Activated Carbon Experiments

The activated carbon experiments consisted of a series of "bottle point" tests to establish various adsorption isotherms for CT, TCE and DCB. The following procedures were used; stock slurry solutions of powdered activated carbon were prepared in the following concentrations (mg/ml); 50, 10, 5, 0.5, and 0.1. A commercially available carbon, Filtrasorb 400 (less than 325 mesh) as manufactured by the Calgon Corporation was used in the isotherm experiments and is hereafter referred to as PAC. Filtrasorb 400 is a bituminous coal-based activated carbon. It has a BET surface area of $1,100 \text{ m}^2/\text{g}$, a particle density of 0.811 g/cm^3 , a pore volume of $0.7570 \text{ cm}^3/\text{g}$ and an average pore diameter of 27.5 angstroms. The appropriate carbon dosages for the isotherm experiments were obtained by pipetting a given amount from the stock slurry solutions into 160 ml serum vials.

Adsorption isotherms were conducted for combined solutes (CT, TCE and DCB) at the concentrations and sorbent doses indicated on Tables 3, 4 and 5. Two stock solutions of combined CT, TCE and DCB were prepared such that a 8 ul injection into each of the 160 ml serum vials would yield the desired initial sorbate concentration. A high solute concentration of approximately 100, 80 and 70 ug/l for CT, TCE and DCB, respectively, was used for the first phase of adsorption experiments to simulate non-airstripped water. The second phase of the experiments used approximate concentrations of 10, 20 and 40 ug/l for CT, TCE and DCB, respectively, to simulate pretreated (air stripped) water. The 160 ml serum vials containing the PAC and solutes were

**Table 3. Isotherm Data for CT in Presence of
TCE and 1,4 DCB on Activated Carbon**

Dose = M (mg/l)	C_f (ug/l)	$C_0 - C_f = X$ (ug/l)	X/M (mg/g)
0	7.3	--	--
1	7.3	--	--
2	5.1	2.2	1.10
5	4.9	2.4	0.48
10	4.1	3.2	0.32
20	2.2	5.1	0.26
33	1.4	5.9	0.18
0	134.0	--	--
1	131.6	2.4	2.40
2	--	--	--
5	120.7	13.3	2.66
10	116.9	17.1	1.71
20	--	--	--
33	103.8	30.2	0.92

$k = .13$

$1/n = .83$

**Table 4. Isotherm Data for TCE in Presence of
CT and 1,4 DCB on Activated Carbon**

Dose = M (mg/l)	C_f (ug/l)	$C_0 - C_f = X$ (ug/l)	X/M (mg/g)
0	13.2	--	--
1	13.2	--	--
2	10.5	2.7	1.35
5	7.9	5.3	1.06
10	4.4	8.8	0.88
20	1.7	11.5	0.58
33	1.1	12.1	0.37
0	125.4	--	--
1	121.3	4.1	4.10
2	--	--	--
5	96.6	28.8	5.76
10	82.2	43.2	4.32
20	43.3	82.1	4.10
33	24.7	100.7	3.05

$k = .40$

$1/n = .56$

Table 5. Isotherm Data for 1,4 DCB in Presence of
CT and TCE on Activated Carbon

Dose = M (mg/l)	C_f (ug/l)	$C_0 - C_f = X$ (ug/l)	X/M (mg/g)
0	19.8	--	--
1	15.2	4.6	4.6
2	8.4	11.4	5.7
5	0	0	0
10	0	0	0
20	0	0	0
33	0	0	0
0	59.1	--	--
1	37.4	27.7	27.7
2	--	--	--
5	15.4	43.7	8.7
10	12.0	47.1	4.7
20	11.4	47.7	2.4
33	9.5	49.6	1.5

$k = .010$

$1/n = 2.41$

placed on a gyratory shaker for 24 hours at a constant laboratory temperature of 25°C. Two bottles without any PAC were prepared in an identical manner to serve as controls for each isotherm.

The bottles were then each placed into a centrifuge at 1,875 rpm for a minimum of 15 minutes, causing the PAC to settle to the bottom of the bottles. Using a gas-tight 60 ml syringe, 120 ml of centrate was transferred to 120 ml serum vials and immediately capped with teflon septa. It is estimated that volatilization losses due to the liquid transfer procedure were less than 5% [40].

Analytical Testing

Analysis of the samples for the air stripping and PAC isotherms was accomplished by conventional gas chromatography separation using a liquid/liquid pentane extraction method. A Hewlett Packard 5790A with a mega bore capillary column (DB-1) and an electron capture detector (ECD) were connected to an HP 3390A reporting integrator. The following settings were utilized for isothermal GC operation:

Injector temperature	250°C
Detector temperature	280°C
Carrier gas flow rate	58-60 ml/min
Column pressure	8-12 psi

Typical sample analysis involved a 5 ml injection (via syringe) of pesticide-grade n-pentane (Fisher Chemical Co.) into the 120 ml serum vial with 5 ml of solution being displaced through an outlet syringe. The sample was then shaken for 2 minutes and a 1 ul aliquot of the pentane phase was removed and injected into the GC. At least one

additional injection was also made from each sample for statistical accuracy.

Prior to sample analysis, calibration solutions were prepared and injected in order to establish the appropriate response factors for quantification of chromatographic peaks. A reasonably good linear response was obtained for the range of concentrations to be examined for each of the subject compounds. TCE and CT samples were analyzed separately from the DCB samples due to difficulties in establishing a single accurate and repeatable separation method for the three compounds. All air stripping and PAC experiments were conducted using parallel samples for this reason. CT and TCE were analyzed at an oven temperature of 27°C and DCB was analyzed at an oven temperature of 100°C. Sample analyses for the air stripping experiments were conducted immediately following the stripper operation. PAC samples were analyzed upon completion of the 24 hour contact period and subsequent liquid transfer procedures.

RESULTS AND DISCUSSION

Air Stripping Experiments

The results of the air stripping experiments reflect only VOC removal which occurred in the packing media itself and do not measure or consider VOC removal due to end effects at the influent flow distributor or effluent air well of the column. Data simulation employs the transfer unit model for analysis of media-related VOC removal. Additionally, some comparisons of the liquid and air loading characteristics between the pall rings and tripacks media are also determined.

Experiments 1 through 5 utilized pall rings with influent concentrations for the VOCs of approximately 40 ug/l for CT, 50 ug/l for TCE and 160 ug/l for 1,4 DCB. The liquid loading rate was 1 gpm ($.136 \text{ gpm/ft}^2$) and air to water ratios ranged from 0 to 314 (vol:vol). Experiments 6 through 9 utilized pall rings with lower influent VOC concentrations of approximately 15 ug/l for CT and 40 ug/l for TCE. The results of the 1,4 DCB are not included because of analytical problems which yielded invalid results. The liquid loading rate was 1 gpm ($.136 \text{ gpm/ft}^2$) and air to water ratios ranged from 0 to 314 (vol:vol). Experiments 10 through 14 utilized 1" size polypropylene tripacks with influent concentrations for the VOCs of approximately 100 ug/l for CT, 80 ug/l for TCE and 70 ug/l for 1,4 DCB. The liquid loading rate was 1.9 gpm ($.259 \text{ gpm/ft}^2$) and air to water ratios ranged from 0 to 205 (vol:vol).

Although two different packing media were utilized, the test parameters such as liquid loading, A:W and influent VOC concentrations were not consistent thus direct comparisons of removal efficiency cannot be made. However, valid comparison can be made with respect to media air loading resistance, liquid and air loading limits and relative mass transfer coefficients.

Mechanical specifications for the two packing media per the manufacturers' literature are shown in Table 6. It is apparent that the pall rings are a generally smaller size as evidenced by the number of units per cubic foot. The surface area, void space and unit weight are comparable. The greatest difference between the two is found in the packing factors. The packing factor is generally representative of the pressure drop characteristics for the air flow through the media. A high packing factor indicates high air flow resistance hence higher blower costs. The pall rings exhibited a significantly higher pressure drop than the tripacks at all air/liquid loading rates examined. Refer to Figures 7 and 8 for graphical plots of the column pressure vs. liquid loading rate at various air loading rates. Note that it was possible to achieve "flooding" at all loading rates examined for the pall rings whereas the tripacks could not be flooded due to geometrical and hydraulic limitations of the experimental stripping column. In general, the tripacks media, having a lower packing factor, can handle greater liquid and air loading rates than the pall rings.

The liquid loading rates used for each packing media were selected based upon the results of the column hydraulics tests. It was

Table 6. Mechanical Specifications for Packing Media

<u>Type</u>	<u>Pall Rings</u>	<u>Tripacks</u>
Manufacturer	Norton	Jaeger Tri-Packs, Inc.
Material	Polypropylene	Polypropylene
Nominal Size (in.)	5/8	1
Surface Area (ft ² /ft ³)	104	85
Packing Factor (ft ⁻¹)	97	28
Void Space (%)	87	91
Weight (lb/ft ³)	7.25	6.20
Units per cubic foot	6,050	2,600
Cost (\$/ft ³)	NA	37

desired to use a liquid loading rate which would approach a flooding condition at the highest air loading rate. This was more relevant for the pall rings tests than it was for the tripacks; however, Figure 8 shows a pronounced column pressure rise at the highest air flow rate for the tripacks. The liquid loading rates used are generally comparable to those used for pilot and full-scale systems [3, 9, 13, 49].

The results of the VOC air stripping experiments are shown on Tables 7 - 16 and Figures 9 - 48. These Tables and Figures are located in the Appendix, Tabulations and Graphical Plots of Results of the Air Stripping Experiments. The data has been analyzed and presented in a number of ways;

(1) Fraction removed relative to top of packing vs. A:W (Table 8, Figures 9 - 11, 24, 25, 34 - 36).

(2) Fraction removed per foot of packing relative to the top of packing vs. A:W (Table 8, Figures 12 - 14, 26, 27, 37 - 39).

(3) Fraction removed per foot of packing for each increment between sample ports vs. A:W (Table 8, Figures 15 - 17, 28, 29, 40 - 42).

(4) Liquid mass transfer coefficient relative to top of packing vs. A:W (Table 15, Figures 18 - 20, 30, 31, 43 - 45).

(5) Liquid mass transfer coefficient for each increment between sample ports vs. A:W (Table 16, Figures 21 - 23, 32, 33, 46 - 48).

Pall Rings - Experiments 1-5

Figures 9 - 11 show that CT achieved the highest removal of the compounds tested at all A:W ratios. This result was consistent with the relative volatility of the three compounds. For both CT and TCE most

removal occurred at A:W ratios of less than 75:1 with very little additional removal at higher A:W ratios or packing depths greater than four feet. The effects of increasing A:W ratios is most evident within the first two feet of packing, showing a steady increase in removal as A:W ratios increase for both CT and TCE. As expected, 1,4 DCB shows a significantly lower removal rate with a leveling-out occurring at A:W ratios of about 150:1. The effects of increasing packing depth and A:W ratios are more evident with the 1,4 DCB samples than with CT or TCE.

Figures 12 - 14 more clearly define the effects of packing depth and A:W on the removal rates. The CT removal is slightly better than TCE within the first two feet of packing with almost identical removal rate per foot at packing depths beyond four feet and A:W ratios greater than 75:1. Figures 15 - 17 show removals on an incremental basis between sample ports. In contrast to Figures 9 - 11, all of the compounds indicate an increase in removal per foot of packing from A:W ratios of 0:1 to 75:1 and then a gradual decline at the higher A:W ratios. Figures 18 - 20 show the calculated liquid mass transfer coefficient, $K_L a$, relative to the top of packing, associated with each experiment sample. The $K_L a$ for CT is expectedly higher than TCE or 1,4 DCB. There is a steep increase in $K_L a$ of CT and TCE for A:W ratios from 0:1 to 75:1 with a gradual increase at higher A:W ratios. The $K_L a$ for 1,4 DCB is markedly lower than CT and TCE but does show a trend similar to that of CT and TCE.

Figures 21 - 23 show the calculated liquid mass transfer coefficient, $K_L a$, on an incremental basis between sample ports. The results are very erratic and do not display any identifiable trends.

Pall Rings - Experiments 6-9

These experiments examined CT and TCE removal at a lower concentration range than experiments 1 through 5. Figures 24 and 25 show the fraction removed relative to the top of packing is greater for CT than TCE at low A:W ratios but are almost identical at higher A:W ratios. The removal levels off at A:W ratios greater than 75:1 and packing depths greater than four feet. Figures 26 and 27 show the fraction removed per foot of packing relative to the top of packing. The conclusions are generally the same as for Figures 12 - 14. Figures 28 and 29 show the fraction removed per foot of packing on an incremental basis between sample ports. The results are generally the same as for Figures 15 - 17.

Tripacks - Experiments 10-14

Figures 34 - 36 show that CT achieved the highest removal of the compounds tested at all A:W ratios, as expected. For both CT and TCE most removal occurred at A:W of less than 40:1. Progressively greater removal of CT and TCE as A:W ratios increases within the first two feet of packing. At greater depths the removal levels out. As anticipated, 1,4 DCB shows a significantly lower removal; however, unlike the pall rings, there is not a leveling out within the A:W ratios tested.

Figures 37 - 39 show the fraction removed per foot of packing relative to the top of packing. CT achieved notably higher removal than TCE within the first two feet of packing. A leveling out of CT and TCE removal is evident for A:W ratios greater than 39:1 and removals are nearly equal with CT slightly above TCE for packing depths

of about four feet. For 1,4 DCB, the data shows a gradual increase in removals as the A:W ratio and packing depth increase. Figures 40 - 42 show the fraction removed per foot of packing on an incremental basis between sample ports. The results are rather erratic but all compounds exhibit an increase in removal rate for A:W ratios from 0:1 to 39:1 with a gradual decrease as A:W ratios further increase.

Overall Comparisons - Experiments 1-14

Among the five different result analyses, additional variables which may be compared are:

- (1) Tripacks vs. pall rings
- (2) CT vs. TCE vs. DCB removal
- (3) High vs. low CT, TCE concentrations

In terms of VOC removal relative to the top of packing, the pall rings showed consistently greater removal than the tripacks at all A:W ratios and packing depths. Also there was a greater differential in the amount of removal versus A:W ratio for tripacks than for pall rings at packing depths greater than four feet. The 1,4 DCB showed significantly greater removal with pall rings than tripacks. The pall rings exhibited a leveling out at higher A:W ratios and packing depths whereas the tripacks showed a still-climbing trend. Experiments 1 through 9 established that both CT and TCE will achieve greater removal rates at high concentrations than at low concentrations. This is probably due to a greater concentration gradient (driving force) for film transfer, at the higher concentration. There is also a greater differential in removal versus packing depth for low concentration than at high concentration.

The evaluation of the results for fraction removed per foot of packing relative to the top of packing also exhibited the same results as those described above.

In terms of the fractions of VOC removed incrementally between sample ports, the data points were very erratic. It does appear, however, that there is a trend which may be described as a gradual decline in the removal per foot of packing at the higher A:W ratios. This was generally observed for all compounds, concentrations, packing depths and packing types.

The comparison of $K_L a$ values between pall rings and tripacks should consider the difference of packing surface area, a , for each packing. Pall rings has $104 \text{ ft}^2/\text{ft}^3$ and tripacks has $85 \text{ ft}^2/\text{ft}^3$. Thus the overall liquid mass transfer coefficient K_L may be determined by factoring out the surface area term, a .

The $K_L a$ values relative to the top of packing (Figures 18 - 20, 30, 31, 43 - 45) presented many interesting comparisons: there is a wider difference in the relative $K_L a$ value of all three compounds for the tripacks than for pall rings. Experiments 1 through 9 for CT and TCE showed that the $K_L a$ was generally lower for the low concentrations than for high concentrations. A consistent upward trend in $K_L a$ was observed as A:W ratio and packing depth increase for all packings, VOCs, and concentrations. An additional observation shows that for CT, the tripacks had a higher $K_L a$ than pall rings, for TCE, the $K_L a$ for tripacks and pall rings were very similar and for 1,4 DCB the $K_L a$ for pall rings was higher than tripacks. In other words, the tripacks are

more effective for highly volatile compounds and the pall rings are more effective for less volatile compounds.

The $K_L a$ values on an incremental basis between sample ports (Figures 21 - 23, 32, 33, 46 - 48) exhibited very erratic data points. Some general trends and comparisons are evident however: CT shows a higher $K_L a$ for tripacks than for pall rings at A:W ratios greater than 40:1.

Experiments 6 through 9 show very close $K_L a$ values for CT and TCE (Figures 32 and 33) with a steady upward trend as A:W ratios increase to about 225:1 and then there is a downward trend to 314:1. Some of the other results for CT and TCE (Figures 21, 22, 46, 47) shows similar $K_L a$ declines at the highest A:W ratios.

Activated Carbon Experiments

The activated carbon experiments consisted of a series of "bottle point" tests for combined solutes (CT, TCE and DCB) at low and high relative concentrations. Single solute isotherms were not conducted; however, the results of other researchers [33, 40, 42, 50] may be useful for establishing the extent of competitive adsorption in this study. Also the effects of dissolved organic carbon (DOC) on adsorption capacity were not analyzed. Since the same water source was used for all tests it is assumed that DOC was similar for all tests and its competitive adsorption effects are constant for all tests. The DOC of the tap water was not determined and the pH of the tap water was not measured.

The results of the adsorption isotherm tests are plotted per the Freundlich equation on Figures 49, 50, and 51 and the data values are listed on Tables 3, 4, and 5. The results were generally as expected with TCE being more strongly adsorbed than CT at all concentrations. DCB, however, exhibited a very steep $1/n$ slope which resulted in a very high adsorption capacity at high equilibrium concentrations and very low capacity at low equilibrium concentrations.

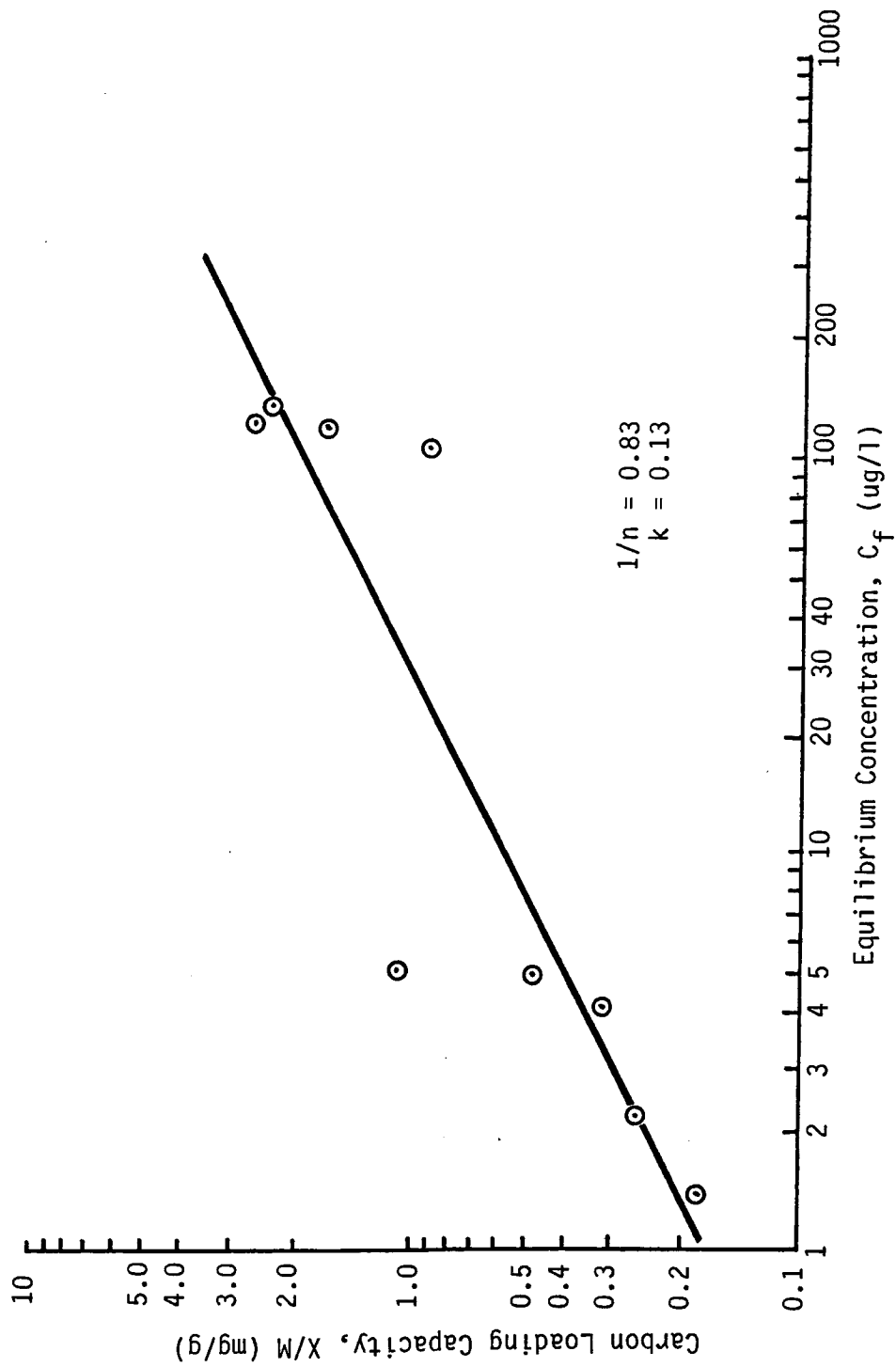


Figure 49. Freundlich Isotherm of CT in Presence of TCE and 1,4 DCB on Activated Carbon.

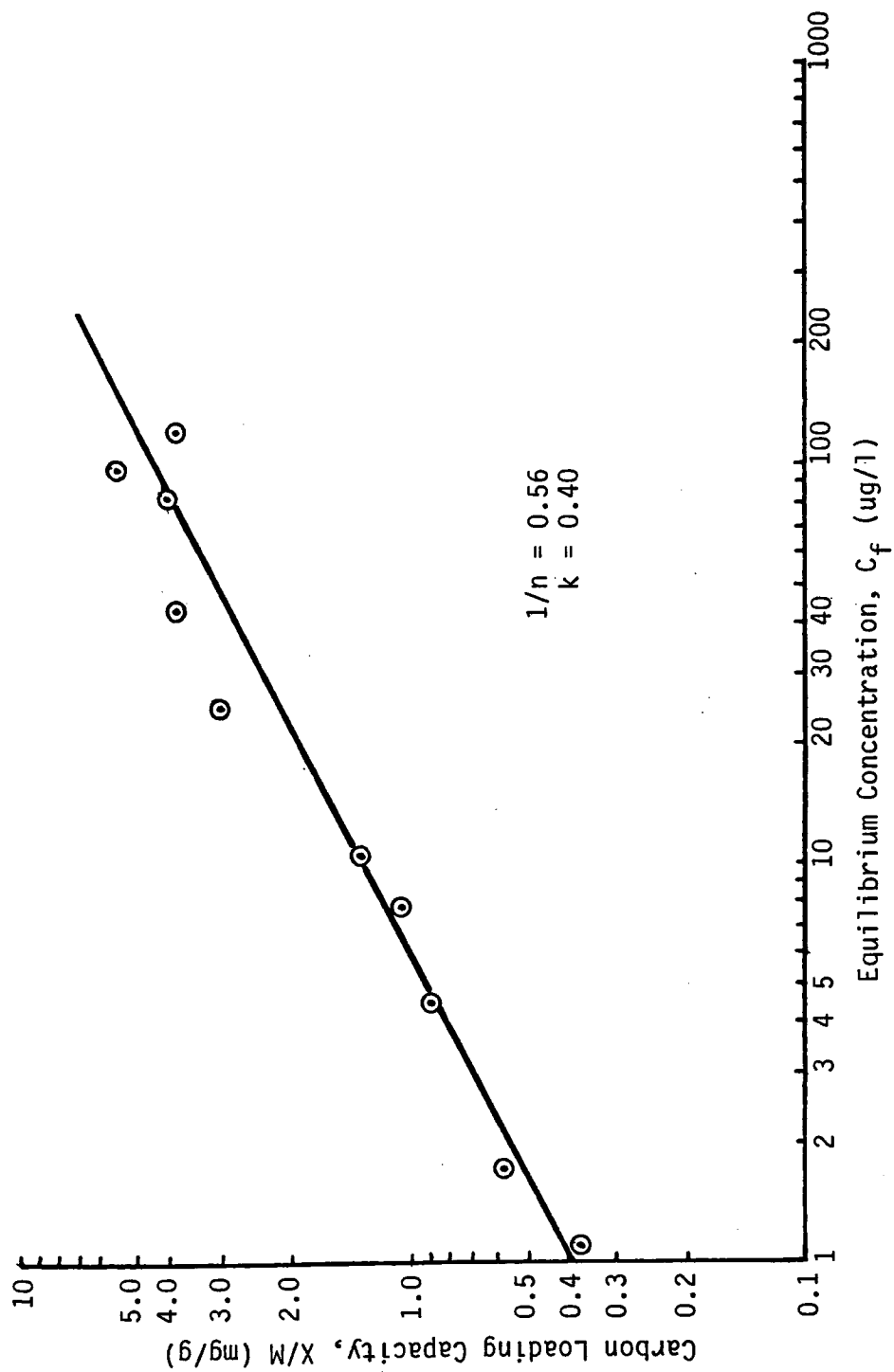


Figure 50. Freundlich Isotherm of TCE in Presence of CT and 1,4 DCB on Activated Carbon.

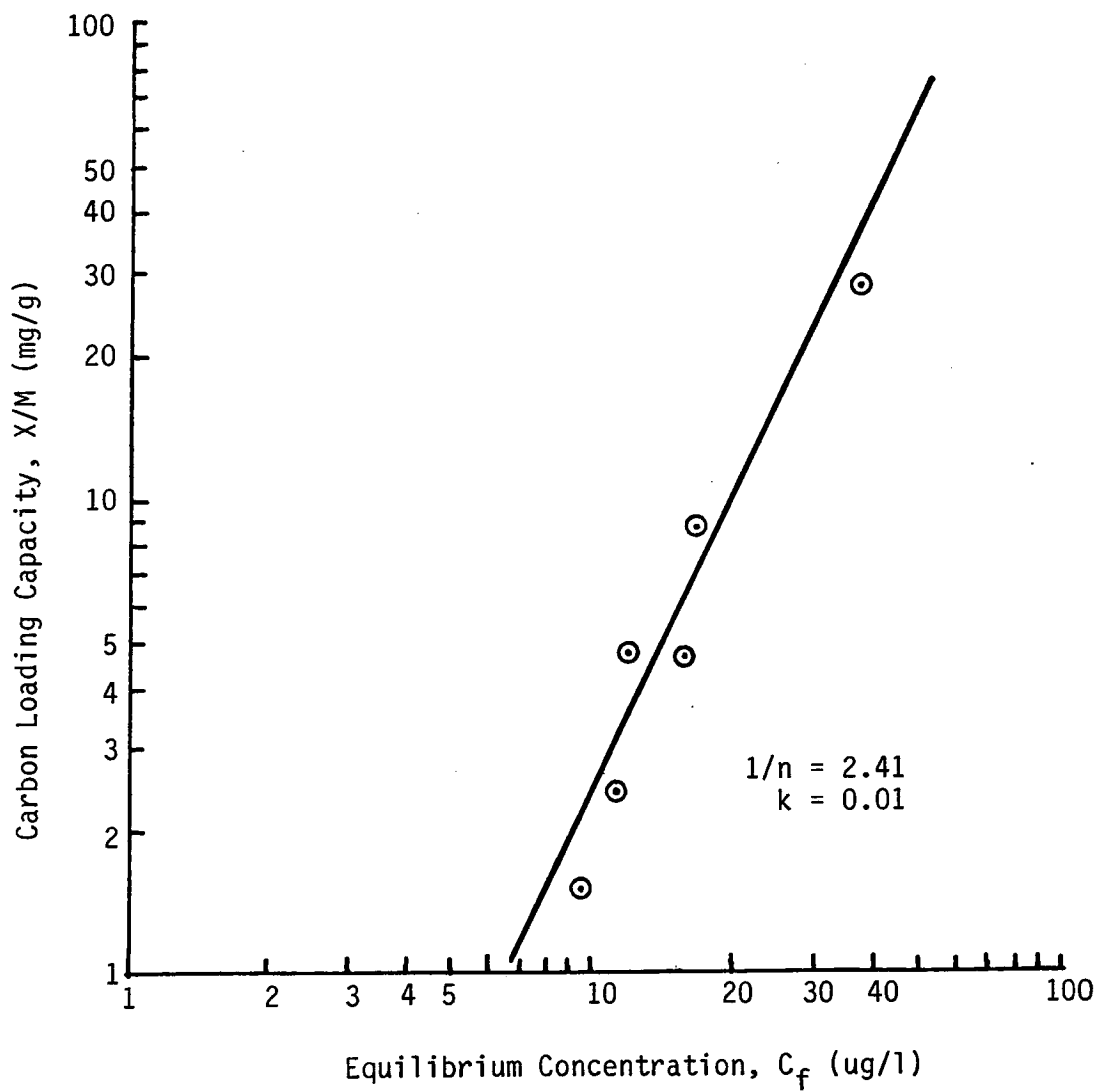


Figure 51. Freundlich Isotherm of 1,4 DCB in Presence of CT and TCE on Activated Carbon.

OPTIMIZATION AND ECONOMICS

The principal advantages of air stripping followed by GAC adsorption are found in the ability of the system to treat waters containing multiple contaminants having different volatilities and adsorption characteristics. Using air stripping as a pretreatment step will remove all or most of the highly volatile compounds as well as various portions of the lesser volatile compounds thus reducing the organic load on the activated carbon. Since the replacement or regeneration of activated carbon is typically very expensive, it is obviously advantageous to minimize the carbon usage rate. However, the capital and operating costs for the air stripper must also be considered.

To determine the optimum design of an air stripper and adsorption system there will be many variables to consider. It is likely that some of the variables may be evaluated on a qualitative or intuitive basis whereas others will require a more rigorous quantitative approach. For example, the variables to consider for the air stripper include;

- ° packing media type, size
- ° media depth
- ° removal efficiency
- ° water loading rate
- ° air to water ratio
- ° temperature

- tower diameter
- pumping costs
- blower costs

and for the adsorption system;

- carbon type
- contactor type
- removal efficiency
- pumping costs
- carbon costs

A qualitative approach may be satisfactory for selecting the packing media and the carbon type. There is generally sufficient background information available from the manufacturer or prior research or designs that a reasonably good choice can be made without extensive lab or pilot tests for comparison. However, the price or availability of certain products may result in compromises.

The next step is to determine the best combination of values for the remaining variables based on the results of actual lab or pilot tests using either the actual contaminated groundwater or synthesized substitutes. It is also important to identify the range of variability for certain parameters such as;

- influent contaminant concentrations
- influent contaminant types
- temperature
- changes in MCLs
- treated water flow rate requirements.

The treatment system should be designed to allow for fluctuations in the above parameters during the effective design life. The means of meeting the worst case treatment objectives may be accomplished by either over-designing or by providing for easy expansion, augmentation or operation modifications of the base design.

Many researchers have presented information on methods and considerations for optimizing an air stripping design in general terms [6, 9, 10, 14, 27, 49, 50]; however, Nirmalakhandan et al [9] has recently developed a computer simulation which evaluates the effects of changing any given design variable. The benefits include being able to determine the optimum design values and also identifying which variables have the most or least effect on overall treatment costs. It is likely that his results and conclusions are valid for most air stripping designs. Some of his conclusions are;

(1) The optimum water loading rate and air to water ratio can be found by plotting overall cost contours for those parameters.

(2) The overall treatment cost is relatively insensitive to modest variations of the operating variables in the vicinity of the optimum region, but increases rapidly as conditions deviate further.

(3) Capital costs may influence overall treatment costs more than power costs.

(4) Overall treatment cost is directly dependent on temperature.

(5) Selection of packing media should be based on pressure loss characteristics and $K_L a$ rather than material cost.

(6) Varying the air loading rate is generally the most efficient way to adjust operating conditions to react to changes in treatment objectives or temperature.

This study examined the following treatment system variables with the objective of removing three organic contaminants (CT, TCE and 1,4 DCB).

- Two different packing media types and sizes were compared.
- Removal vs. media depth was examined.
- Removal vs. air to water ratio was examined.
- Both packings were analyzed for pressure drop vs. liquid loading and air loading rates.
- The effect of influent concentration was examined.
- The effects of competitive adsorption was accounted for.

The three VOCs, carbon tetrachloride, trichloroethylene and 1,4 dichlorobenzene represent a relatively broad range in volatility and adsorptivity. Carbon tetrachloride is the most volatile but the least adsorbable whereas 1,4 dichlorobenzene is the least volatile and most adsorbable. Trichloroethylene is in-between and represents a good reference compound.

The results of the laboratory studies indicate that the following choices should be made regarding the design variables.

(1) The tripacks media will allow for much higher liquid loading rates and lower pressure losses than the pall rings. Although the $K_L a$ of the tripacks is lower than the pall rings, the tripacks can still achieve satisfactory removals with only slightly greater media depths.

(2) The air to water ratio should be fairly low, 39:1 or less for tripacks media. The higher A:W ratios did not improve removals significantly.

(3) Filtrasorb 400 is effective in adsorbing the combined compounds to desired effluent concentrations.

(4) Since 1,4 DCB will achieve the least removal by air stripping, it will probably be the controlling compound for the carbon polishing step.

For this study, optimization of the design variables will be accomplished as follows;

(1) Selection of packing media for air stripping: Use tripacks because of lower pressure loss and greater liquid loading rate.

(2) Determine air to water ratio: The results of experiments 10 through 14 indicate that an A:W ratio of 39:1 will accomplish most of the removal of CT and TCE and a significant portion of the 1,4 DCB. Higher A:W ratios will not accomplish much greater removals.

(3) Determine a liquid loading rate: The tripacks media offer a wide range of possible water loading rates. Figure 8 indicates that at a low air loading rate, the liquid loading rate may vary considerably with very little change in column pressure loss. For the purposes of this study the liquid loading rate was set at 13.9 gpm/ft^2 and will be used for further design calculations so that the removal data and $K_L a$ values will be valid.

(4) Optimize the final two design parameters which are packing media depth and carbon usage. For the purposes of this study, the

treatment objective is to achieve an effluent concentration of 1 ug/l or less. The influent concentrations are assumed to be 40 ug/l for carbon tetrachloride and trichloroethylene and 70 ug/l for 1,4 dichlorobenzene (150 ug/l total VOCs).

A simple mathematical approach will be applied to determine the approximate overall cost minimum for the variables versus percent removal. In order to present that methodology it is necessary to establish several unit cost factors and make certain assumptions with regard to system conditions and configurations.

Assumed unit costs are estimated as follows; carbon cost = \$1.25 per pound. Air stripping tower and packing varies from \$9,000 per vertical foot at 3-foot height to \$3,333 per vertical foot at 30-foot height. Column shell is assumed to be carbon steel and 2-inch plastic media is used. Pump costs include the pump, motor, valves, piping and electrical equipment needed to lift the process water flow from ground level to the top of the tower. The cost varies slightly according to packing height, \$20,000 for 3-foot height and \$25,000 for 30-foot height. Reference [2] is used as a basis of these costs. Pump operating cost also varies as a function of the packing height. Using an electrical power cost of \$0.10 per Kwhr, the pump operating costs will range from \$2.00 per million gallons (MG) at 3-foot height to \$12.00 per MG at 30-foot height.

The capital costs associated with the tower, packing and pump equipment have been amortized over 20 years at an annual interest rate of 10% in order to assess the cost per MG value. Other equipment and

operating costs such as the air blower and carbon contactor have been considered to be relatively constant with respect to changes in the packing height.

A water flow rate of 700 gpm (1 MGD) is assumed for calculation purposes. It will operate constantly and the influent contaminant concentrations will not vary.

Figure 52 shows the packing media depths necessary to achieve up to 99% removal by air stripping. The transfer unit model was used, with NTU, R and $K_L a$ values as determined in laboratory experiments for this study. Figure 53 shows estimated carbon usage that would occur for various levels of air stripping pretreatment. The results of the carbon adsorption isotherm experiments were used to estimate the carbon usage requirements. The adsorption capacity, X/M value varied in relation to the level of air stripping pretreatment. In other words, after obtaining a given level of air stripping pretreatment (% removal), that effluent VOC concentration then becomes the activated carbon influent and C_f concentration for which a corresponding adsorption capacity, X/M value exists. The carbon usage, in terms of lb/MG, were then calculated from the C_f and X/M values. For CT and TCE this relationship resulted in a gradual decline in carbon usage (lb/MG) as the level of air stripping pretreatment increased (% removal). However, for 1,4 DCB, the $1/n$ slope of the adsorption isotherm is so steep that the carbon usage actually increases as the level of air stripping pretreatment increases.

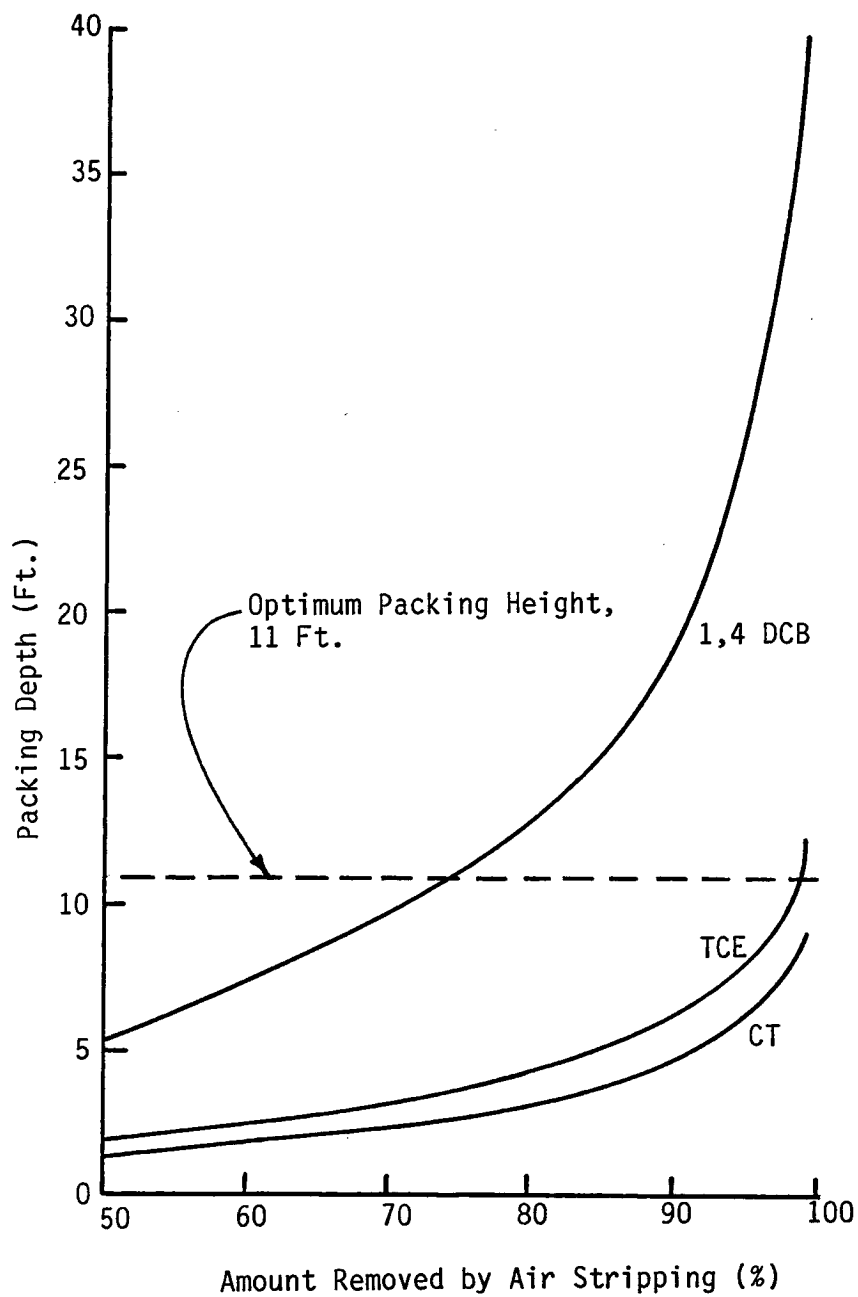


Figure 52. Packing Depth vs. Percent Removal by Air Stripping Pretreatment.

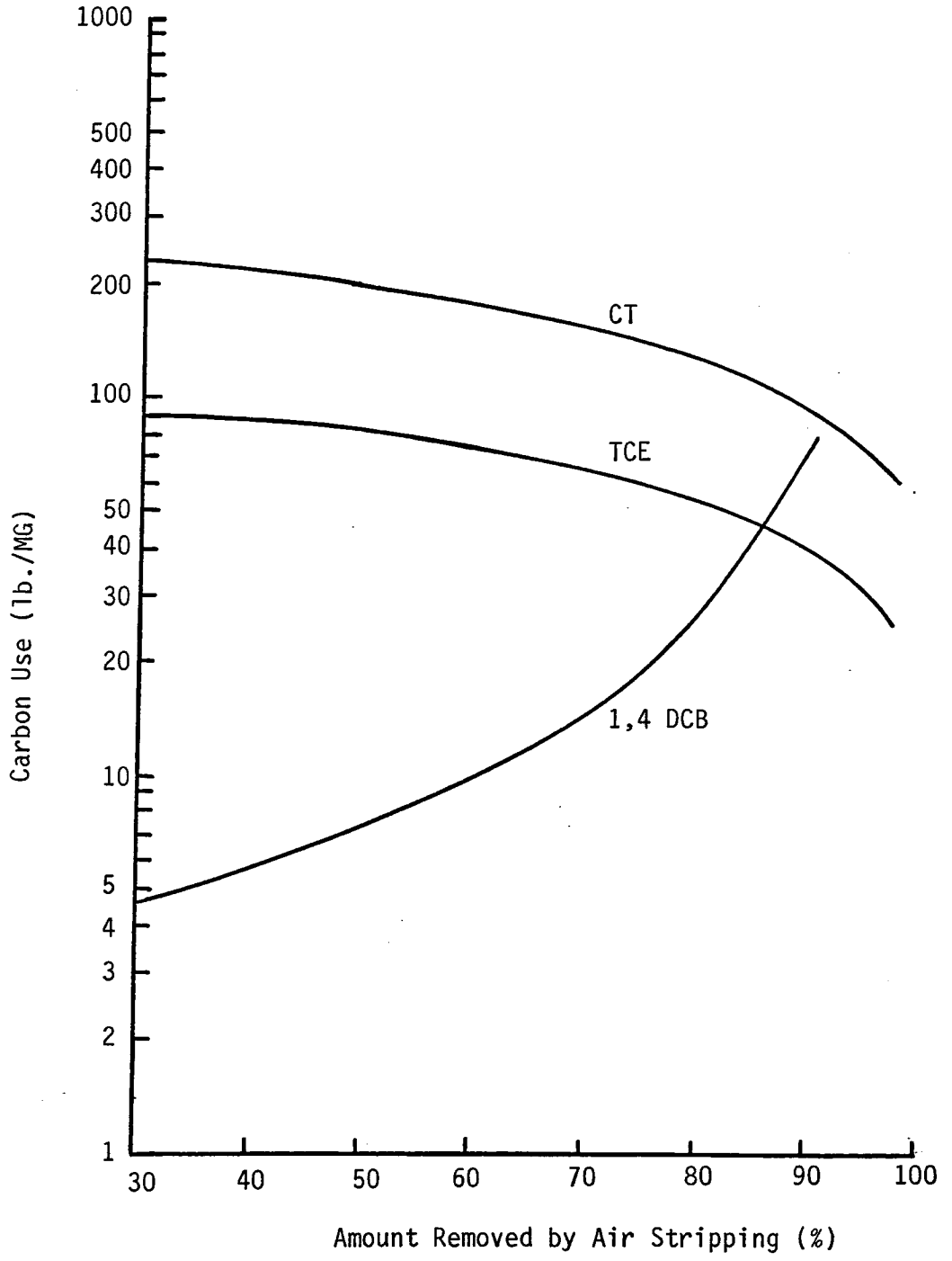


Figure 53. Activated Carbon Use vs. Percent Removal by Air Stripping Pretreatment.

The optimum design combination of air stripping and activated carbon adsorption is determined by finding the cost minimum in terms of dollars per million gallons treated for column height versus carbon usage. Figure 54 shows the combined costs and the optimum design point. For this particular combination of compounds and treatment objectives it appears that 1,4 DCB is the controlling compound. The cost minimum is reached at a packing height of 11 feet, a carbon usage rate of 20 lb/MG and an overall treatment cost of \$50/MG. The air stripping pretreatment will remove greater than 99% of the CT and greater than 98% of the TCE contaminants but only about 78% of the 1,4 DCB. Thus the activated carbon is necessary to accomplish additional 1,4 DCB removal to at least 98.6%

For comparison purposes, if all three compounds were to be removed by air stripping alone, it would be necessary to have a packing height of about 40 feet (1,4 DCB controls). The approximate treatment cost would be \$72/MG. However, the capital costs associated with the carbon contacting system would have to be less than \$70,000 in order for air stripping alone to be economical. Conversely, if all three compounds were to be removed by activated carbon adsorption alone (CT controls), the carbon usage rates would be about 280 lb/MG or \$350/MG.

The advantages of using air stripping as a pretreatment step to activated carbon adsorption is quite apparent in this example design. Although most of the contaminant removal will occur in the air stripper, the carbon does allow for a more economical operating cost. Furthermore the activated carbon step may prove to be indispensable if

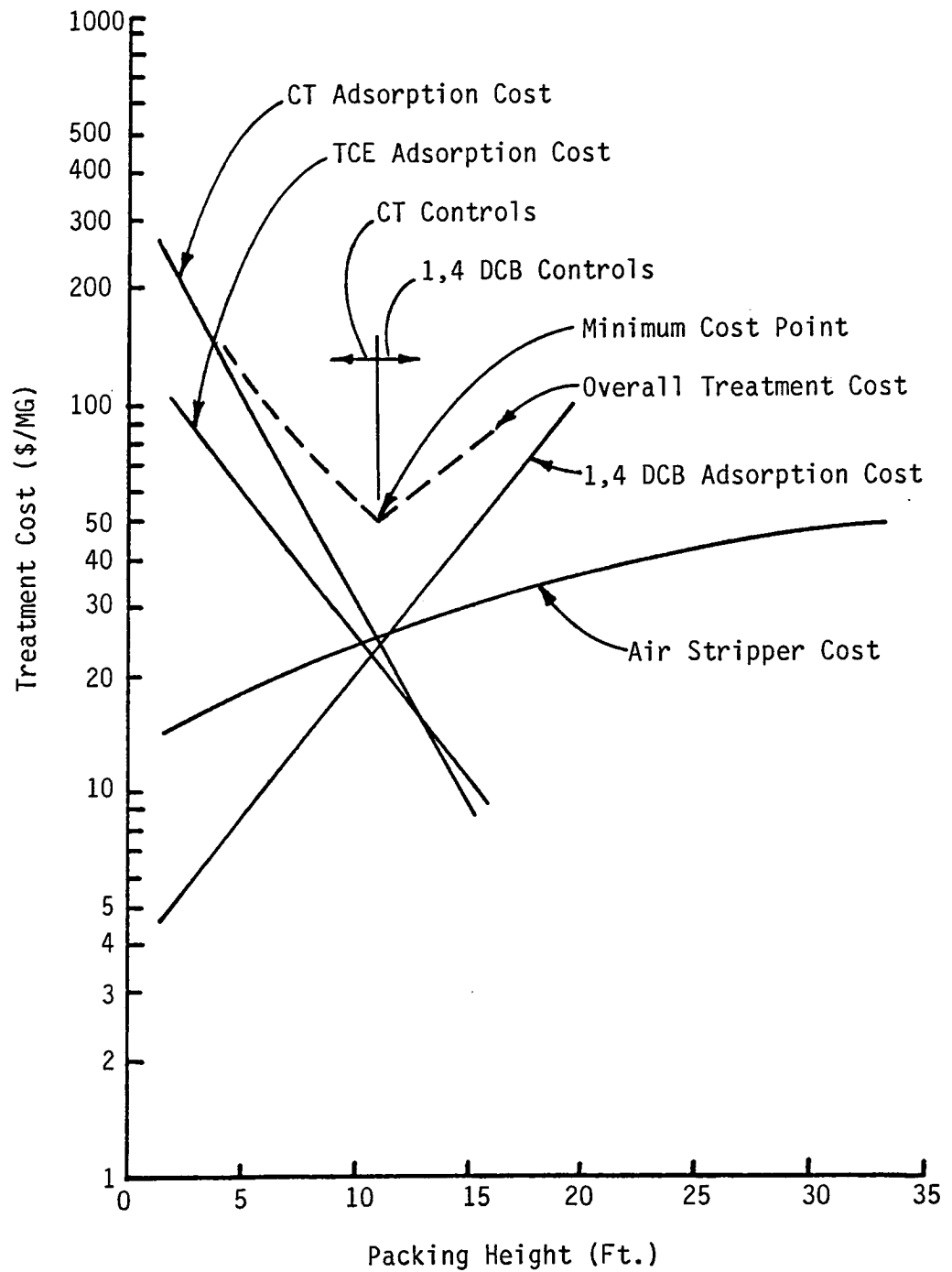


Figure 54. Treatment Costs vs. Packing Height, Optimum Design Point.

highly non-volatile compounds are present. Additionally, any fluctuations in contaminant concentrations during the design life of the facility will most likely be handled by adjusting the A:W ratio of the air stripper or by increased carbon usage. These combined processes offer a wide range of operational flexibility not available with any single unit process.

CONCLUSIONS

It is apparent that treatment of groundwater containing multiple contaminants may best be accomplished by a combination of air stripping and activated carbon adsorption. In general, if the compounds are of varying volatilities and different adsorption capacities, then the combined treatment processes will be most economical.

The results of this study reveal that there are two controlling compounds which define the optimization. At low packing heights (0 to 11 ft.), CT is controlling and at higher packing heights (11 to 15 ft.), DCB is controlling. Other types or combinations of VOCs may result in only a single compound controlling the optimization. Furthermore, in actual practice, other water constituents such as humic substances or dissolved organic carbon would influence the adsorption capacity of activated carbon for VOCs [50].

Although the method of optimization utilized in the study was a simple economics problem a more detailed and comprehensive approach could be developed by applying computer model simulation as was done by Nirmalakhandan [9]. This would be a good direction for further research.

APPENDIX

TABULATIONS AND GRAPHICAL PLOTS OF RESULTS OF THE AIR STRIPPING EXPERIMENTS

**(Includes Tables 7 through 16
and Figures 9 through 48.)**

Table 7. Sample Concentrations for VOC Experiments (ug/l)

Packing Media	Exp #	L (gpm/ft ²)	A:W (vol:vol)	Carbon Tetrachloride				Trichloroethylene				1,4 Dichlorobenzene			
				0 ft.	1.92 ft.	3.92 ft.	5.67 ft.	0 ft.	1.92 ft.	3.92 ft.	5.67 ft.	0 ft.	1.92 ft.	3.92 ft.	5.67 ft.
Pall Rings	1	7.3	0	40.4	24.5	10.3	7.3	44.7	30.7	25.0	20.0	162	110	--	86.0
Pall Rings	2	7.3	75	41.8	7.40	1.40	0.70	60.2	15.0	2.50	1.10	170	69.0	43.1	22.7
Pall Rings	3	7.3	150	40.0	--	1.20	1.20	42.8	26.0	1.70	1.80	166	52.9	27.9	--
Pall Rings	4	7.3	224	42.7	5.40	1.20	1.00	64.1	10.0	1.40	1.30	162	57.3	45.9	36.9
Pall Rings	5	7.3	314	90.0	8.50	2.30	1.40	45.4	15.9	3.90	--	148	49.8	22.8	--
Pall Rings	6	7.3	0	14.6	9.80	6.60	5.30	38.3	32.4	25.3	22.5	--	--	--	--
Pall Rings	7	7.3	150	14.2	4.75	1.85	0.95	37.7	12.6	5.25	2.60	--	--	--	--
Pall Rings	8	7.3	224	15.0	4.75	2.15	0.75	39.0	12.4	5.80	2.30	--	--	--	--
Pall Rings	9	7.3	314	14.6	3.15	1.70	0.80	38.3	8.80	4.95	2.55	--	--	--	--
Tripacks	10	13.9	0	90.7	62.0	57.9	50.6	--	--	--	--	86.1	71.6	--	--
Tripacks	11	13.9	39	100	34.3	10.5	5.00	75.3	45.9	17.8	8.20	108	50.5	40.1	33.7
Tripacks	12	13.9	79	128	37.5	11.7	7.50	76.8	46.3	19.2	11.5	60.3	57.7	34.8	29.3
Tripacks	13	13.9	118	90.9	31.0	12.3	5.00	92.4	40.7	19.6	7.5	63.5	53.4	35.1	23.1
Tripacks	14	13.9	205	86.4	22.7	10.6	5.50	82.0	31.0	16.9	8.3	69.6	50.4	33.1	18.3

Packing Media	Exp #	L (gpm/ft ²)	A:W (vol:vol)	Carbon Tetrachloride			Trichloroethylene			1,4 Dichlorobenzene		
				1.92 ft. 3.92 ft. 5.67 ft.	1.92 ft. 3.92 ft. 5.67 ft.	1.92 ft. 3.92 ft. 5.67 ft.	1.92 ft. 3.92 ft. 5.67 ft.	1.92 ft. 3.92 ft. 5.67 ft.	1.92 ft. 3.92 ft. 5.67 ft.	1.92 ft. 3.92 ft. 5.67 ft.		
Pall Rings	1	7.3	0	39.3 (20.5)	74.5 (19.0)	81.9 (14.4)	31.3 (16.3)	44.1 (11.3)	55.3 (9.8)	32.1 (16.7)	-- (--)	46.9 (8.3)
Pall Rings	2	7.3	75	82.3 (42.9)	96.7 (24.7)	98.3 (17.3)	75.1 (39.1)	95.8 (24.4)	98.2 (17.3)	59.5 (31.0)	74.8 (19.1)	86.7 (15.3)
Pall Rings	3	7.3	150	-- (--)	97.0 (24.4)	-- (--)	39.0 (20.5)	96.0 (24.5)	-- (--)	68.2 (35.5)	83.2 (21.2)	-- (--)
Pall Rings	4	7.3	224	87.4 (45.5)	97.2 (24.8)	97.7 (17.2)	84.4 (44.0)	97.8 (24.9)	97.9 (17.3)	64.6 (33.6)	71.6 (18.3)	77.2 (13.6)
Pall Rings	5	7.3	314	90.6 (47.2)	97.4 (24.8)	98.4 (17.4)	64.9 (33.9)	91.4 (23.3)	-- (--)	66.3 (34.5)	84.6 (21.6)	-- (--)
Pall Rings	6	7.3	0	32.8 (17.1)	54.7 (14.0)	64.0 (11.3)	15.5 (8.1)	34.1 (8.7)	41.3 (7.3)	-- (--)	-- (--)	-- (--)
Pall Rings	7	7.3	150	66.4 (34.6)	86.9 (22.2)	93.3 (16.5)	66.7 (34.7)	86.1 (22.0)	93.1 (16.4)	-- (--)	-- (--)	-- (--)
Pall Rings	8	7.3	224	68.3 (35.6)	85.7 (21.9)	95.0 (16.8)	68.3 (35.6)	85.1 (21.7)	94.1 (16.6)	-- (--)	-- (--)	-- (--)
Pall Rings	9	7.3	314	78.4 (40.8)	88.3 (22.5)	94.5 (16.7)	77.0 (40.1)	87.1 (22.2)	93.3 (16.5)	-- (--)	-- (--)	-- (--)
Tripacks	10	13.9	0	31.6 (16.5)	36.2 (9.2)	44.2 (7.8)	-- (--)	-- (--)	-- (--)	16.8 (8.8)	-- (--)	-- (--)
Tripacks	11	13.9	39	65.8 (34.3)	89.5 (22.8)	95.0 (16.8)	39.0 (20.3)	76.4 (19.5)	89.1 (15.7)	53.4 (27.8)	63.0 (16.1)	68.9 (12.2)
Tripacks	12	13.9	79	70.7 (36.8)	90.9 (23.2)	94.1 (16.6)	39.7 (20.7)	75.0 (19.1)	85.0 (15.0)	4.3 (2.2)	42.3 (10.8)	51.4 (9.1)
Tripacks	13	13.9	118	65.9 (34.3)	86.5 (22.1)	94.5 (16.7)	55.9 (29.1)	78.8 (20.1)	91.9 (16.2)	15.9 (8.3)	44.7 (11.4)	63.6 (11.2)
Tripacks	14	13.9	205	73.7 (38.4)	87.7 (22.4)	93.6 (16.5)	62.2 (32.4)	79.4 (20.3)	89.9 (15.9)	27.6 (14.4)	52.4 (13.4)	73.7 (13.0)

Packing Media	Exp #	L (gpm/ft ²)	A:W (vol:vol)	Carbon Tetrachloride			Trichloroethylene			1,4 Dichlorobenzene		
				0-1.92 ft.	1.92-3.92 ft.	3.92-5.67 ft.	0-1.92 ft.	1.92-3.92 ft.	3.92-5.67 ft.	0-1.92 ft.	1.92-3.92 ft.	3.92-5.67 ft.
Pall Rings	1	7.3	0	39.3 (20.5)	58.0 (29.0)	29.1 (16.6)	31.3 (16.3)	18.6 (9.3)	20.0 (11.4)	32.1 (16.7)	-- (--)	22.0 (5.9)
Pall Rings	2	7.3	75	82.3 (42.9)	81.1 (40.6)	50 (28.6)	75.1 (39.1)	83.3 (41.7)	56.0 (32.0)	59.5 (31.0)	60.1 (30.1)	47.3 (27.0)
Pall Rings	3	7.3	150	-- (--)	97 (24.4)	0 (0)	39.3 (20.5)	93.5 (46.8)	-- (--)	68.2 (35.5)	47.3 (23.7)	-- (--)
Pall Rings	4	7.3	224	87.4 (45.5)	77.8 (38.9)	16.7 (9.5)	84.4 (44.0)	86.0 (43.0)	7.1 (4.1)	64.6 (33.6)	20.0 (10.0)	19.6 (11.2)
Pall Rings	5	7.3	314	90.6 (47.2)	72.9 (36.5)	39.1 (22.3)	65.0 (33.9)	75.5 (37.8)	-- (--)	66.3 (34.5)	54.2 (27.1)	-- (--)
Pall Rings	6	7.3	0	32.8 (17.1)	32.7 (16.4)	20.5 (11.7)	15.5 (8.1)	21.9 (11.0)	10.9 (6.2)	-- (--)	-- (--)	-- (--)
Pall Rings	7	7.3	150	66.4 (34.6)	61.0 (30.5)	48.6 (27.8)	66.7 (34.7)	58.2 (29.1)	50.5 (28.9)	-- (--)	-- (--)	-- (--)
Pall Rings	8	7.3	224	68.3 (35.6)	54.7 (27.4)	65.1 (37.2)	68.3 (35.6)	53.0 (26.5)	60.3 (34.5)	-- (--)	-- (--)	-- (--)
Pall Rings	9	7.3	314	78.4 (40.8)	46.0 (23.0)	52.9 (30.2)	77.0 (40.1)	43.8 (21.9)	48.5 (27.7)	-- (--)	-- (--)	-- (--)
Tripacks	10	13.9	0	31.6 (16.5)	6.6 (3.3)	12.6 (7.2)	-- (--)	-- (--)	-- (--)	16.8 (8.8)	-- (--)	-- (--)
Tripacks	11	13.9	39	65.8 (34.3)	69.4 (34.7)	52.4 (29.9)	39.0 (20.3)	61.2 (30.6)	53.9 (30.8)	53.4 (27.8)	20.6 (10.3)	16.0 (9.1)
Tripacks	12	13.9	79	70.7 (36.8)	68.8 (34.4)	35.9 (20.5)	39.7 (20.7)	58.5 (29.3)	40.1 (22.9)	4.3 (2.2)	7.7 (3.9)	15.8 (9.0)
Tripacks	13	13.9	118	65.9 (34.3)	60.3 (30.2)	59.3 (33.9)	55.9 (29.1)	51.8 (25.9)	61.7 (35.3)	15.9 (8.3)	34.3 (17.2)	34.2 (19.5)
Tripacks	14	13.9	205	73.7 (38.4)	53.3 (26.7)	48.1 (27.5)	62.2 (32.4)	45.5 (22.8)	50.9 (29.1)	27.6 (14.4)	34.3 (17.2)	44.7 (25.5)

Table 10. (C_{in}/C_{out}) Relative to Top of Packing

Packing Media	Exp #	G/L (vol/vol)	L ² /m ² s	G/m ² s	Carbon Tetrachloride		Trichloroethylene		1,4 Dichlorobenzene				
					.585 m.	1.20 m.	1.73 m.	.585 m.	1.20 m.	1.73 m.	.585 m.	1.20 m.	1.73 m.
Pall Rings	1	0	4.98	0	1.65	3.92	5.53	1.46	1.79	2.24	1.47	--	1.89
Pall Rings	2	75	4.98	373	5.65	29.9	59.7	4.0	24.1	54.7	2.47	3.95	7.50
Pall Rings	3	150	4.98	745	--	33.33	33.33	1.65	25.2	--	3.15	5.96	--
Pall Rings	4	224	4.98	1,118	7.91	35.6	42.7	6.41	45.8	49.3	2.82	3.52	4.38
Pall Rings	5	314	4.98	1,565	10.6	39.1	64.3	2.86	11.6	--	2.95	6.49	--
Pall Rings	6	0	4.98	0	1.49	2.21	2.78	1.18	1.52	1.70	--	--	--
Pall Rings	7	150	4.98	745	2.98	7.65	14.9	3.00	7.17	14.5	--	--	--
Pall Rings	8	224	4.98	1,118	3.16	6.98	20.0	3.16	6.72	16.9	--	--	--
Pall Rings	9	314	4.98	1,565	4.63	8.59	18.3	4.35	7.74	15.0	--	--	--
Tripacks	10	0	9.46	0	1.46	1.57	1.79	--	--	--	1.20	--	--
Tripacks	11	39	9.46	373	2.92	9.54	20.0	1.64	4.23	9.18	2.15	2.70	3.22
Tripacks	12	79	9.46	745	3.42	10.9	17.1	1.66	4.00	6.68	1.05	1.73	2.06
Tripacks	13	118	9.46	1,118	2.93	7.39	18.2	2.27	4.71	12.3	1.19	1.81	2.75
Tripacks	14	205	9.46	1,937	3.81	8.15	15.7	2.65	4.85	9.88	1.38	2.10	3.80

Table 11. Incremental (C_{in}/C_{out}) Between Sample Ports

Packing Media	Exp #	G/L (vol/vol)	L $1/m^2$	G $1/m^2$	Carbon Tetrachloride		Trichloroethylene		1,4 Dichlorobenzene				
					0 - .585 m. 1.20 m. 1.73 m.	.585 - 1.20 m. 1.73 m.	0 - .585 m. 1.20 m. 1.73 m.	.585 m. 1.20 m. 1.73 m.	0 - .585 m. 1.20 m. 1.73 m.	.585 m. 1.20 m. 1.73 m.			
Pall Rings	1	0	4.98	0	1.65	2.38	1.41	1.46	1.23	1.25	1.47	--	--
Pall Rings	2	75	4.98	373	5.65	5.29	2.0	4.0	6.0	2.27	2.47	1.60	1.90
Pall Rings	3	150	4.98	745	--	--	--	1.65	15.30	--	3.15	1.90	--
Pall Rings	4	224	4.98	1,118	7.91	4.5	1.2	6.41	7.14	1.08	2.82	1.25	1.24
Pall Rings	5	314	4.98	1,565	10.59	3.70	1.64	2.86	4.08	--	2.95	2.18	--
Pall Rings	6	0	4.98	0	1.49	1.49	1.26	1.18	1.28	1.12	--	--	--
Pall Rings	7	150	4.98	745	2.98	2.57	1.95	3.00	2.39	2.02	--	--	--
Pall Rings	8	224	4.98	1,118	3.16	2.21	2.87	3.16	2.13	2.52	--	--	--
Pall Rings	9	314	4.98	1,565	4.63	1.85	2.13	4.35	1.78	1.94	--	--	--
Tripacks	10	0	9.46	0	1.46	1.07	1.14	--	--	--	1.20	--	--
Tripacks	11	39	9.46	373	2.92	3.27	2.10	1.64	2.58	2.17	2.15	1.26	1.19
Tripacks	12	79	9.46	745	3.42	3.21	1.56	1.66	2.41	1.67	1.05	1.66	1.19
Tripacks	13	118	9.46	1,118	2.93	2.52	2.46	2.27	2.08	2.61	1.19	1.52	1.52
Tripacks	14	205	9.46	1,937	3.81	2.14	1.93	2.65	1.83	2.04	1.38	1.52	1.81

Table 12. Stripping Factors, R (at 25°C)

Packing Media	Exp #	G/L (vol/vol)	L 1/m ² s	G 1/m ² s	Carbon Tetrachloride (H = 1678 atm)		Trichloroethylene (H = 684 atm)		1,4 Dichlorobenzene (H = 238 atm)	
Pall Rings	1	0	4.98	0	1.27	0.516	0.180			
Pall Rings	2	75	4.98	373	95	39	13.5			
Pall Rings	3	150	4.98	745	190	77	27			
Pall Rings	4	224	4.98	1,118	284	115	40			
Pall Rings	5	314	4.98	1,565	398	162	56			
Pall Rings	6	0	4.98	0	1.27	0.516	--			
Pall Rings	7	150	4.98	745	190	77	--			
Pall Rings	8	224	4.98	1,118	284	115	--			
Pall Rings	9	314	4.98	1,565	398	162	--			
Tripacks	10	0	9.46	0	1.27	0.516	0.180			
Tripacks	11	39	9.46	373	49.4	20	7.0			
Tripacks	12	79	9.46	745	100	41	14.2			
Tripacks	13	118	9.46	1,118	149	61	21.2			
Tripacks	14	205	9.46	1,937	260	106	37			

Table 13. Number of Transfer Units, NTU, Relative to Top of Packing

Packing Media	Exp #	Carbon Tetrachloride		Trichloroethylene		1,4 Dichlorobenzene				
		.585 m.	1.73 m.	.585 m.	1.73 m.	.585 m.	1.73 m.			
Pall Rings	1	.608	2.27	3.17	.602	1.44	--	--	--	--
Pall Rings	2	1.74	3.42	4.12	1.40	3.24	4.08	.928	1.422	2.104
Pall Rings	3	--	3.52	3.52	0.502	3.26	--	1.165	1.821	--
Pall Rings	4	2.07	3.58	3.76	1.87	3.85	3.92	1.047	1.272	1.495
Pall Rings	5	2.36	3.67	4.17	1.053	2.460	--	1.089	1.889	--
Pall Rings	6	0.466	1.077	1.510	0.197	0.713	1.139	--	--	--
Pall Rings	7	1.094	2.041	2.710	1.104	1.984	2.697	--	--	--
Pall Rings	8	1.152	1.947	3.00	1.155	1.914	2.843	--	--	--
Pall Rings	9	1.534	2.154	2.912	1.475	2.054	2.719	--	--	--
Tripacks	10	0.439	0.538	0.730	--	--	--	0.531	--	--
Tripacks	11	1.080	2.28	3.037	0.500	1.477	2.286	0.800	1.049	1.243
Tripacks	12	1.235	2.404	2.858	0.509	1.402	1.925	0.049	0.557	0.738
Tripacks	13	1.078	2.008	2.915	0.824	1.562	2.536	0.175	0.600	1.030
Tripacks	14	1.340	2.103	2.761	0.978	1.586	2.304	0.323	0.748	1.351

Table 14. Number of Transfer Units, NTU, Incremental Between Sample Ports

Packing Media	Exp #	Carbon Tetrachloride		Trichloroethylene		1,4 Dichlorobenzene				
		0-.585 m.	1.2-1.73 m.	0-.585 m.	1.2-1.73 m.	0-.585 m.	1.2-1.73 m.			
Pall Rings	1	.608	1.210	0.393	.602	0.259	0.285	--	--	
Pall Rings	2	1.74	1.675	0.695	1.40	1.817	0.826	.928	0.477	0.654
Pall Rings	3	--	--	--	0.502	2.751	--	1.165	0.648	--
Pall Rings	4	2.07	1.507	0.182	1.87	1.975	0.077	1.047	0.223	0.216
Pall Rings	5	2.36	1.310	0.495	1.053	1.410	--	1.089	0.784	--
Pall Rings	6	0.466	0.466	0.253	0.197	0.325	0.127	--	--	--
Pall Rings	7	1.094	0.946	0.669	1.104	0.875	0.706	--	--	--
Pall Rings	8	1.152	0.794	1.056	1.155	0.758	0.927	--	--	--
Pall Rings	9	1.534	0.616	0.757	1.475	0.577	0.663	--	--	--
Tripacks	10	0.439	0.069	0.138	--	--	--	0.531	--	--
Tripacks	11	1.080	1.195	0.746	0.500	0.965	0.787	0.800	0.235	0.176
Tripacks	12	1.235	1.171	0.446	0.509	0.887	0.516	0.049	0.515	0.175
Tripacks	13	1.078	0.926	0.902	0.824	0.736	0.965	0.175	0.422	0.422
Tripacks	14	1.340	0.762	0.658	0.978	0.606	0.715	0.323	0.422	0.600

Table 15. Liquid Mass Transfer Coefficient, $K_L a$ (hr^{-1}),
Relative to Top of Packing

Packing Media	Exp #	L' $\frac{\text{kg mole}}{\text{m}^2 \text{ hr}}$	Carbon Tetrachloride		Trichloroethylene		1,4 Dichlorobenzene				
			.585 m.	1.2 m.	1.73 m.	.585 m.	1.2 m.	1.73 m.	.585 m.	1.2 m.	1.73 m.
Pall Rings 1	997		18.6	33.92	31.93	18.45	21.52	--	--	--	--
Pall Rings 2	997		53.3	51.11	41.50	42.91	48.42	41.10	28.45	21.25	21.20
Pall Rings 3	997		--	52.60	35.46	15.39	48.71	--	35.71	27.21	--
Pall Rings 4	997		63.5	53.50	37.88	57.32	57.53	39.49	32.09	19.01	15.06
Pall Rings 5	997		72.3	54.84	42.01	32.28	36.76	--	33.38	28.23	--
Pall Rings 6	997		14.28	16.09	15.21	6.039	10.65	11.47	--	--	--
Pall Rings 7	997		33.53	30.50	27.30	33.84	29.65	27.17	--	--	--
Pall Rings 8	997		35.31	29.09	30.22	35.40	28.60	28.64	--	--	--
Pall Rings 9	997		47.02	32.19	29.34	45.21	30.69	27.39	--	--	--
Tripacks 10	1,894		25.56	15.27	13.97	--	--	--	30.92	--	--
Tripacks 11	1,894		62.89	64.72	58.12	29.12	41.93	43.75	46.58	29.78	23.79
Tripacks 12	1,894		71.91	68.24	54.69	29.64	39.80	36.84	2.85	15.81	14.12
Tripacks 13	1,894		62.77	57.00	55.79	47.98	44.34	48.53	10.19	17.03	19.71
Tripacks 14	1,894		78.03	59.70	52.84	56.95	45.02	44.09	18.81	21.23	25.85

Table 16. Liquid Mass Transfer Coefficient, K_{La} (hr^{-1}),
Incremental Between Sample Ports

Packing Media	Exp #	L' $\frac{\text{kg mole}}{\text{m}^2 \text{ hr}}$	Carbon Tetrachloride		Trichloroethylene		1,4 Dichlorobenzene				
			0-.585 m. .585-1.2 m. 1.2-1.73 m.	1.2-1.73 m.	0-.585 m. .585-1.2 m. 1.2-1.73 m.	1.2-1.73 m.	0-.585 m. .585-1.2 m. 1.2-1.73 m.	1.2-1.73 m.			
Pall Rings	1	997	18.6	35.3	12.2	18.5	7.60	8.80	--	--	--
Pall Rings	2	997	53.3	48.8	21.5	42.9	53.0	25.5	28.5	13.9	20.2
Pall Rings	3	997	--	--	--	15.4	80.2	--	35.7	18.9	--
Pall Rings	4	997	63.5	43.9	5.60	57.3	57.6	2.40	32.1	6.50	6.70
Pall Rings	5	997	72.3	38.2	15.3	32.3	41.1	--	33.4	22.9	--
Pall Rings	6	997	14.3	13.6	7.80	6.04	9.5	3.90	--	--	--
Pall Rings	7	997	33.5	27.6	20.7	33.8	25.5	21.8	--	--	--
Pall Rings	8	997	35.3	23.2	32.6	35.4	22.1	28.7	--	--	--
Pall Rings	9	997	47.0	18.0	23.4	45.2	16.8	20.5	--	--	--
Tripacks	10	1,894	25.6	3.80	8.10	--	--	--	30.9	--	--
Tripacks	11	1,894	62.9	66.2	43.8	29.1	53.5	46.2	46.6	13.0	10.3
Tripacks	12	1,894	71.9	64.9	26.2	29.6	49.1	30.3	2.85	28.5	10.3
Tripacks	13	1,894	62.8	51.3	53.0	48.0	40.8	56.7	10.2	23.4	24.8
Tripacks	14	1,894	78.0	42.2	38.6	57.0	33.6	42.0	18.8	23.4	35.2

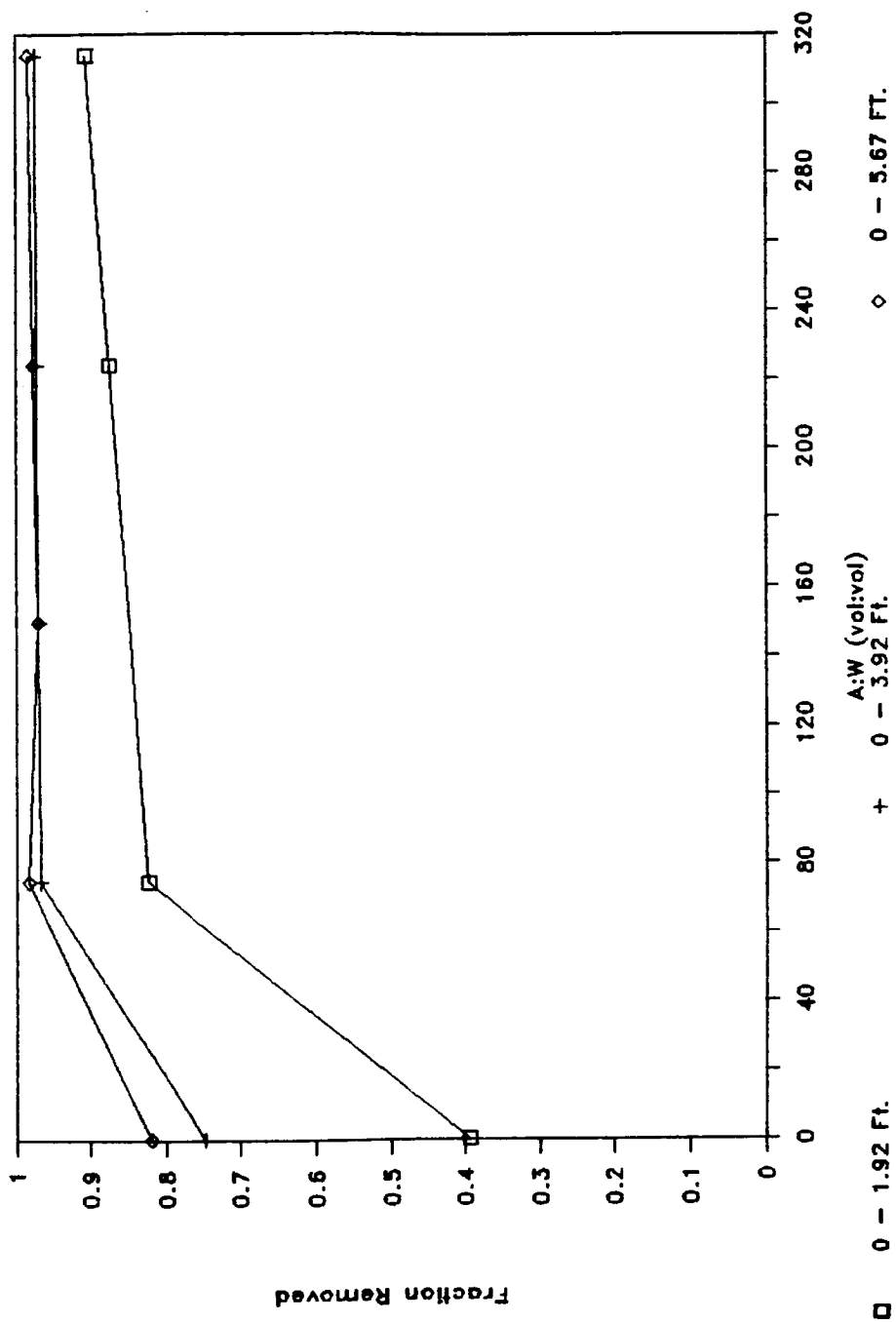


Figure 9. Experiments 1-5, Pall Rings, CT Removal Relative to Top of Packing vs. A:W.

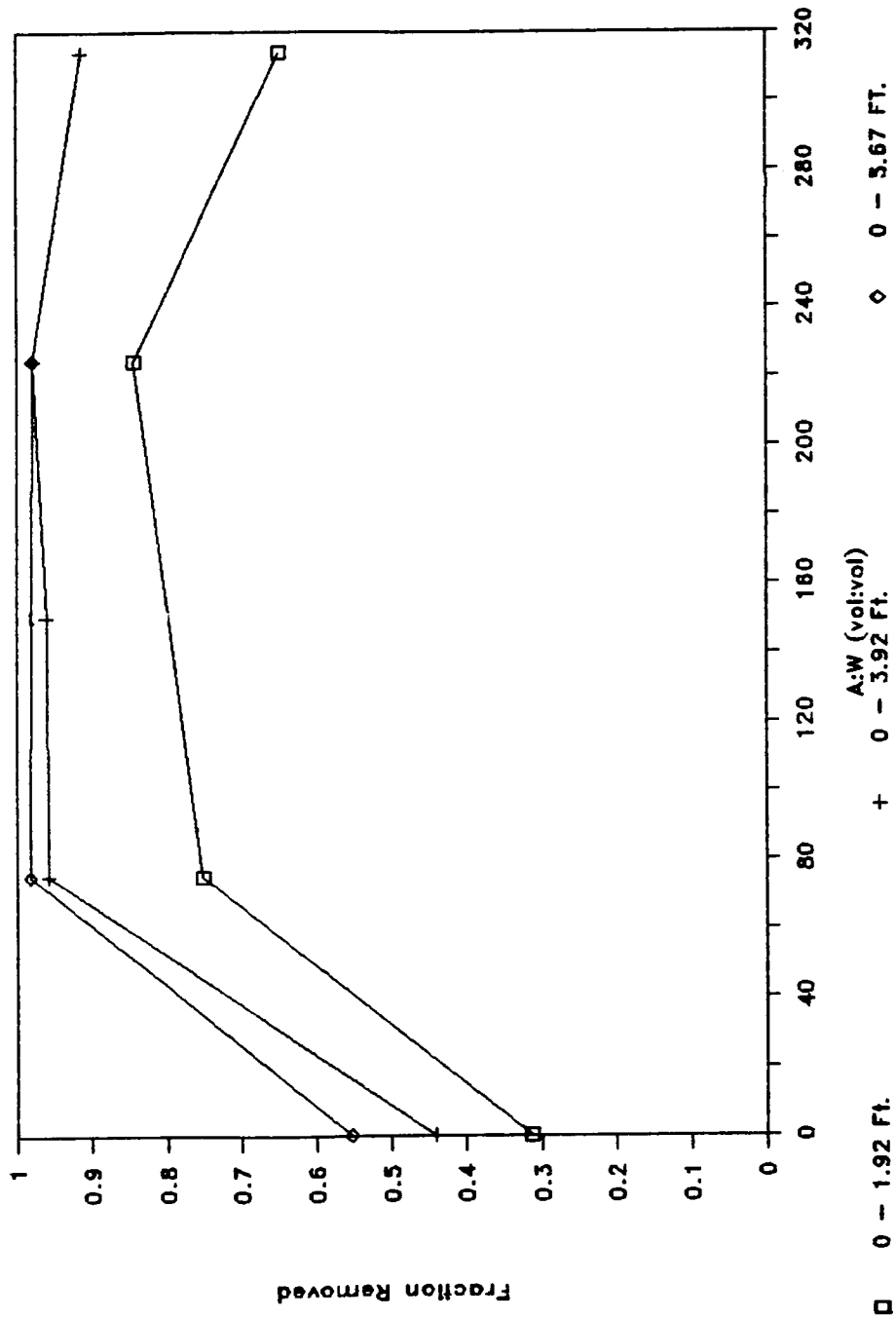


Figure 10. Experiments 1-5, Pall Rings, TCE Removal Relative to Top of Packing vs. A:W.

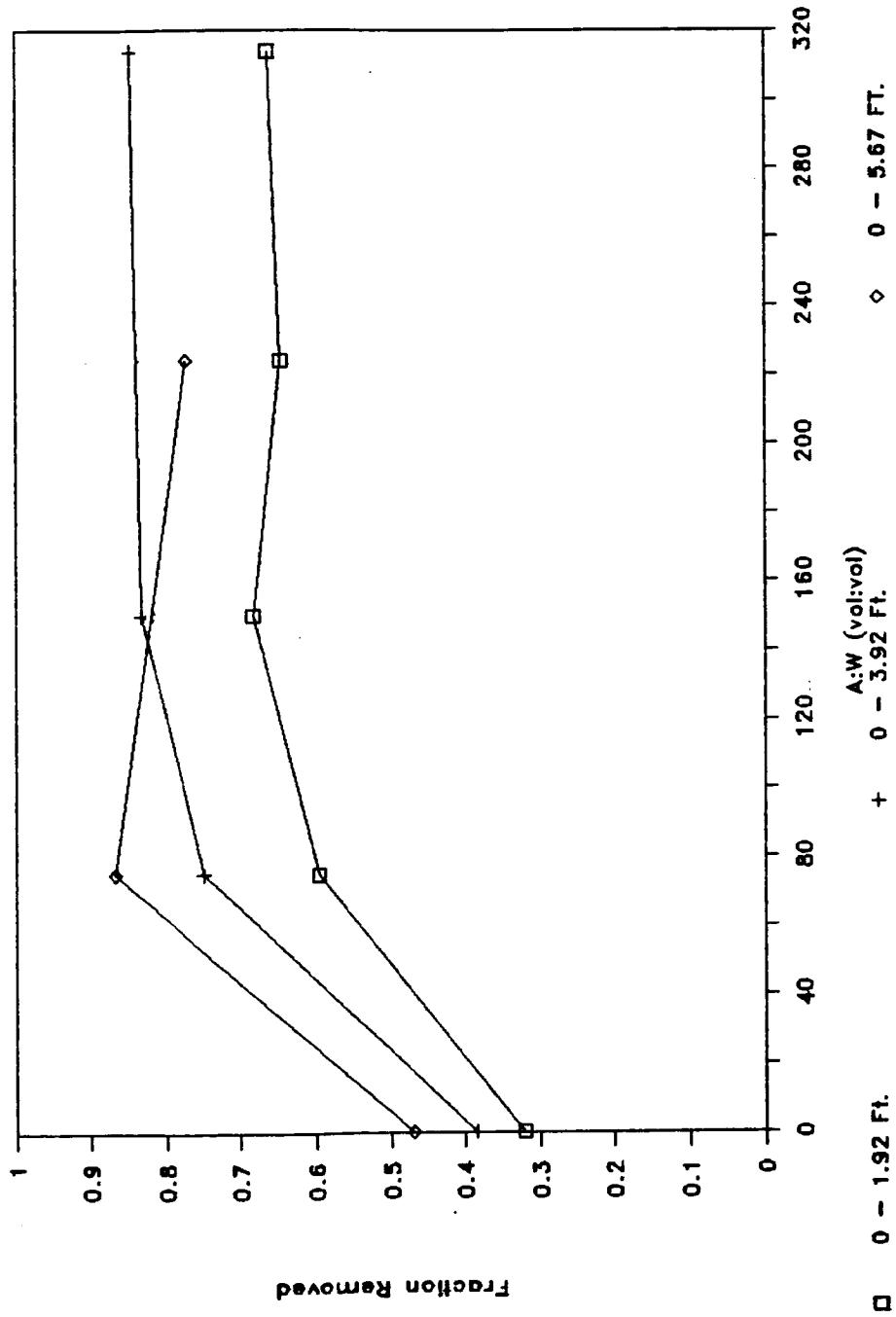


Figure 11. Experiments 1-5, Pall Rings, 1,4 DCB Removal Relative to Top of Packing vs. A:W.

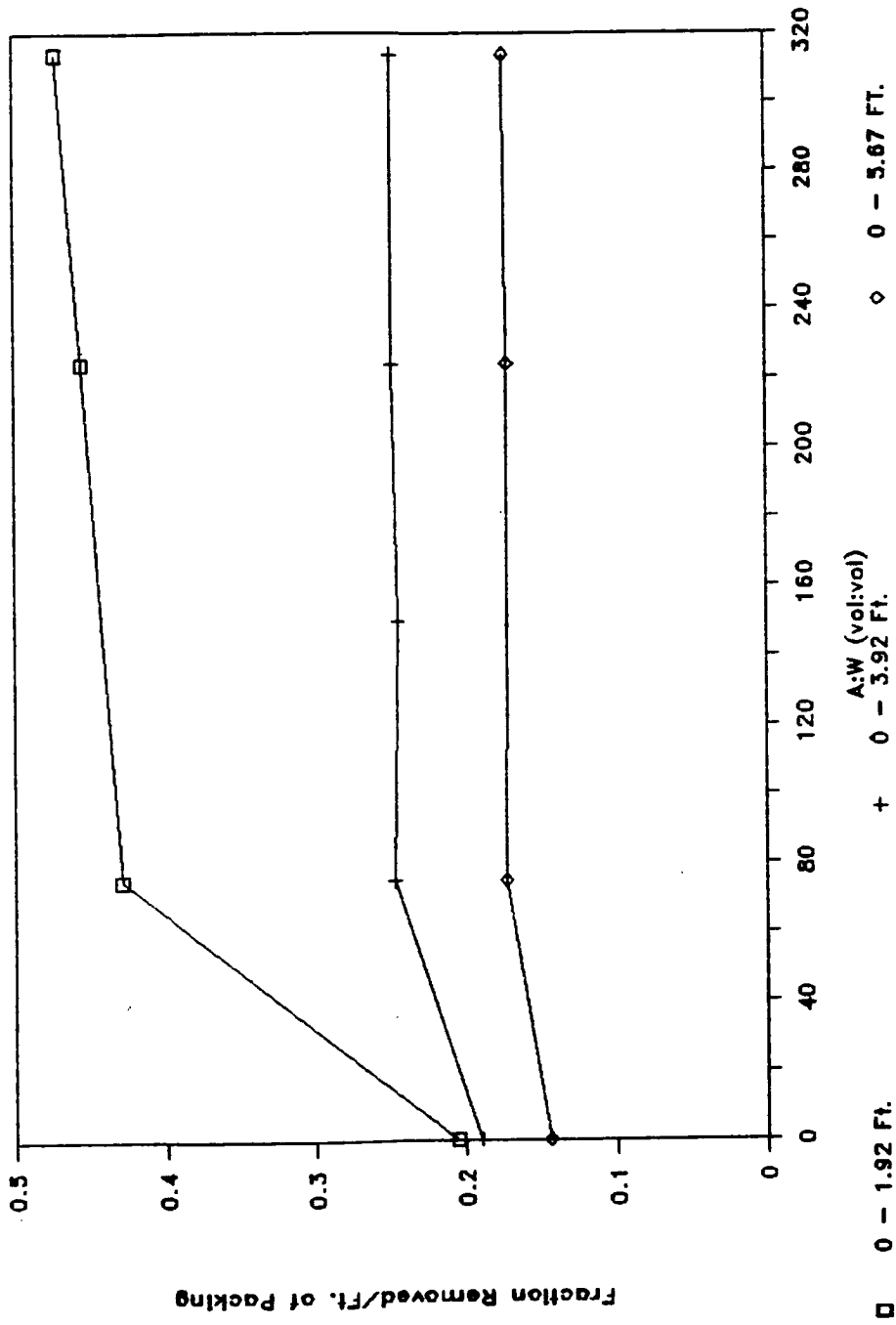


Figure 12. Experiments 1-5, Pall Rings, CT Removal per Foot Relative to Top of Packing vs. A:W.

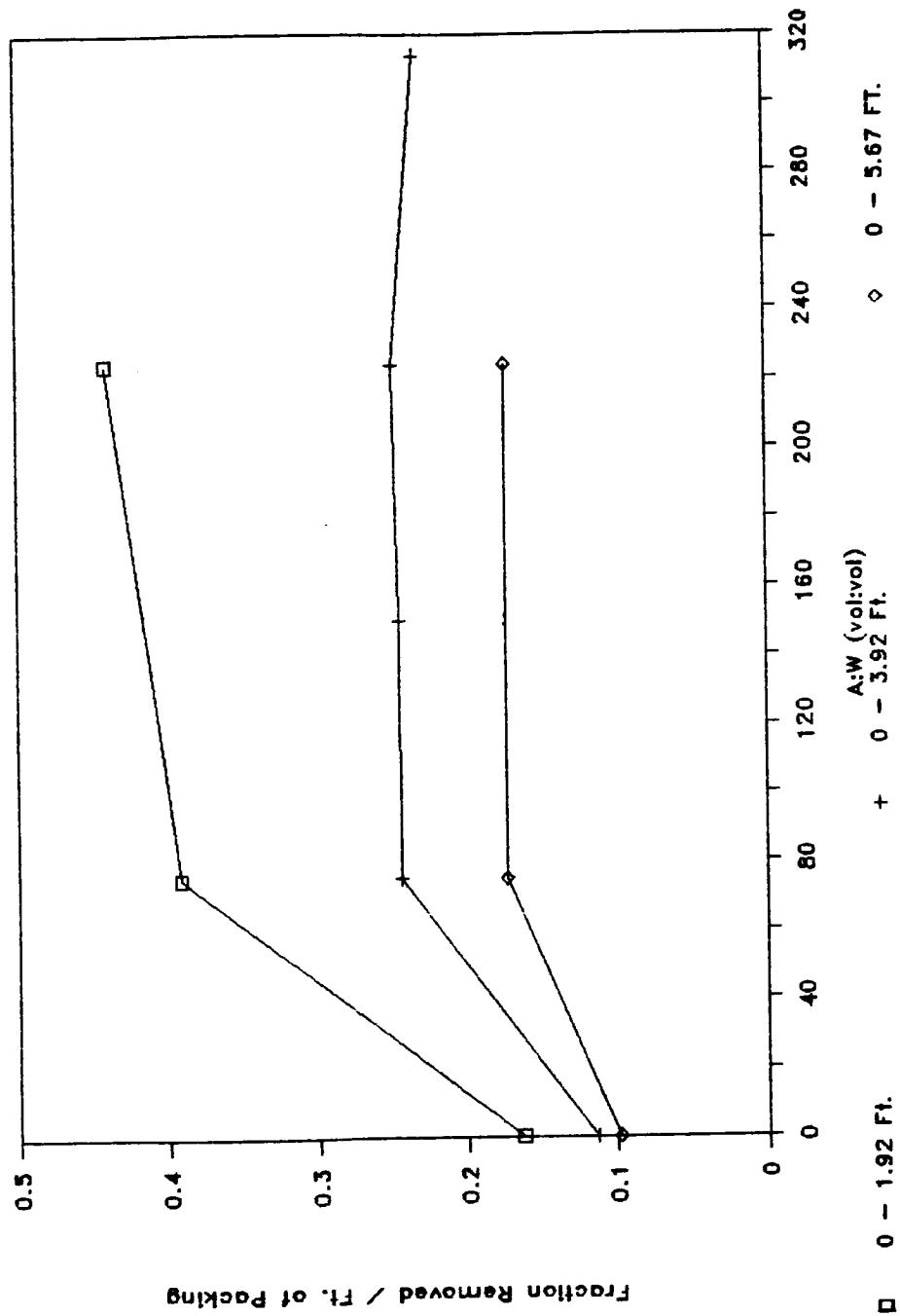


Figure 13. Experiments 1-5, Pall Rings, TCE Removal per Foot Relative to Top of Packing vs. A:W.

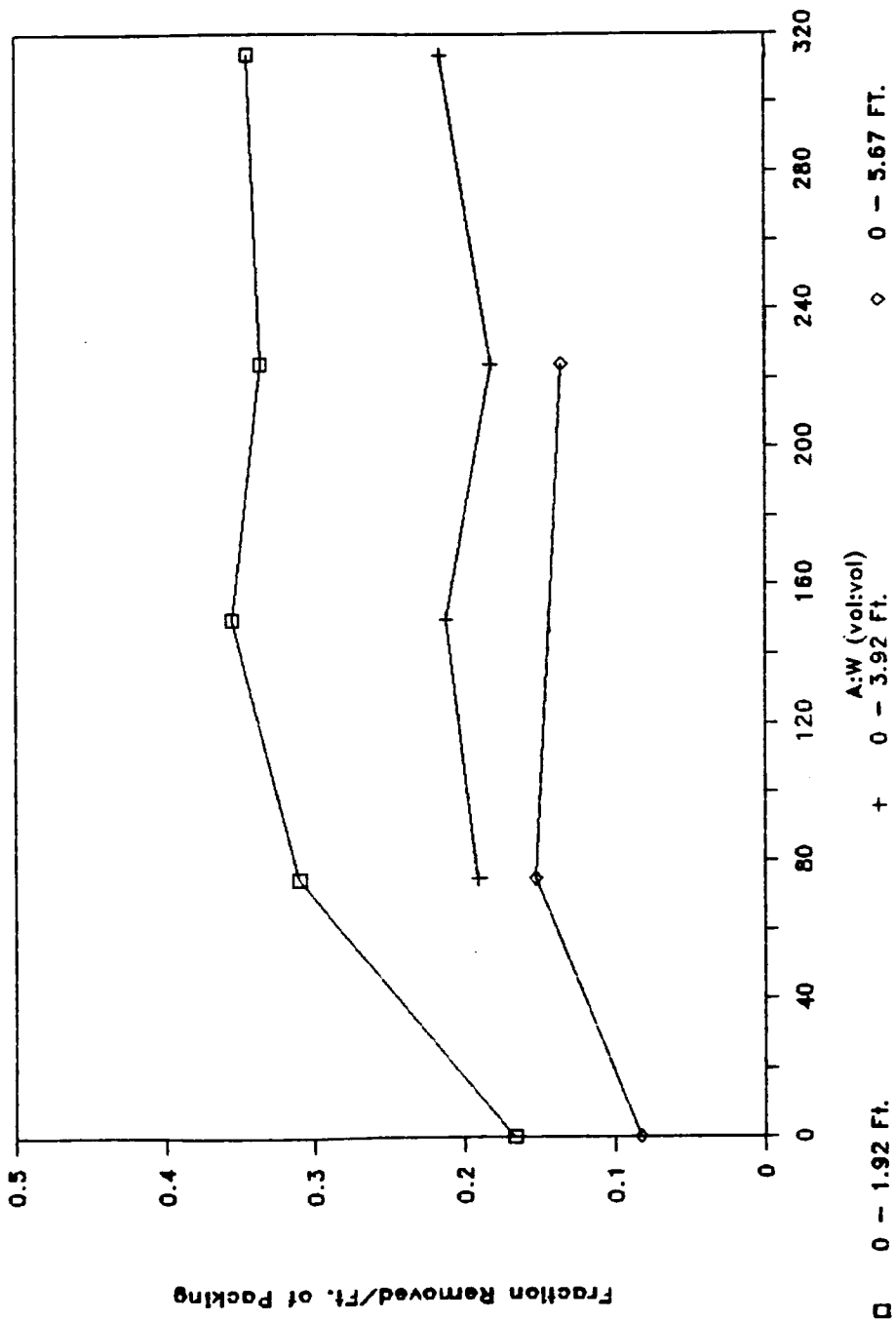


Figure 14. Experiments 1-5, Pall Rings, 1,4 DCB Removal per Foot Relative to Top of Packing vs. A:W.

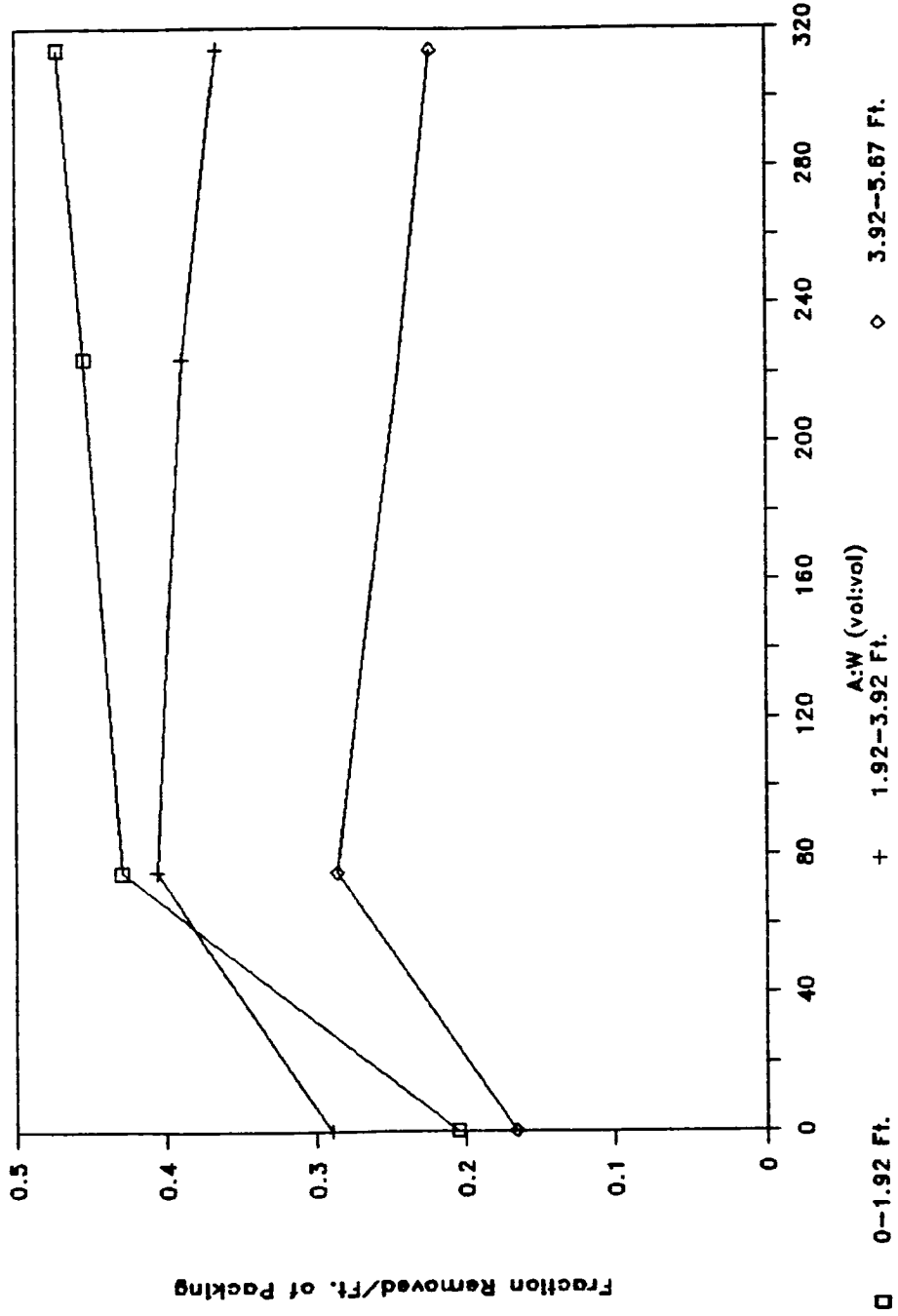


Figure 15. Experiments 1-5, Pall Rings, CT Removal per Foot, Incremental Between Sample Ports vs. A:W.

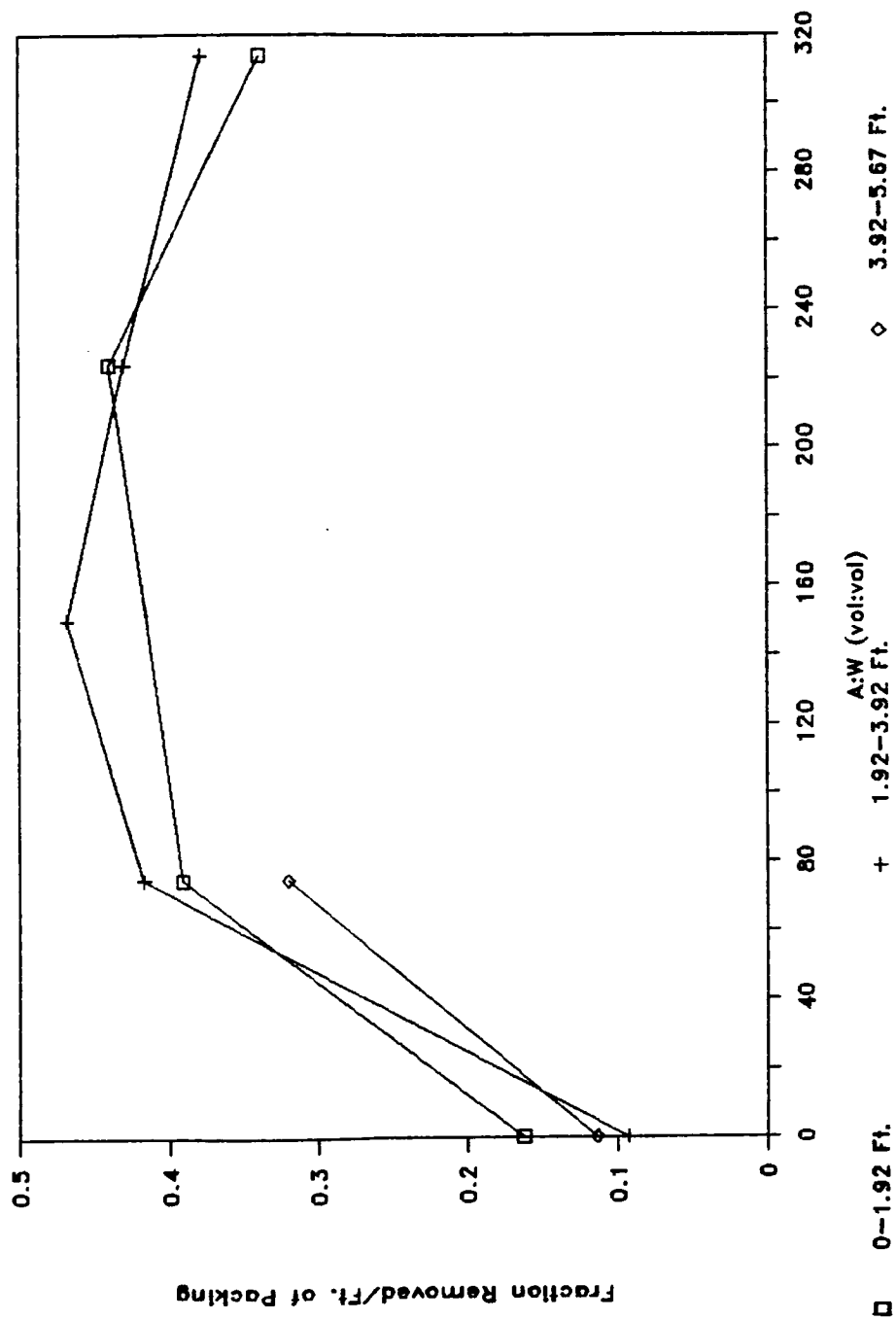


Figure 16. Experiments 1-5, Pall Rings, TCE Removal per Foot, Incremental Between Sample Ports vs. A:W.

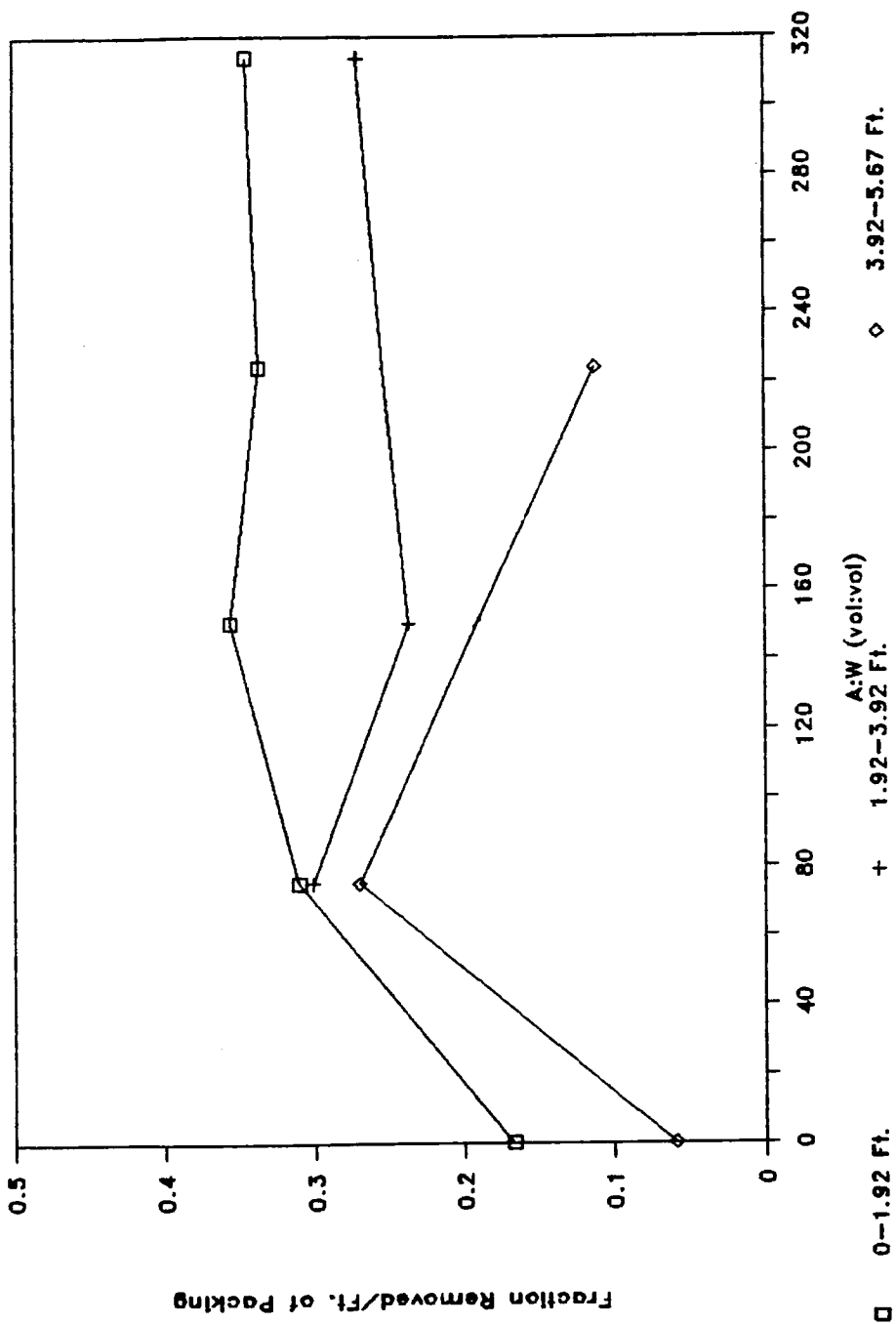


Figure 17. Experiments 1-5, Pall Rings, 1,4 DCB Removal per Foot, Incremental Between Sample Ports vs. A:W.

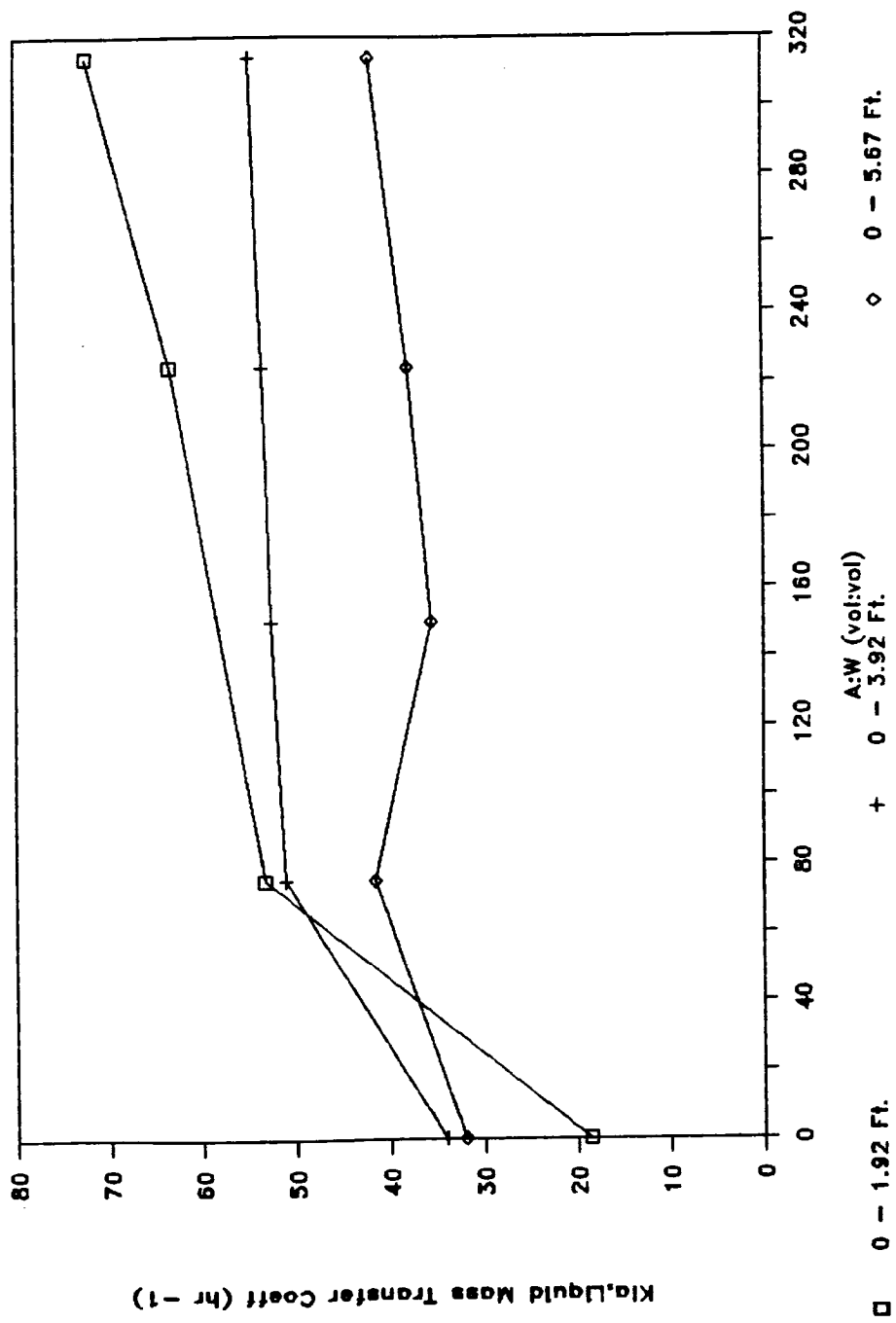


Figure 18. Experiments 1-5, Pall Rings, CT, K_L Relative to Top of Packing vs. A:W.

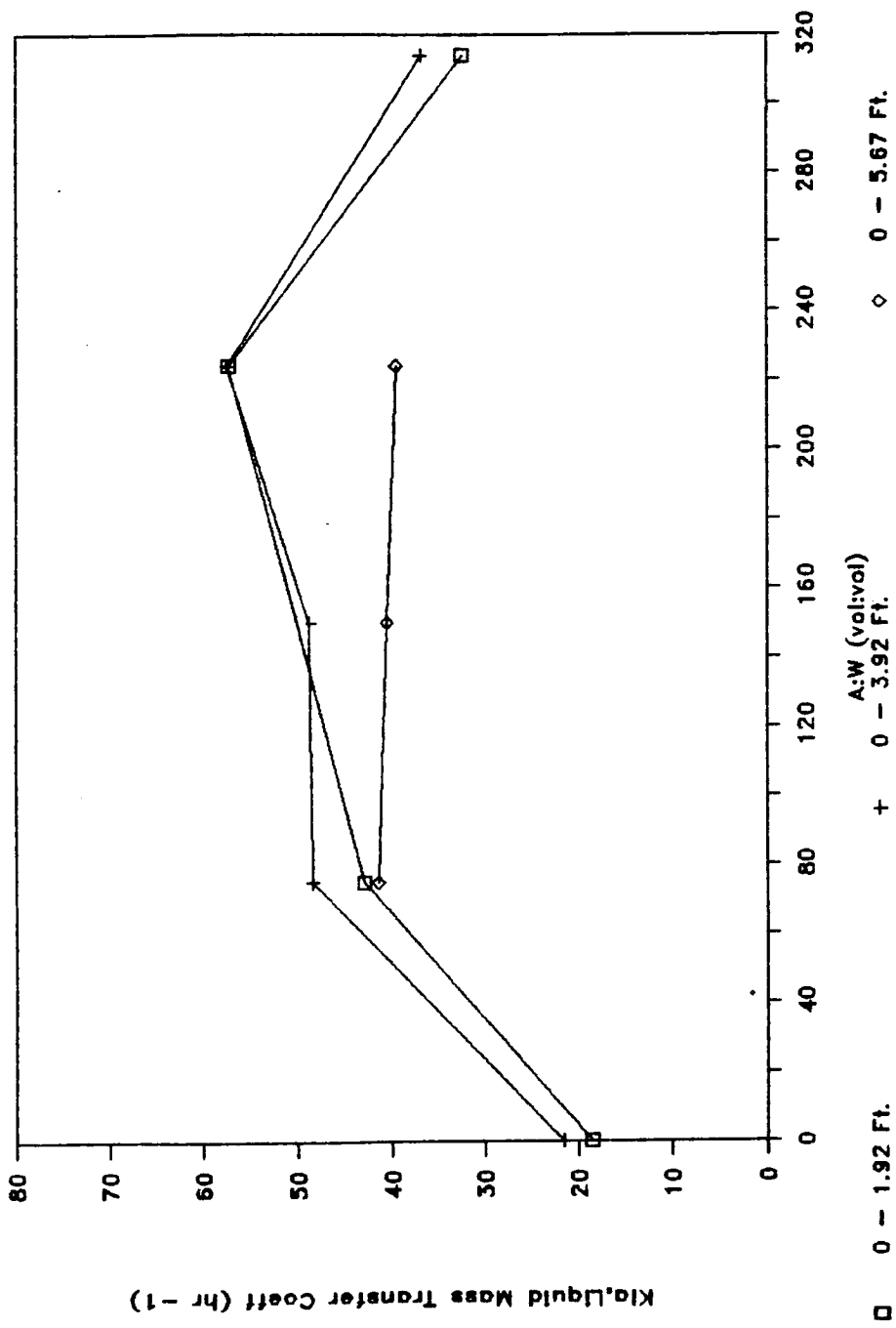


Figure 19. Experiments 1-5, Pall Rings, TCE, K_L a Relative to Top of Packing vs. $A:W$.

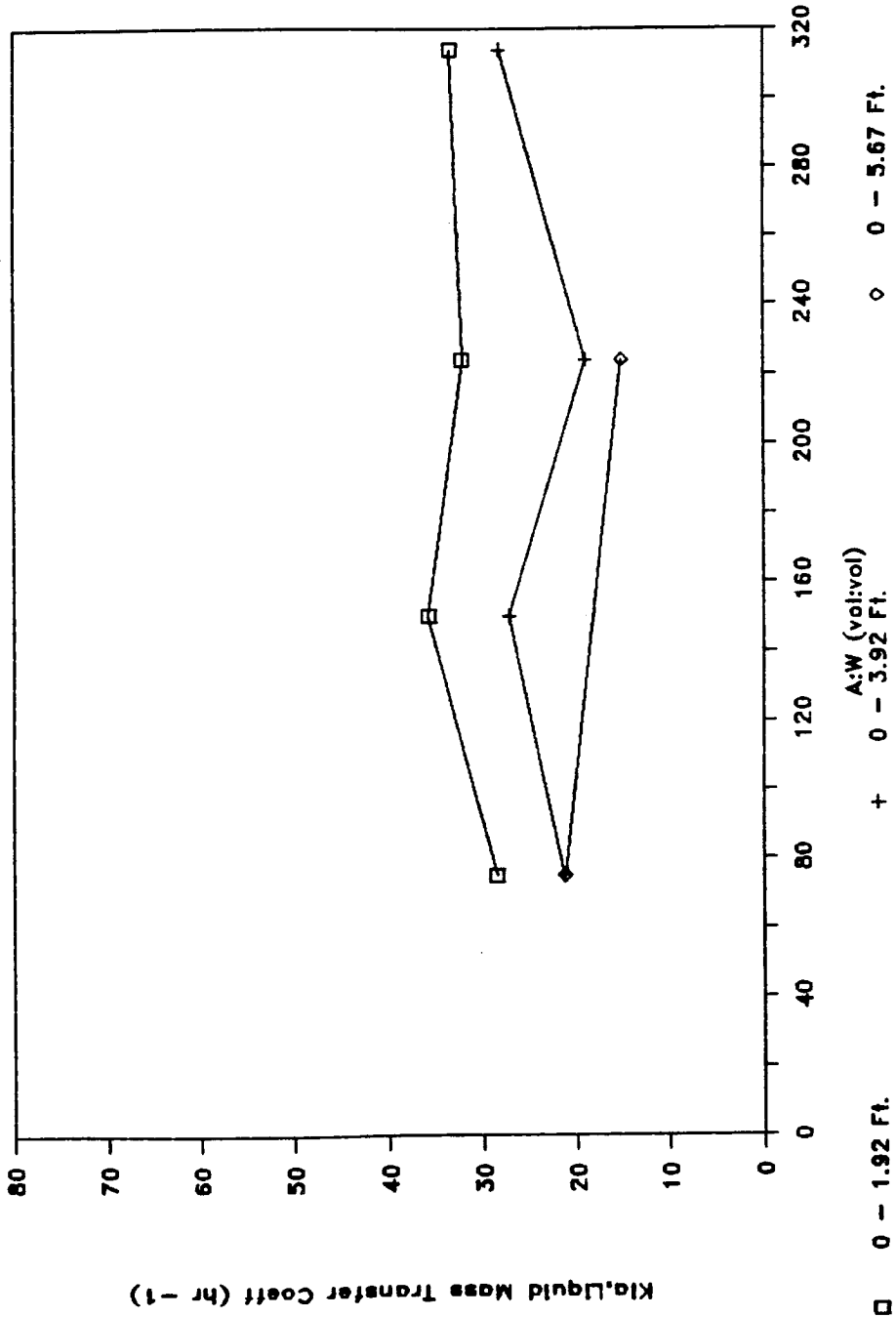


Figure 20. Experiments 1-5, Pall Rings, 1,4 DCB, K_L a Relative to Top of Packing vs. A:W.

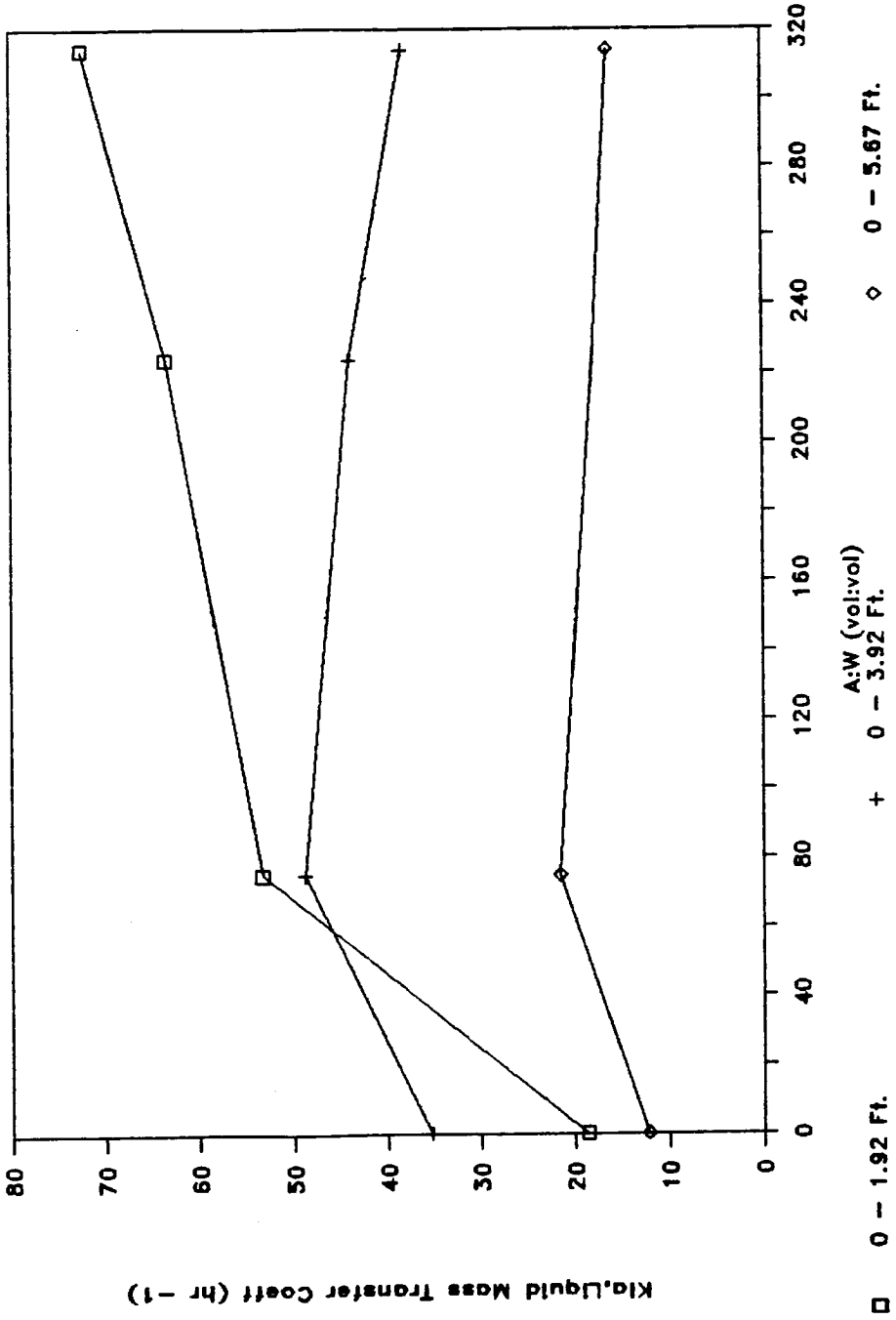


Figure 21. Experiments 1-5, Pall Rings, CT, K_L a Incremental Between Sample Ports vs. A:W.

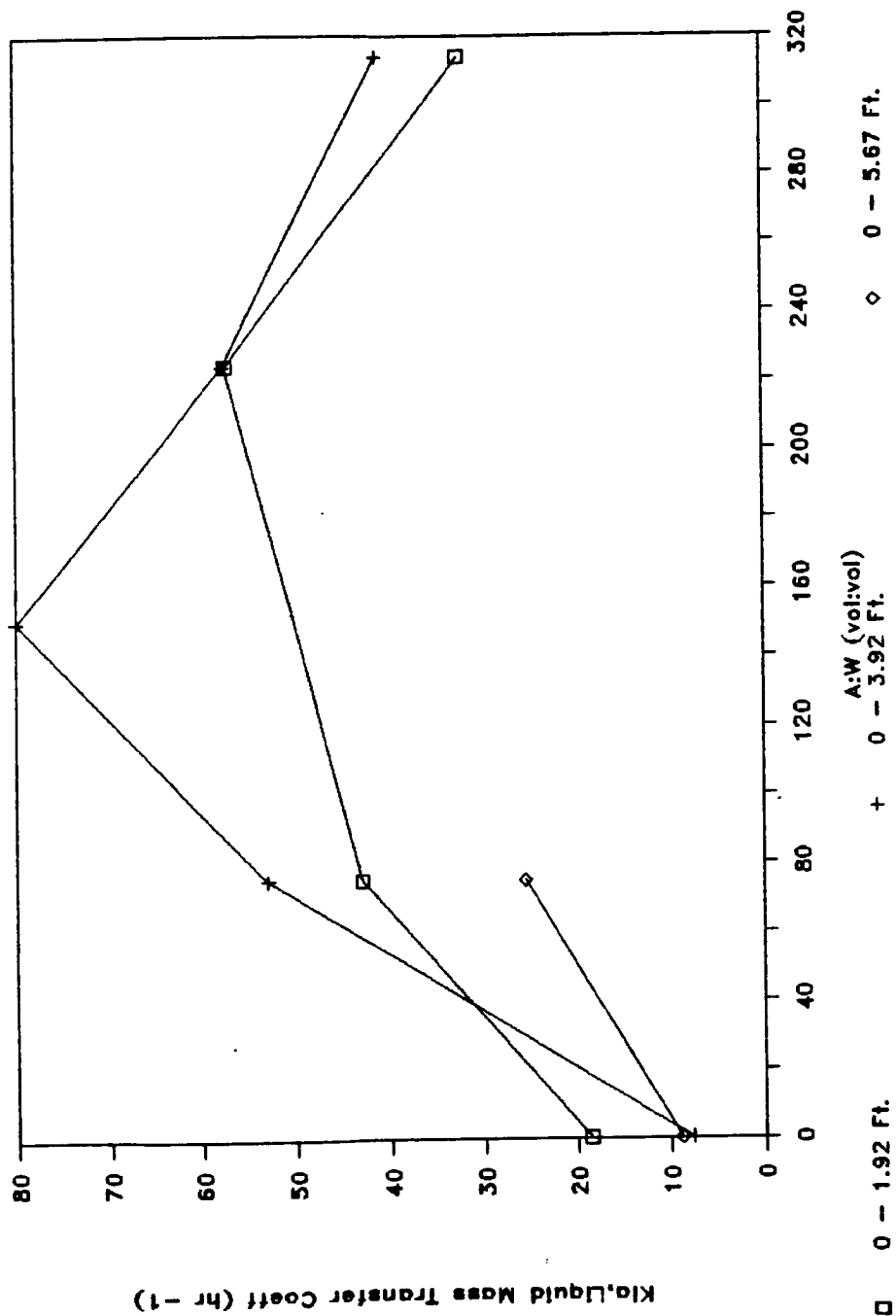


Figure 22. Experiments 1-5, Pall Rings, TCE, K_L a Incremental Between Sample Ports vs. A:W.

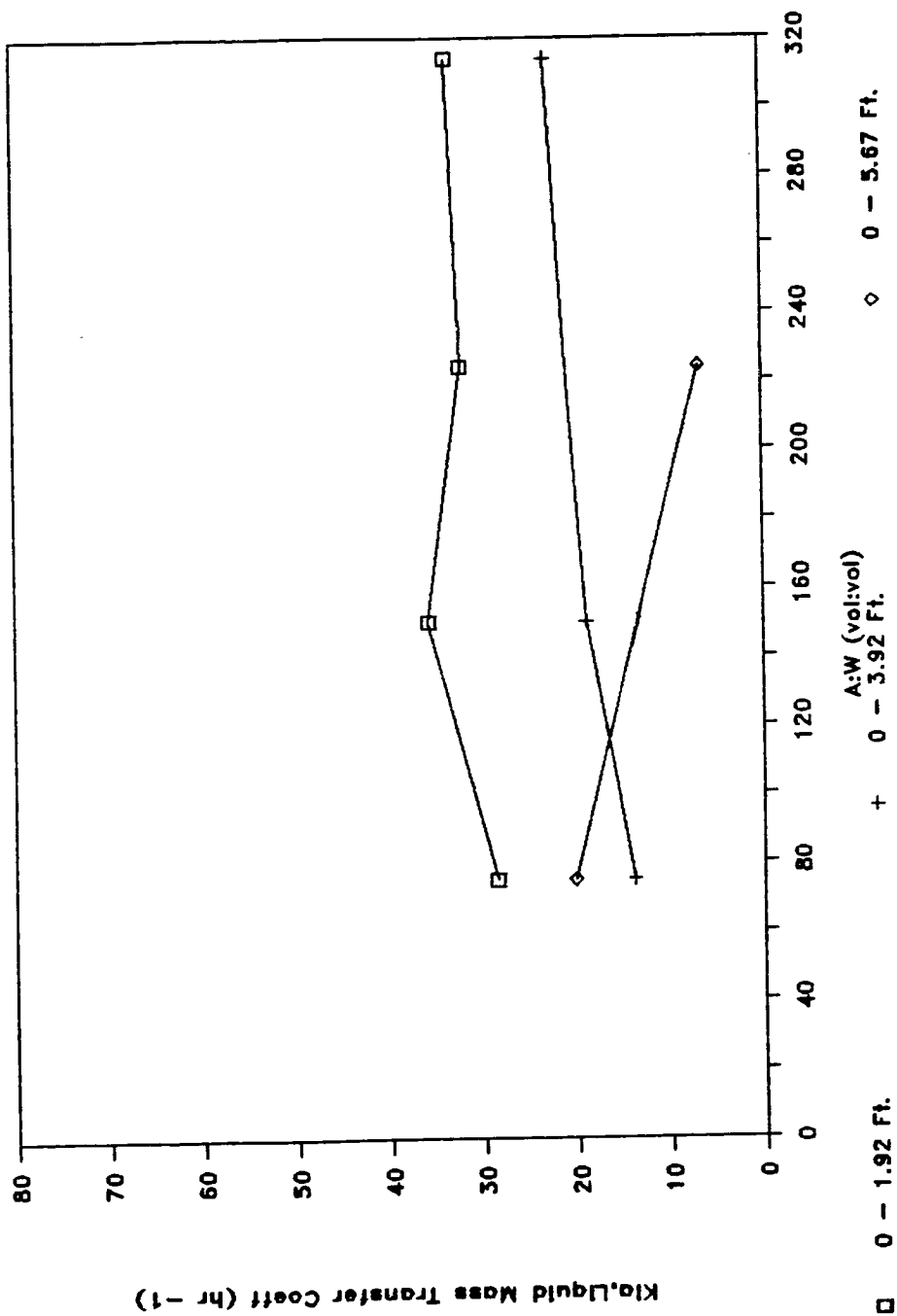


Figure 23. Experiments 1-5, Pall Rings, 1,4 DCB, K_L a Incremental Between Sample Ports vs. A:W.

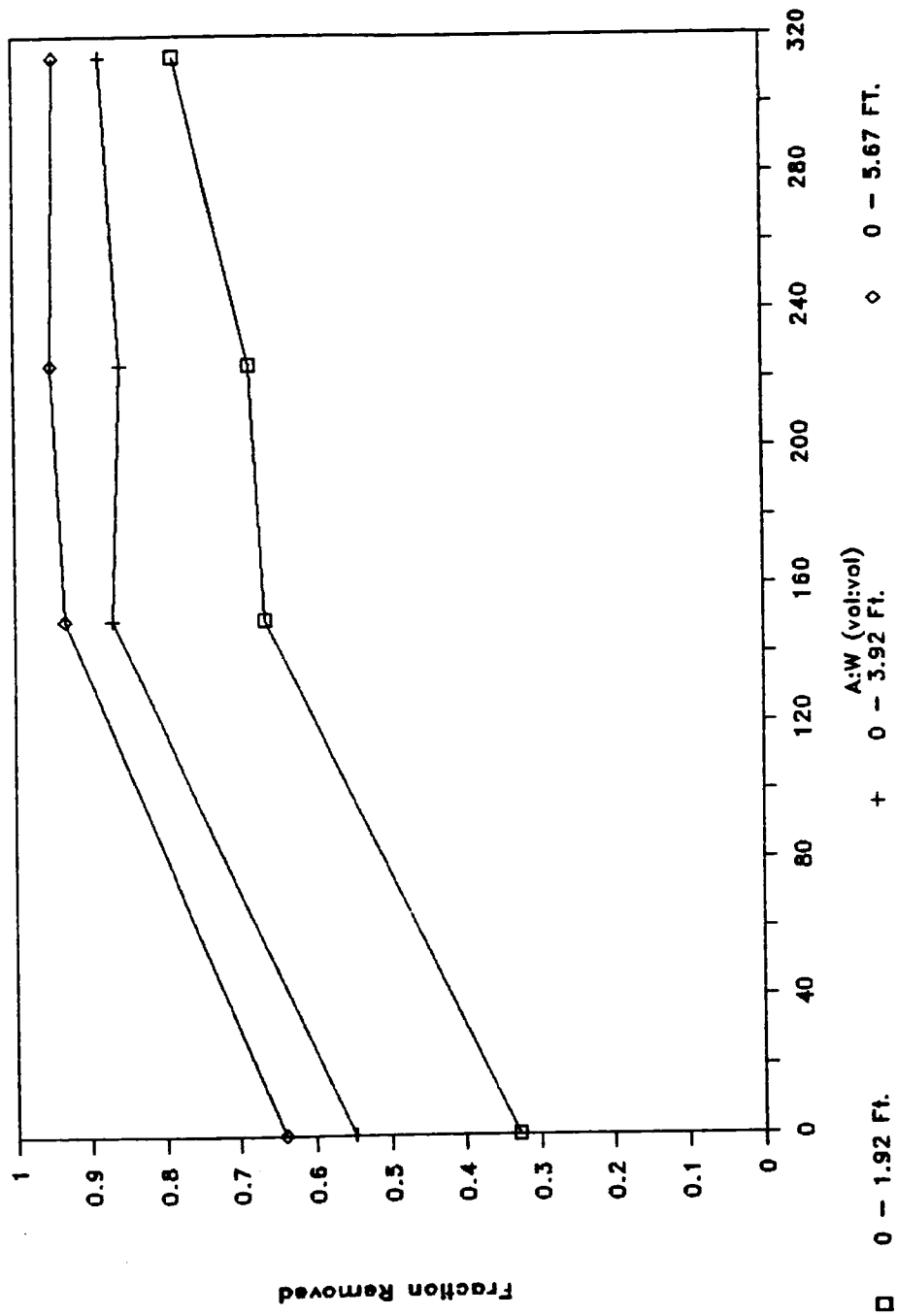


Figure 24. Experiments 6-9, Pall Rings, CT Removal Relative to Top of Packing vs. A:W.

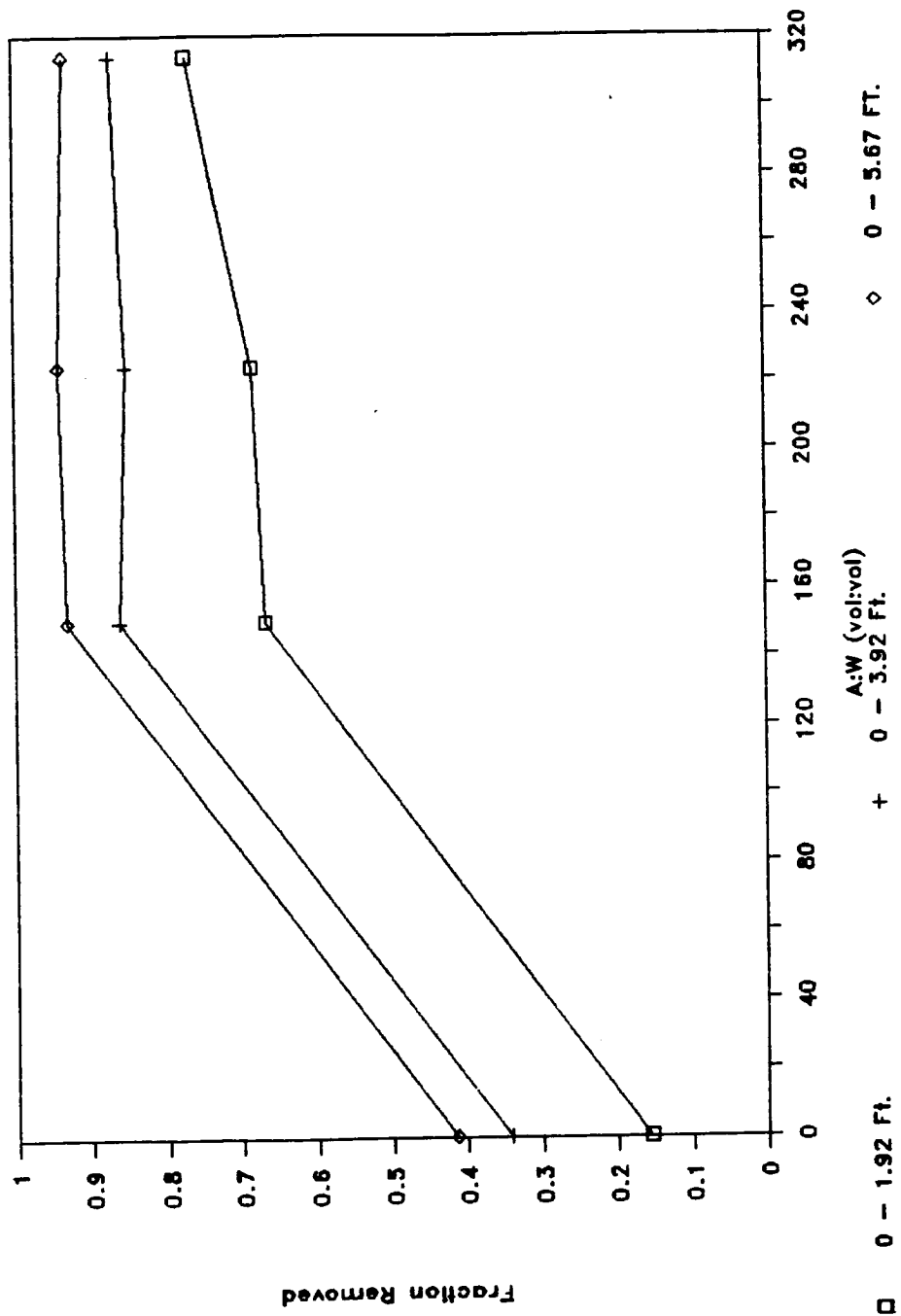


Figure 25. Experiments 6-9, Pall Rings, TCE Removal Relative to Top of Packing vs. A:W.

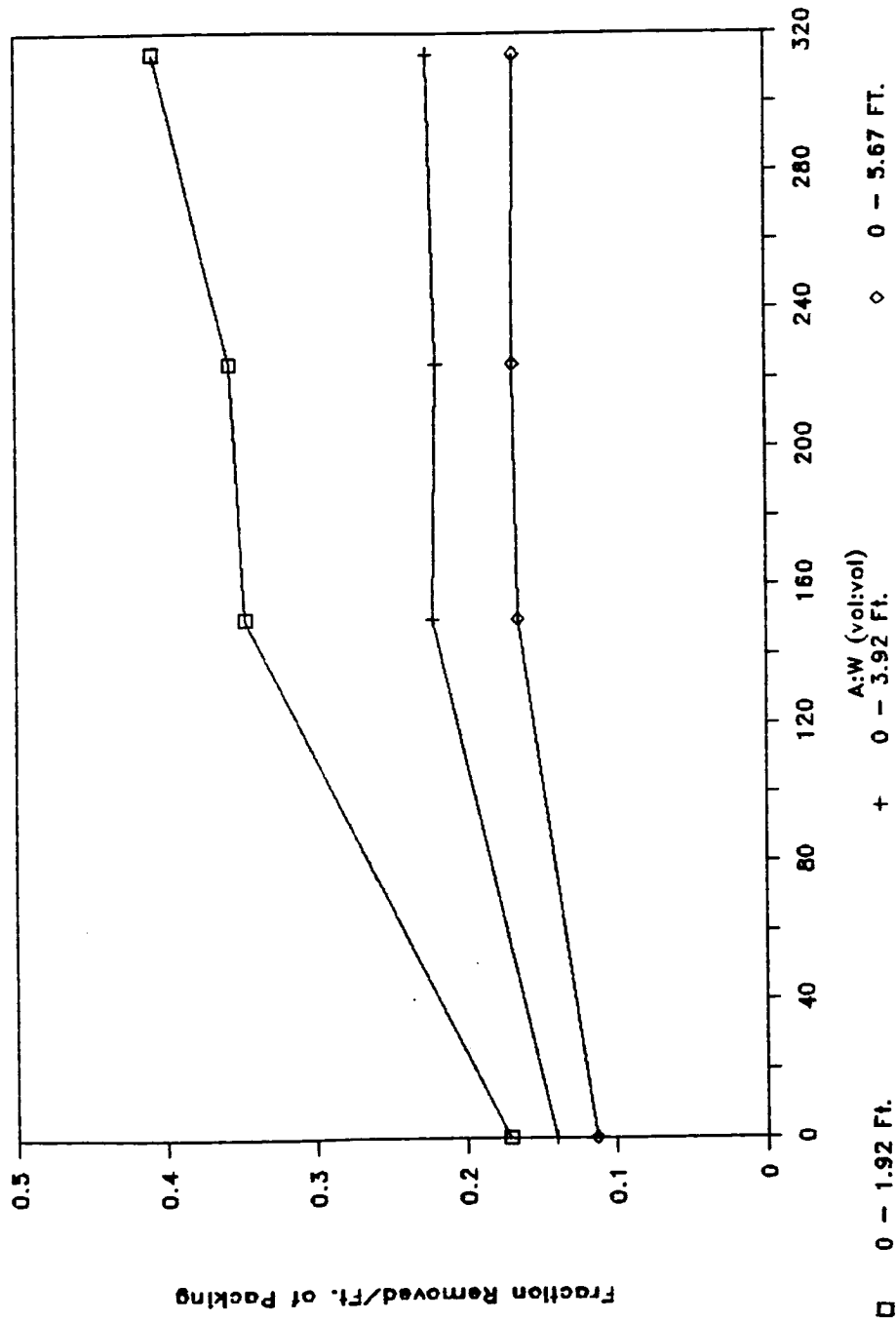


Figure 26. Experiments 6-9, Pall Rings, CT Removal per Foot Relative to Top of Packing vs. A:W.

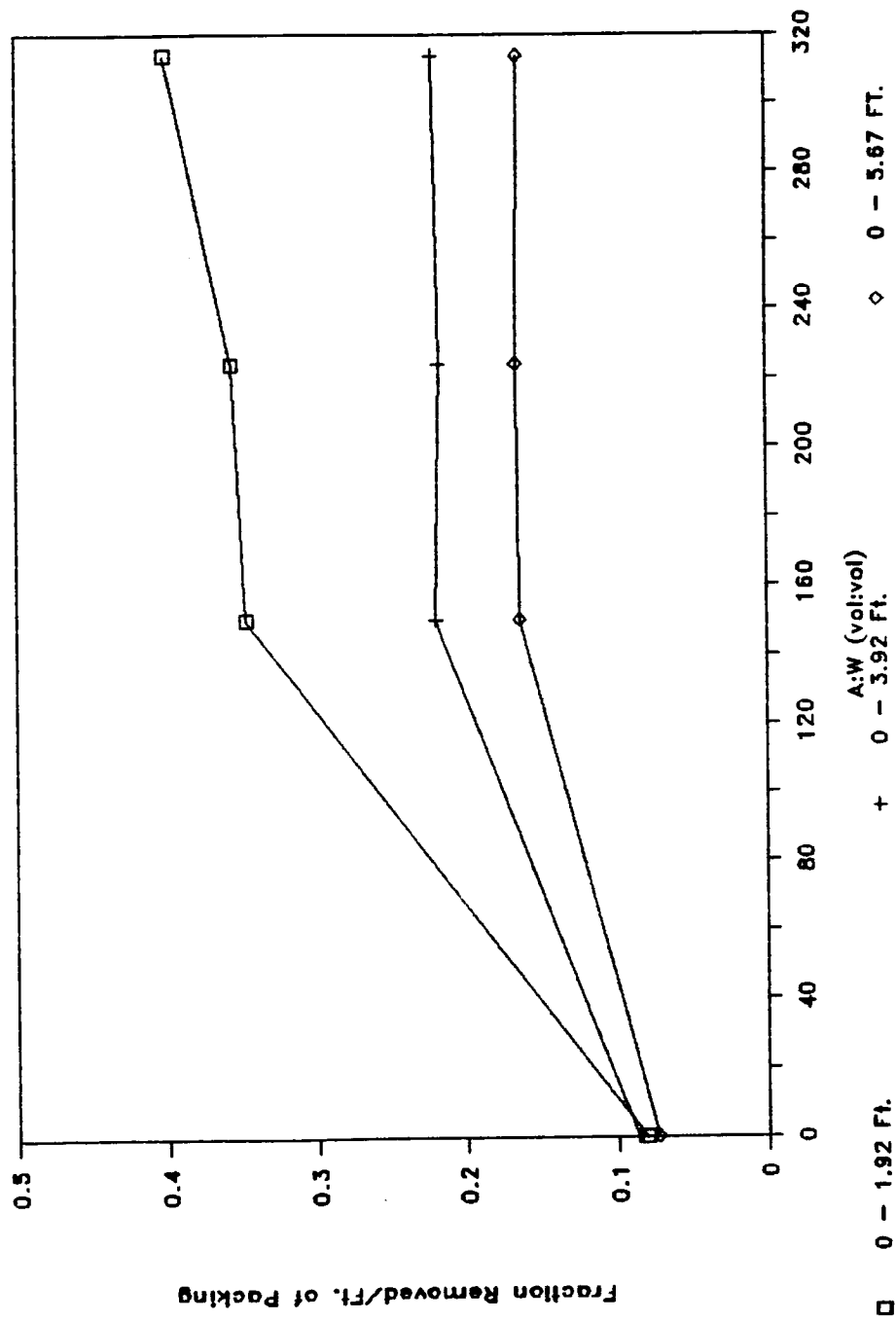


Figure 27. Experiments 6-9, Pall Rings, TCE Removal per Foot Relative to Top of Packing vs. A:W.

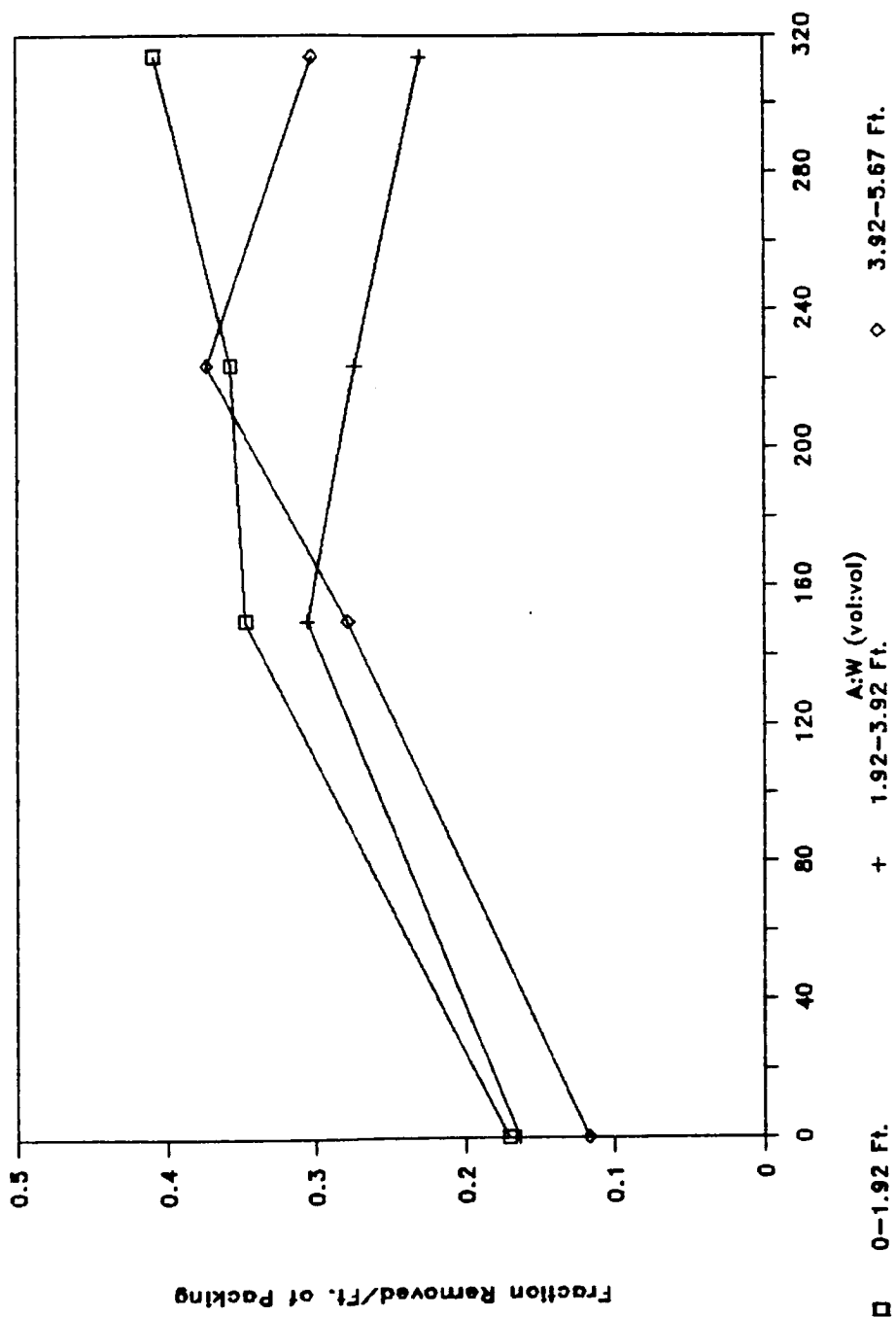


Figure 28. Experiments 6-9, Pall Rings, CT Removal per Foot, Incremental Between Sample Ports vs. A:W.

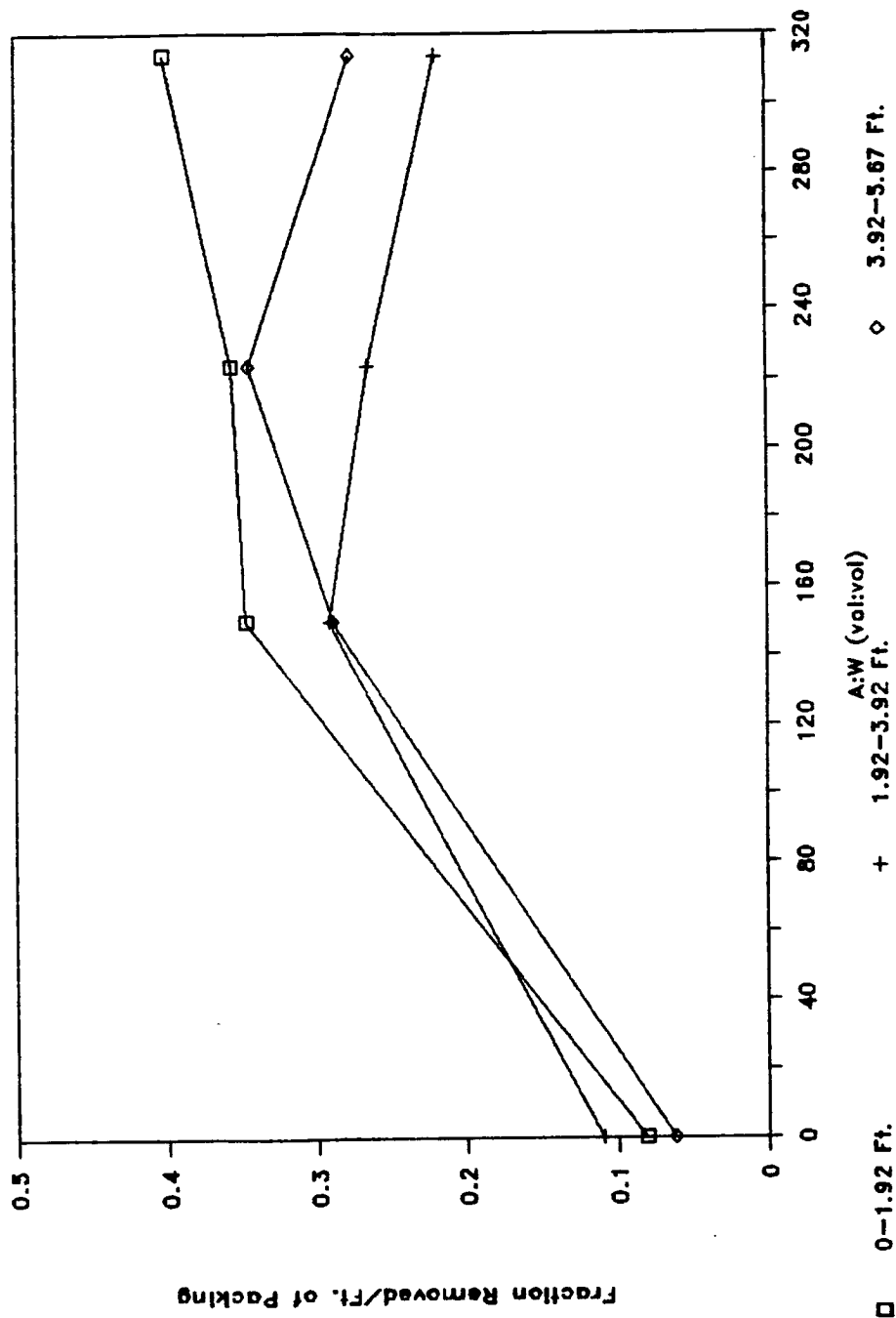


Figure 29. Experiments 6-9, Pall Rings, TCE Removal per Foot, Relative to Top of Packing vs. A:W.

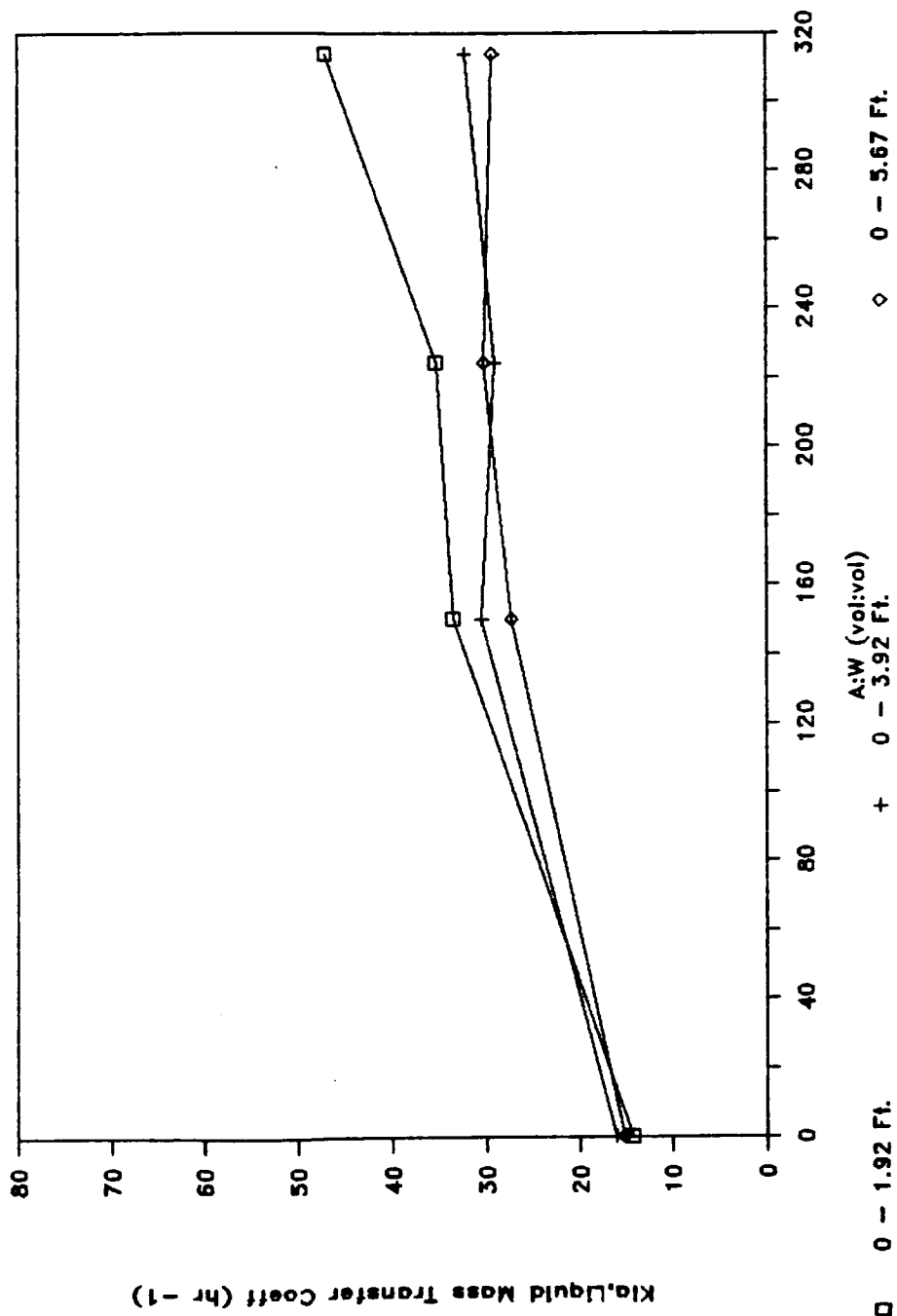


Figure 30. Experiments 6-9, Pall Rings, CT, K_L a Relative to Top of Packing vs. A:W.

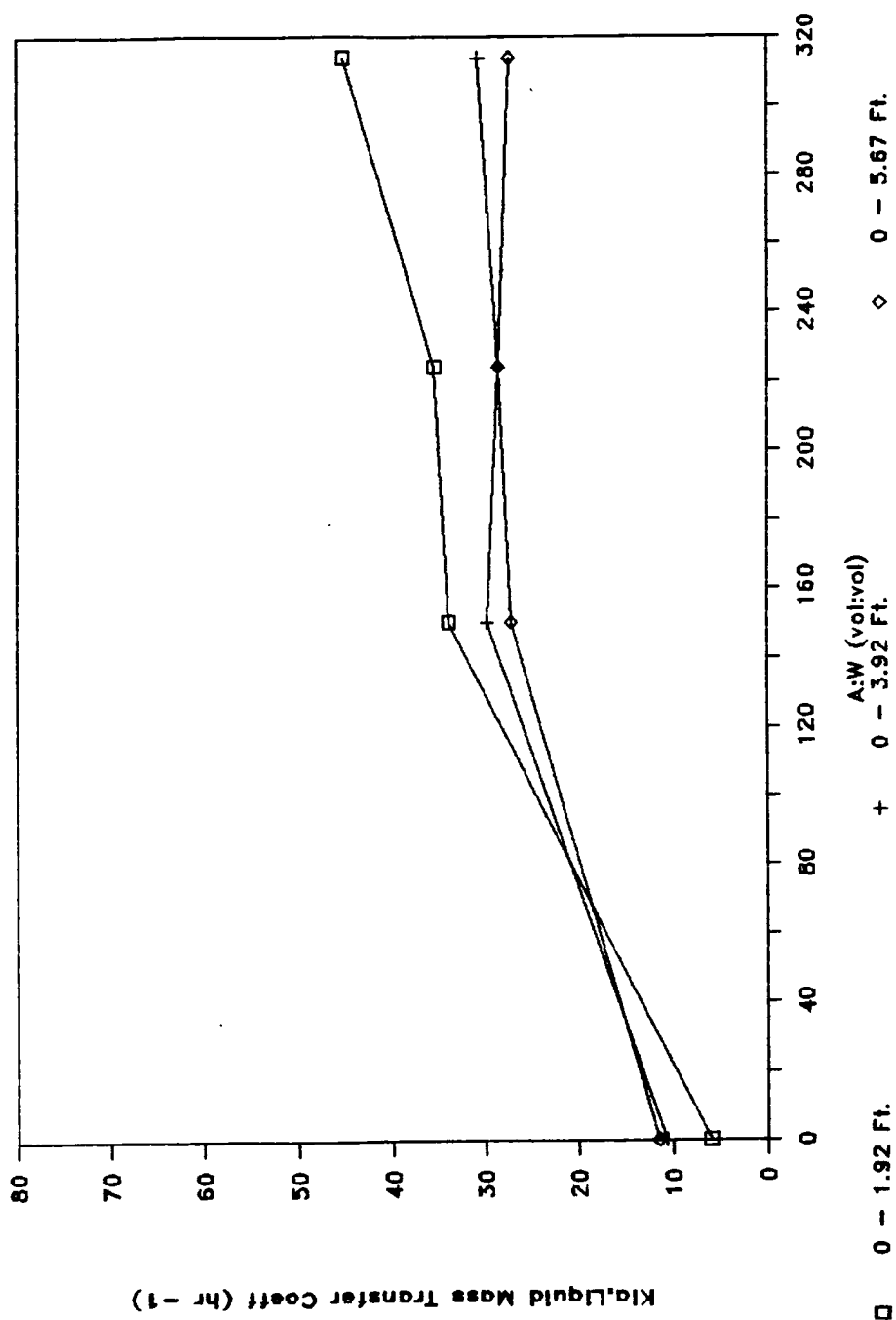


Figure 31. Experiments 6-9, Pall Rings, TCE, K_L Relative to Top of Packing vs. A:W.

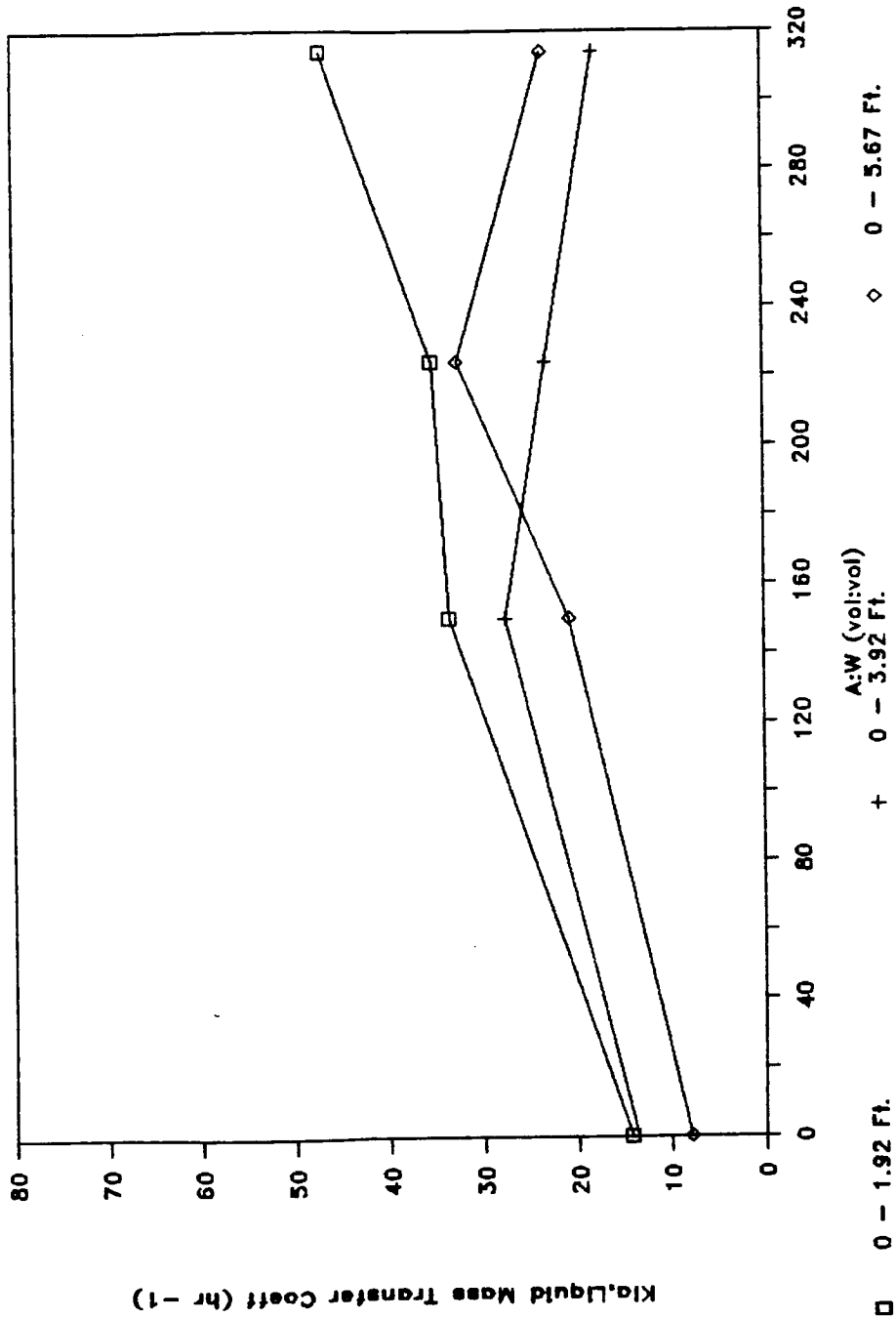


Figure 32. Experiments 6-9, Pall Rings, CT, K_L a Incremental Between Sample Ports vs. A:W.

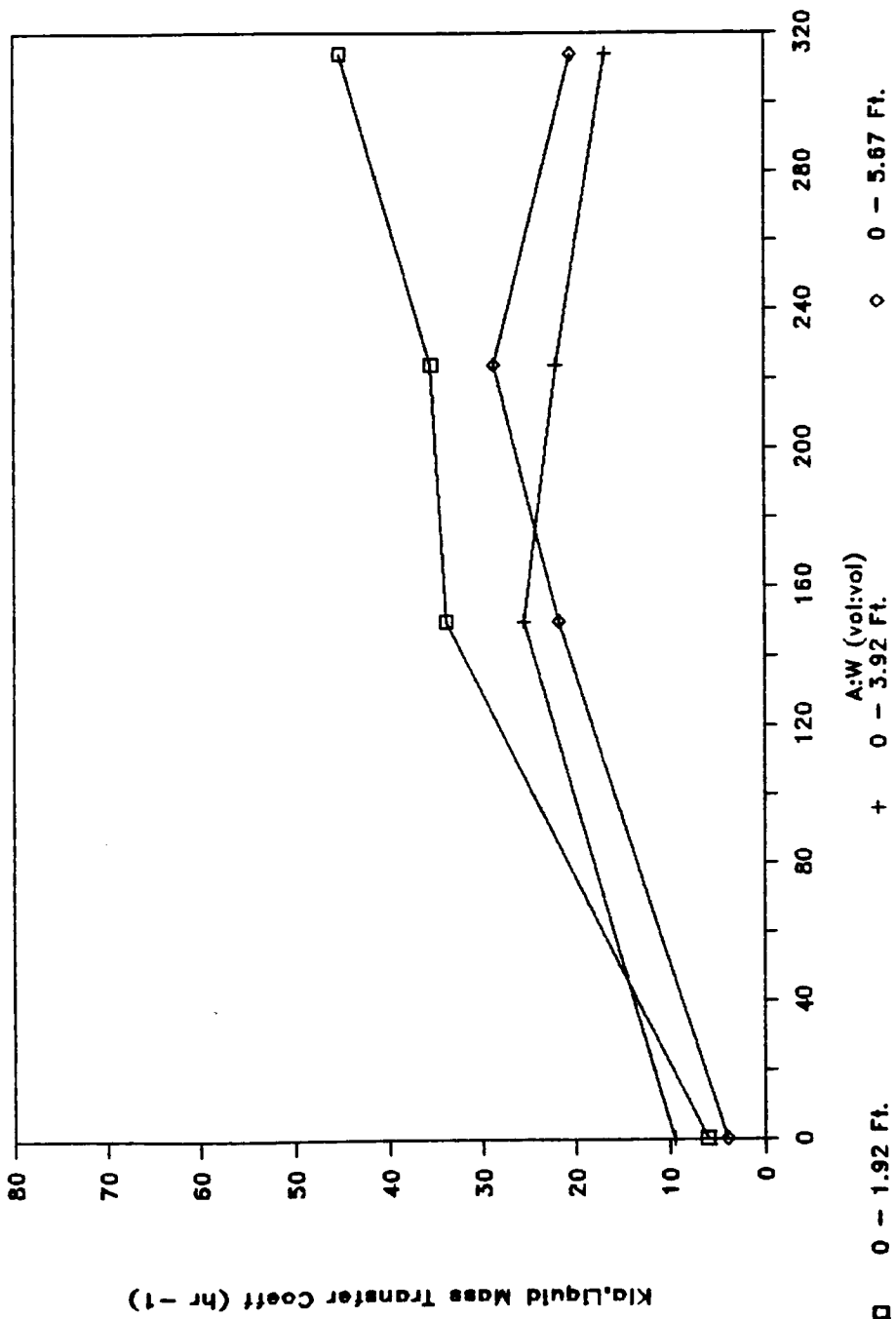


Figure 33. Experiments 6-9, Pall Rings, TCE, K_L a Incremental Between Sample Ports vs. A:W.

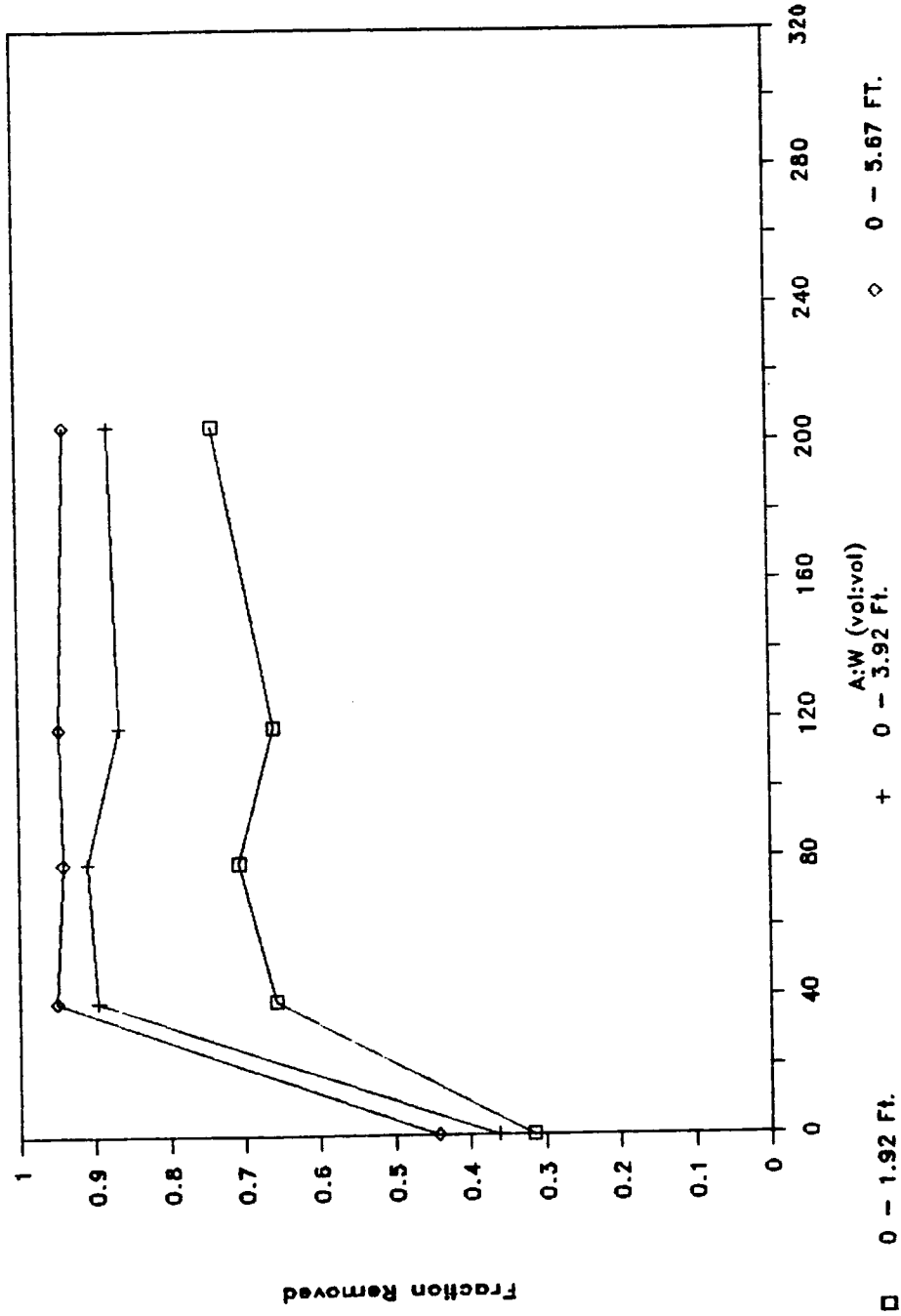


Figure 34. Experiments 10-14, Tripacks, CT Removal Relative to Top of Packing vs. A:W.

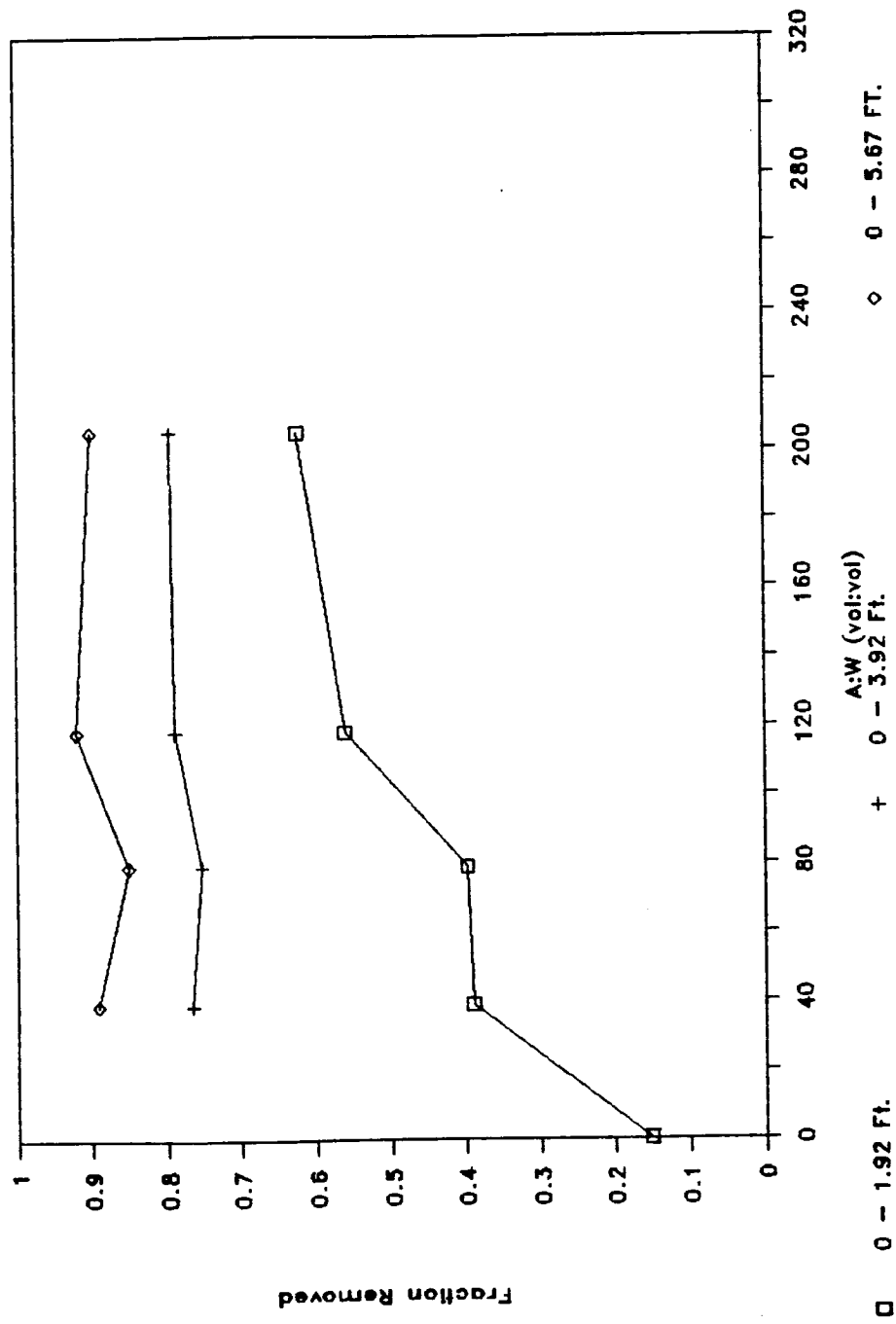


Figure 35. Experiments 10-14, Tripacks, TCE Removal Relative to Top of Packing vs. A:W.

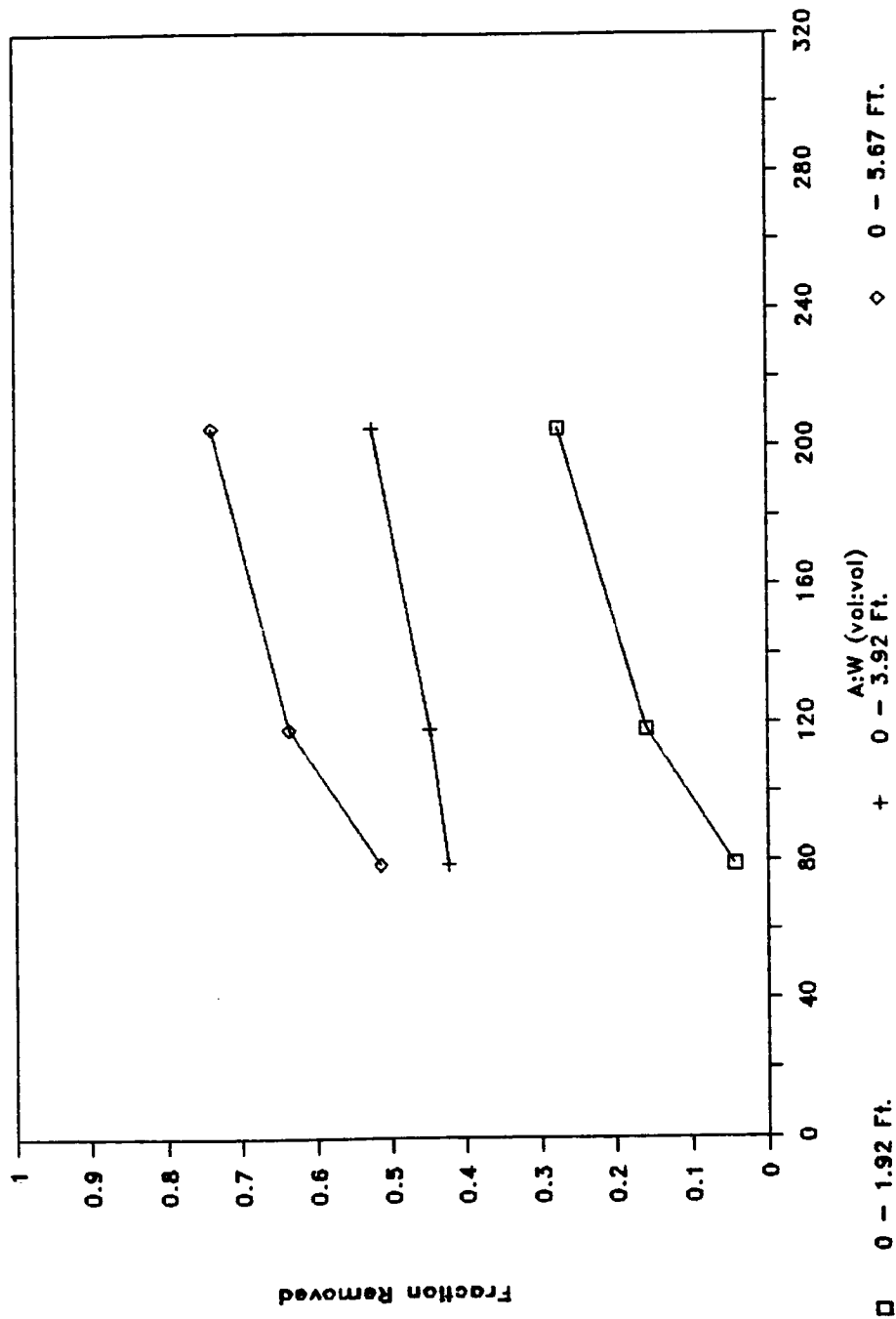


Figure 36. Experiments 10-14, Tripacks, 1,4 DCB Removal Relative to Top of Packing vs. A:W.

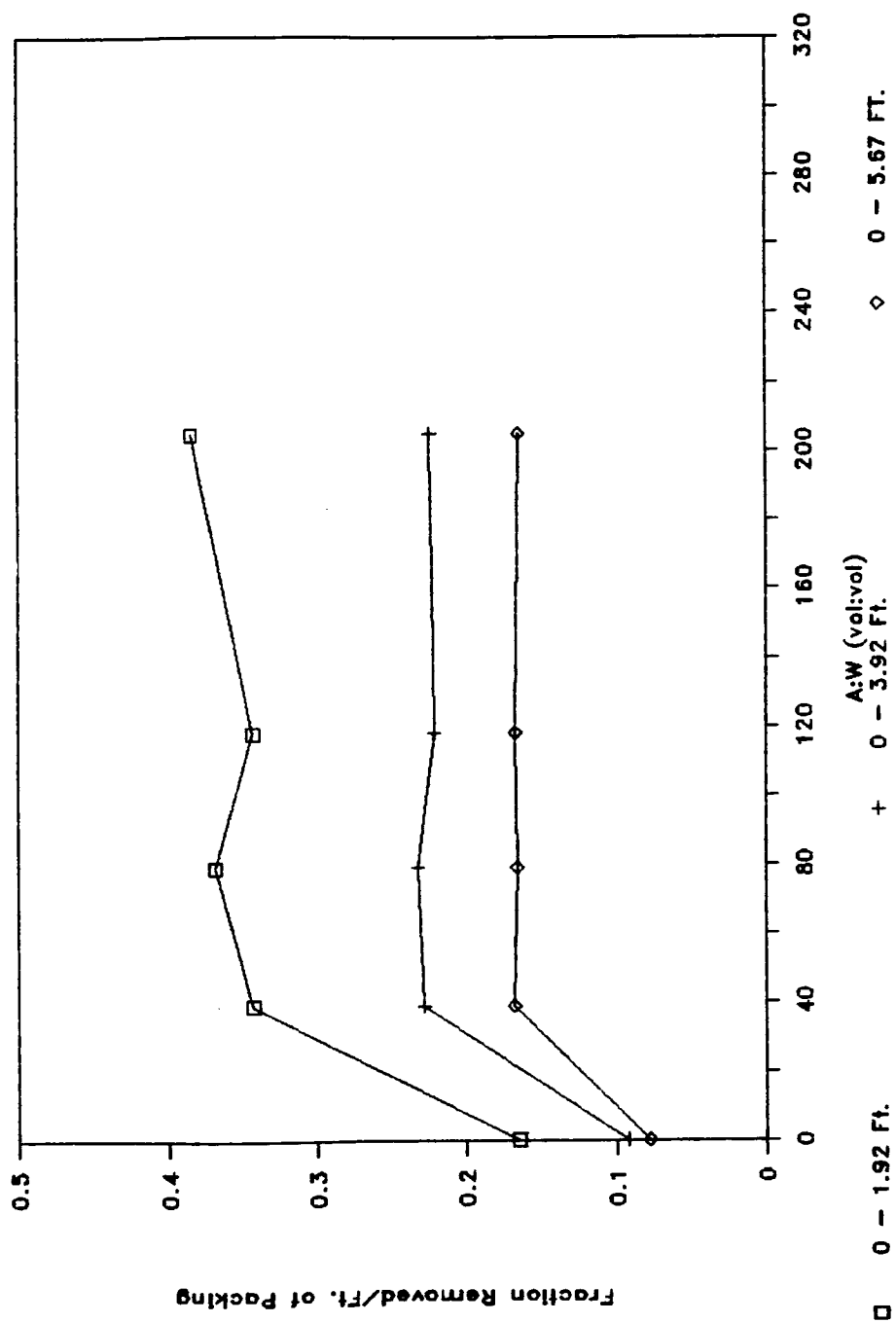


Figure 37. Experiments 10-14, Tripacks, CT Removal per Foot Relative to Top of Packing vs. A:W.

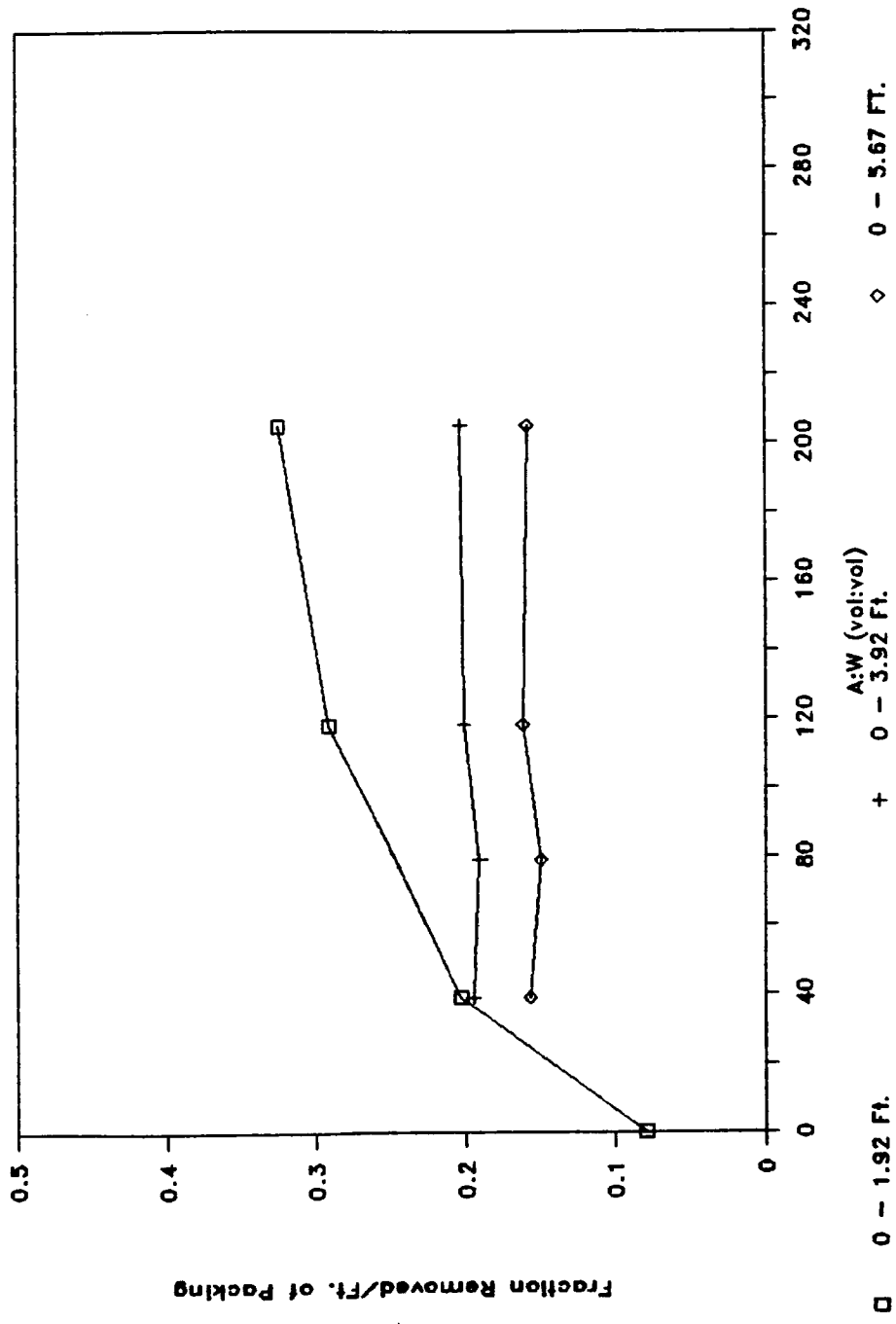


Figure 38. Experiments 10-14, Tripacks, TCE Removal per Foot Relative to Top of Packing vs. A:W.

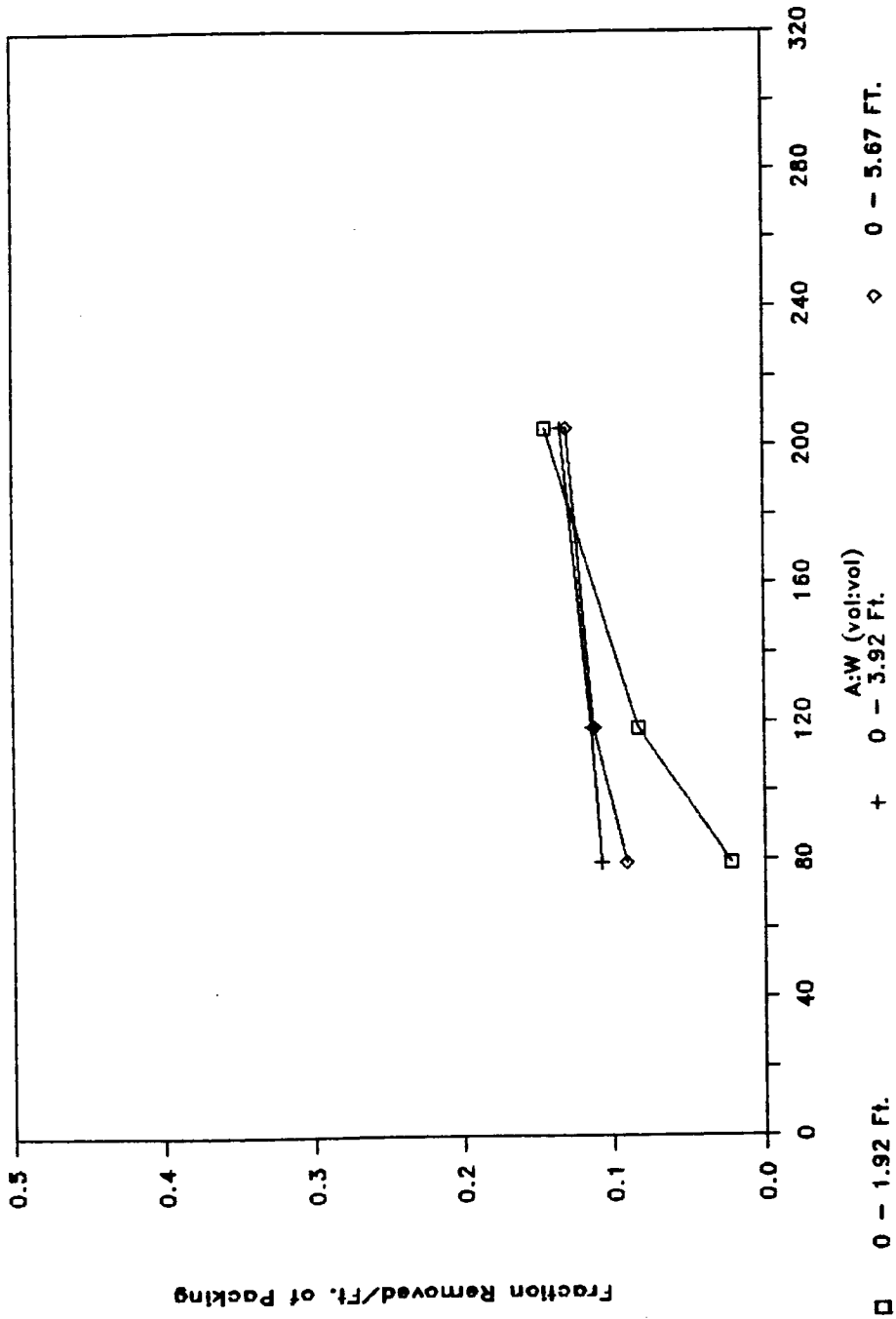


Figure 39. Experiments 10-14, Tripacks, 1,4 DCB Removal per Foot Relative to Top of Packing vs. A:W.

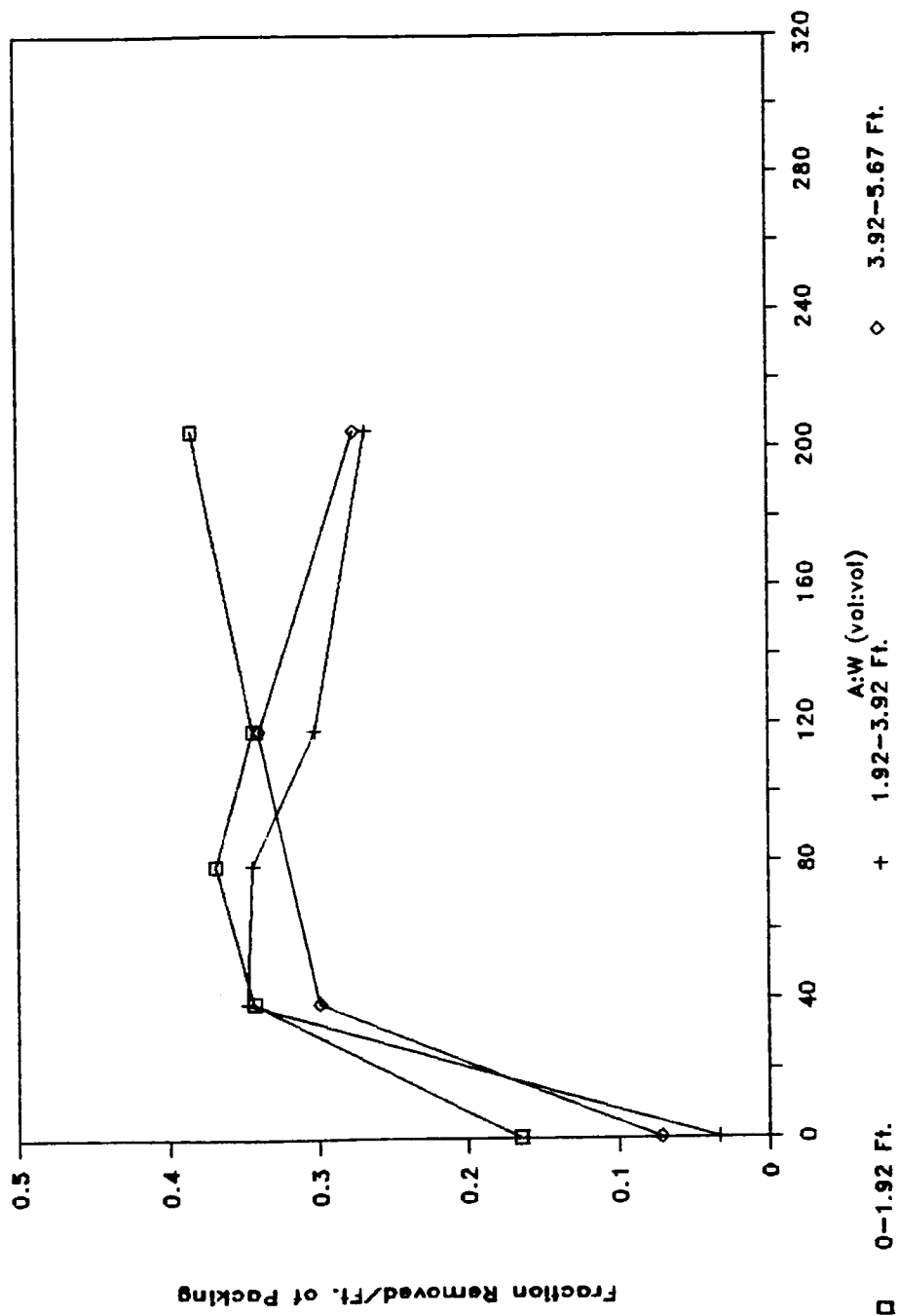


Figure 40. Experiments 10-14, Tripacks, CT Removal per Foot Incremental Between Sample Ports vs. A:W.

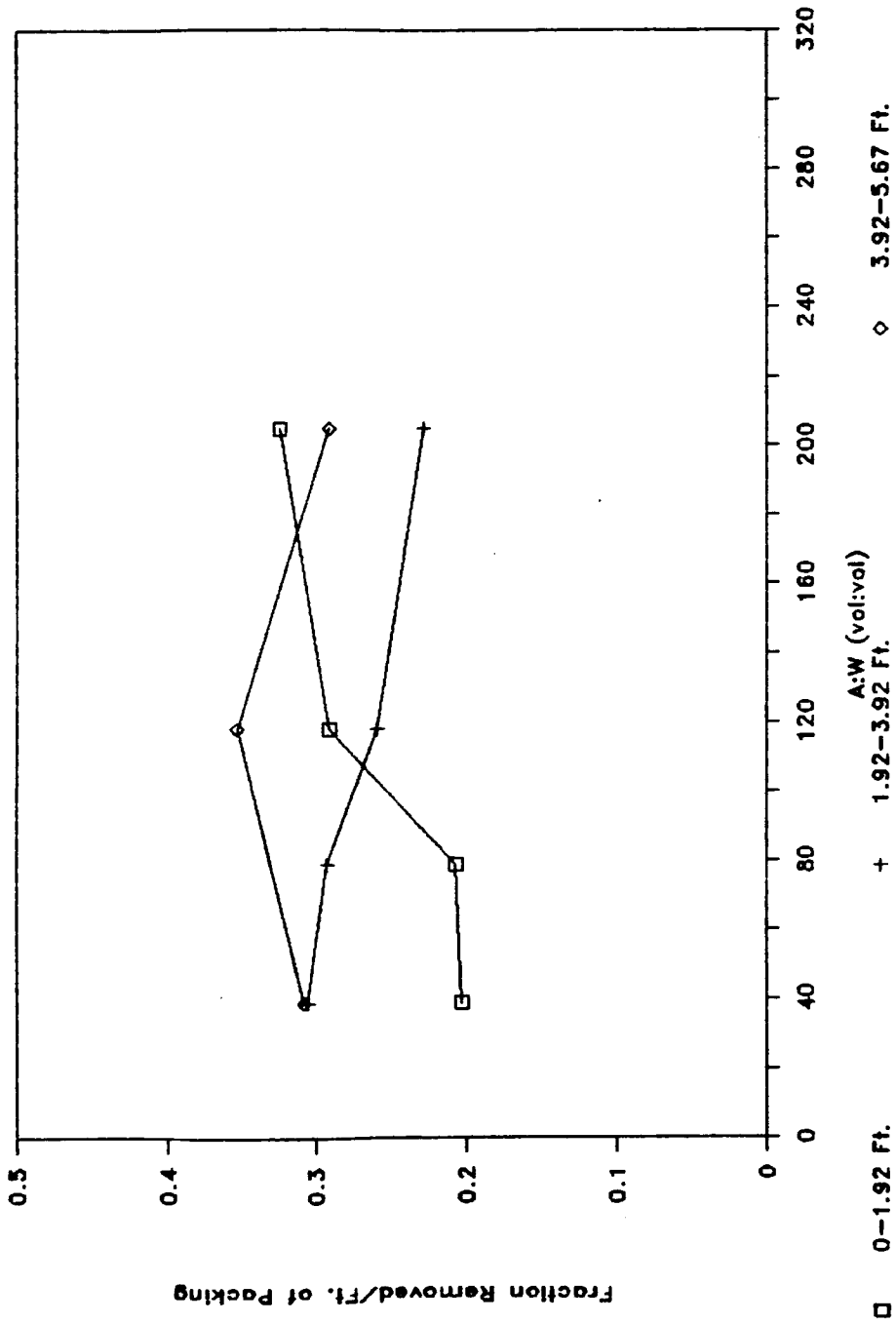


Figure 41. Experiments 10-14, Tripacks, TCE Removal per Foot Incremental Between Sample Ports vs. A:W.

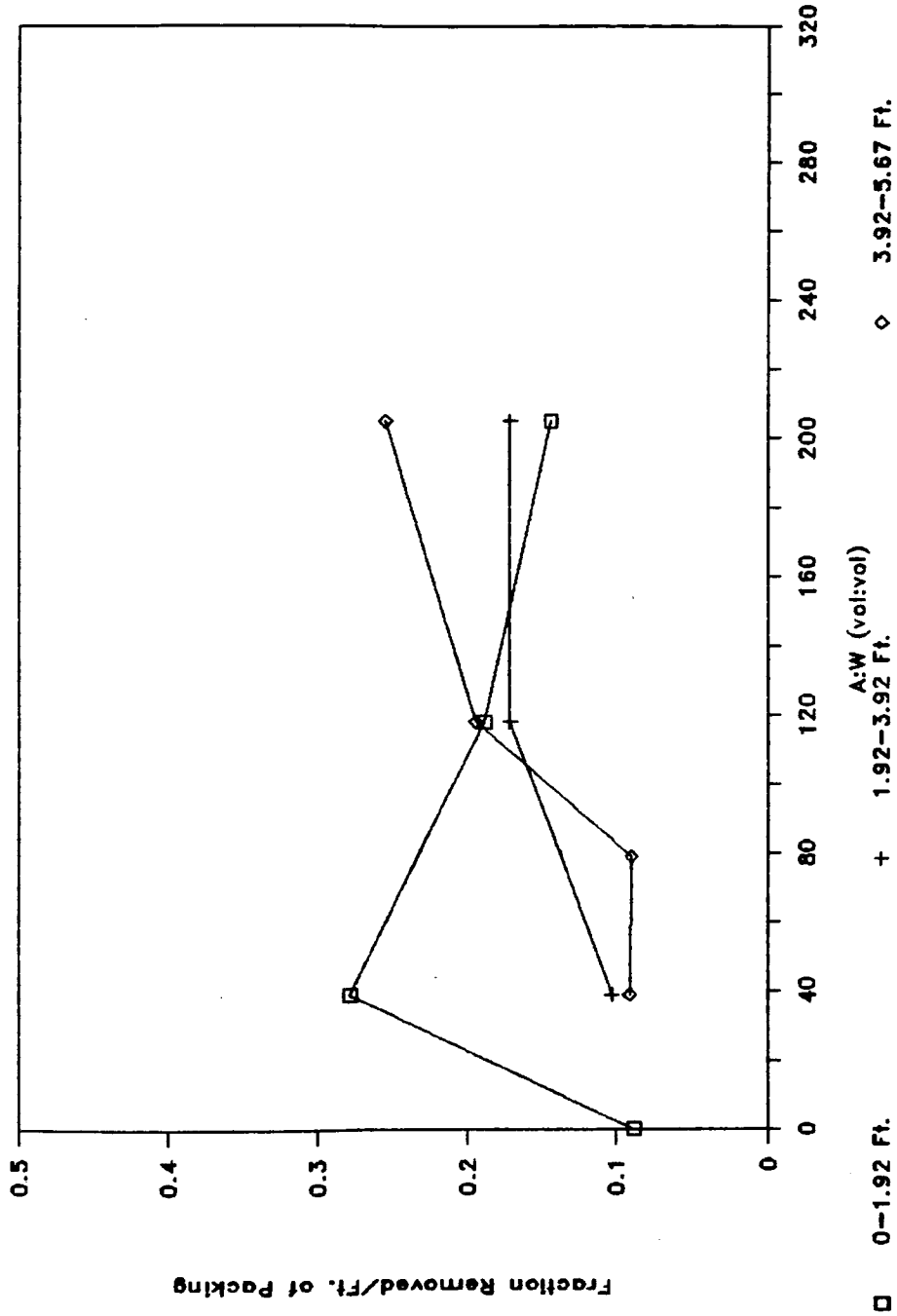


Figure 42. Experiments 10-14, Tripacks, 1,4 DCB Removal per Foot Incremental Between Sample Ports vs. A:W.

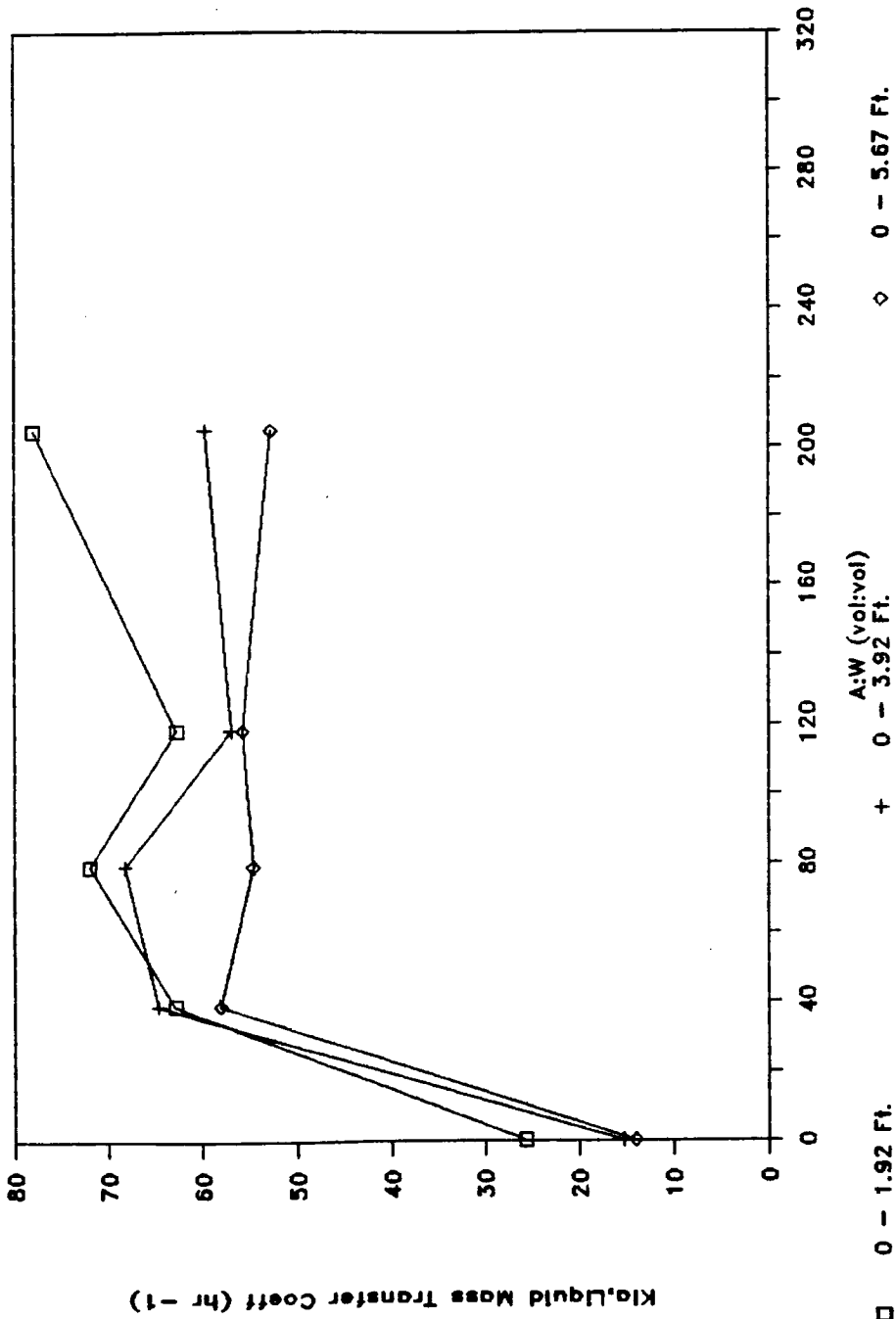


Figure 43. Experiments 10-14, Tripacks, CT, K_L a, Relative to Top of Packing vs. A:W.

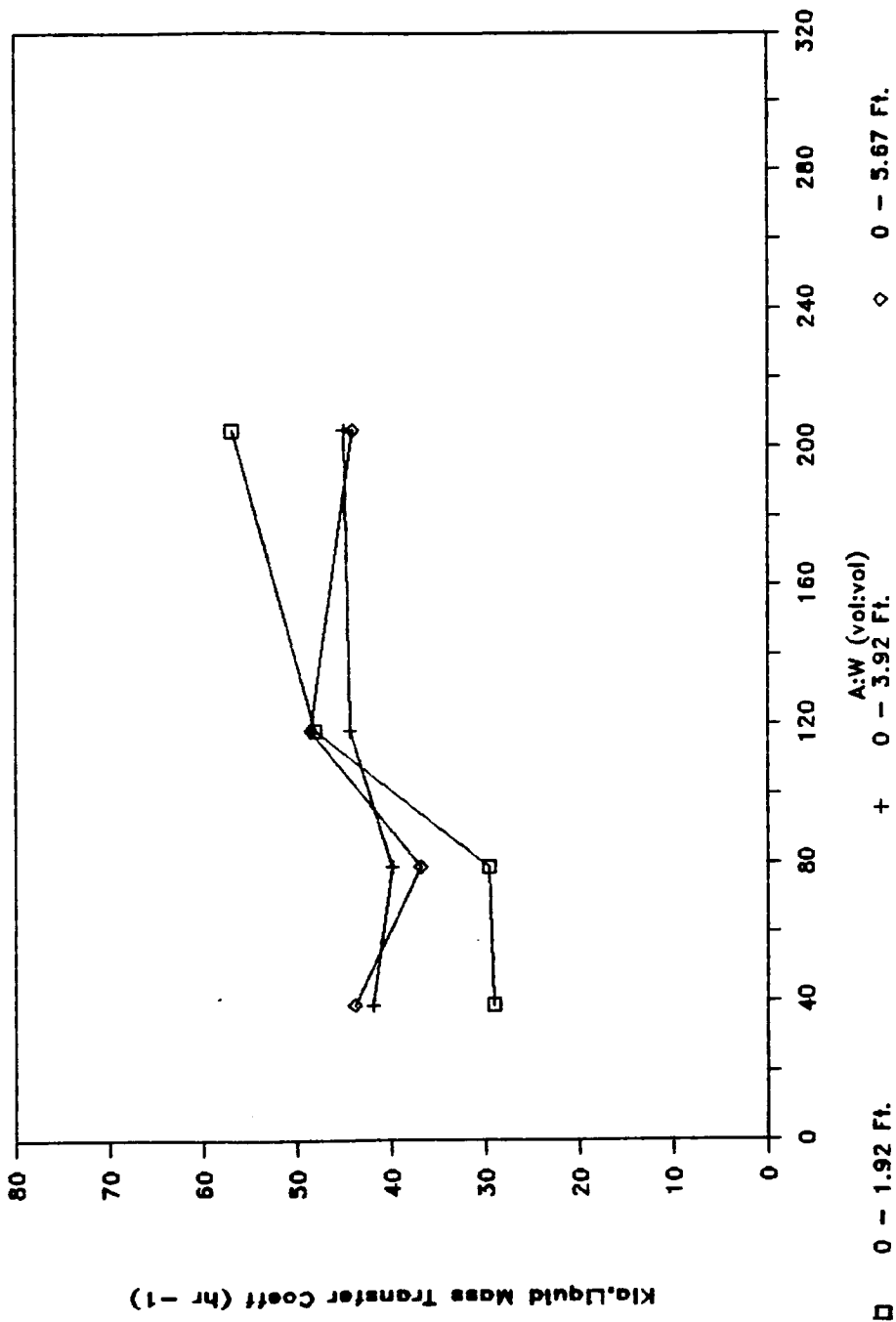


Figure 44. Experiments 10-14, Tripacks, TCE, K_{La} , Relative to Top of Packing vs. A:W.

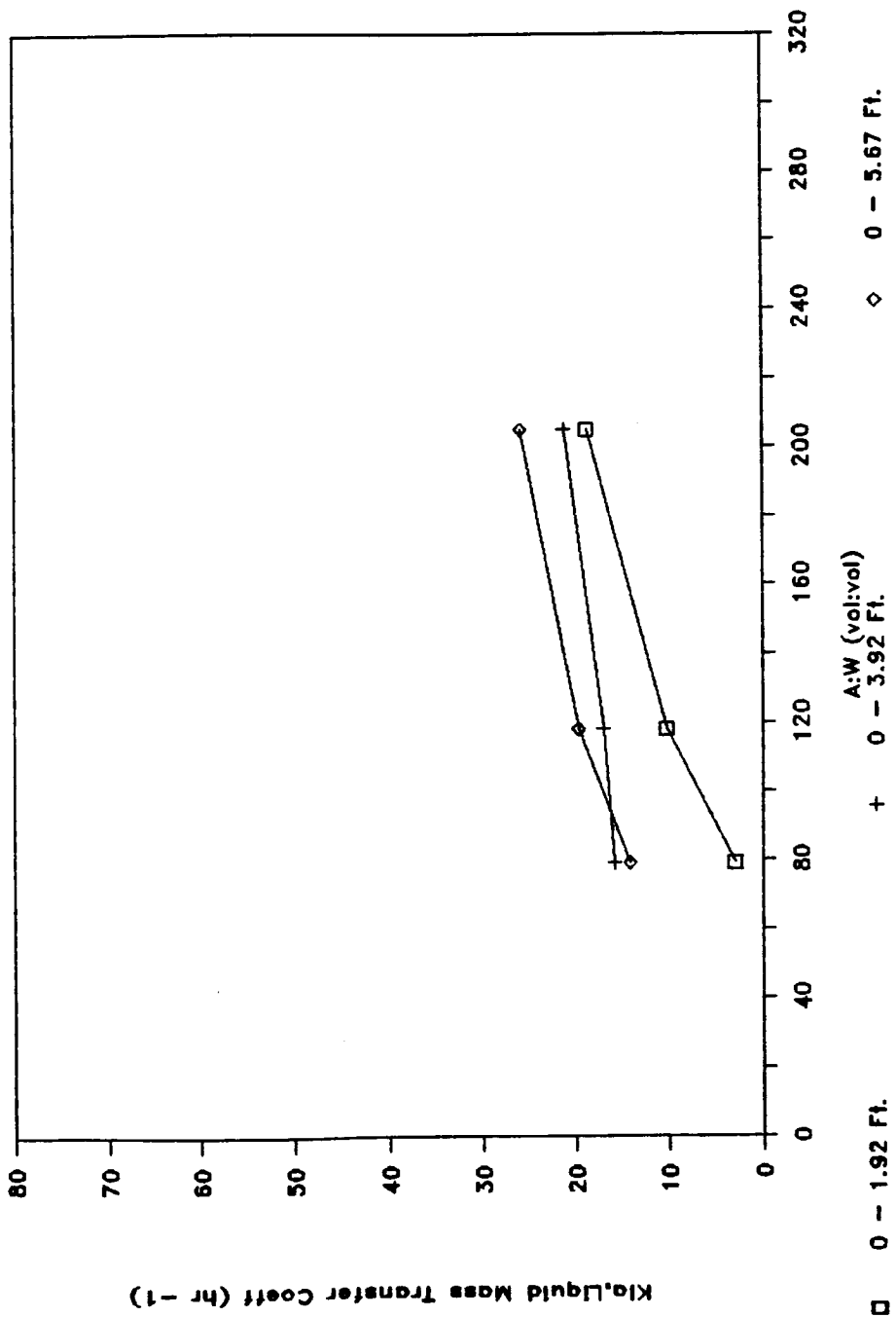


Figure 45. Experiments 10-14, Tripacks, 1,4 DCB, $K_L a$, Relative to Top of Packing vs. A:W

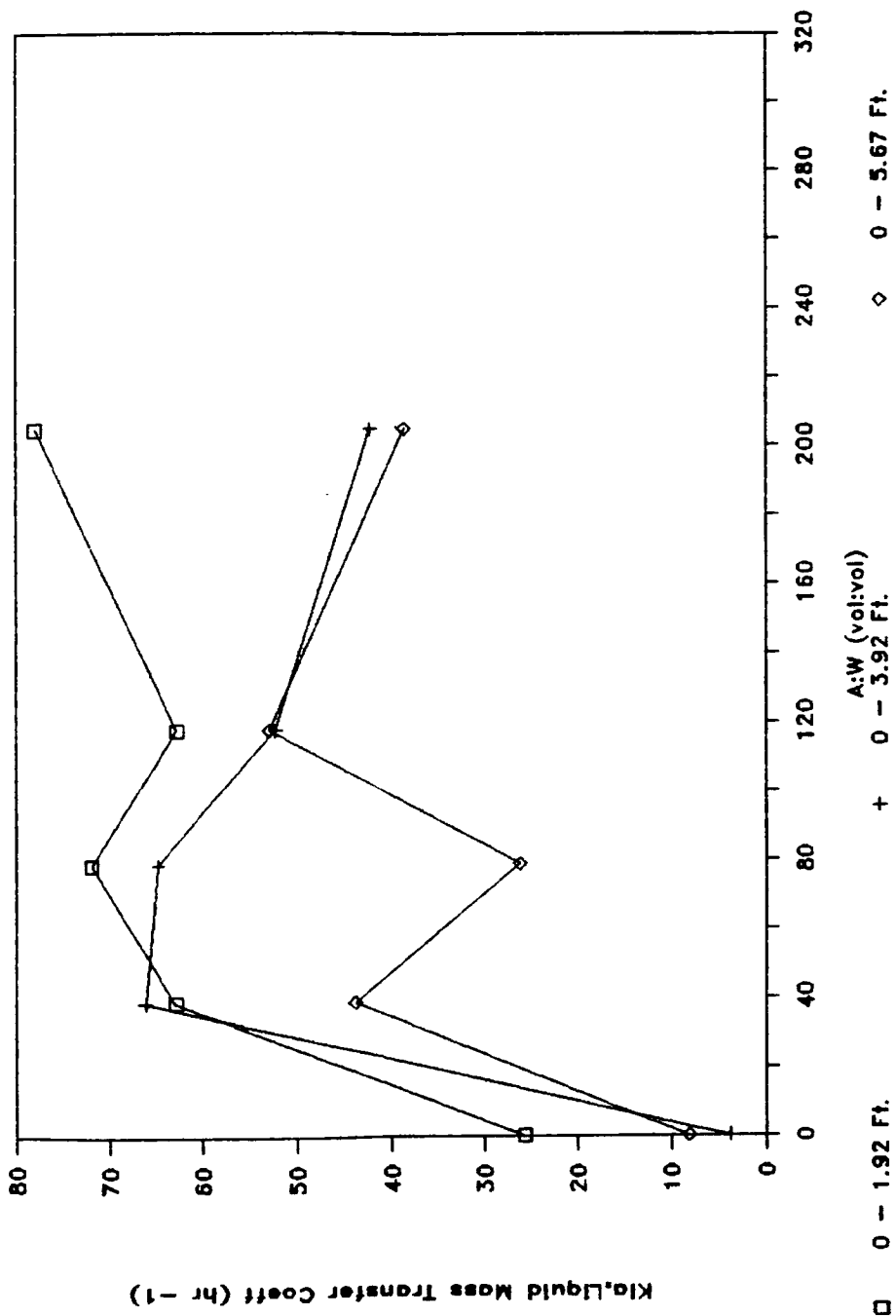


Figure 46. Experiments 10-14, Tripacks, CT, K_L a Incremental Between Sample Ports vs. A:W.

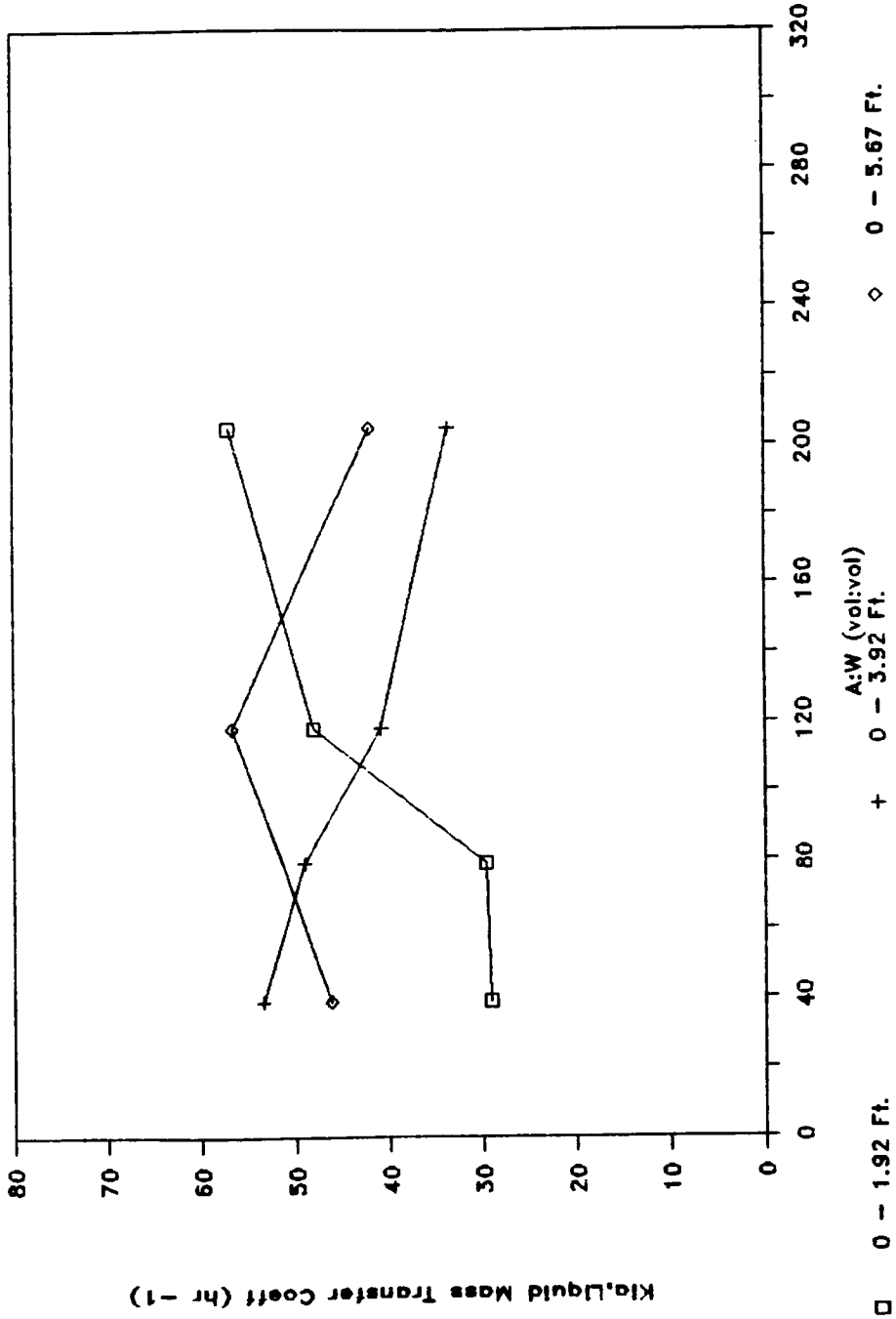


Figure 47. Experiments 10-14, Tripacks, TCE, K_L a Incremental Between Sample Ports vs. A:W.

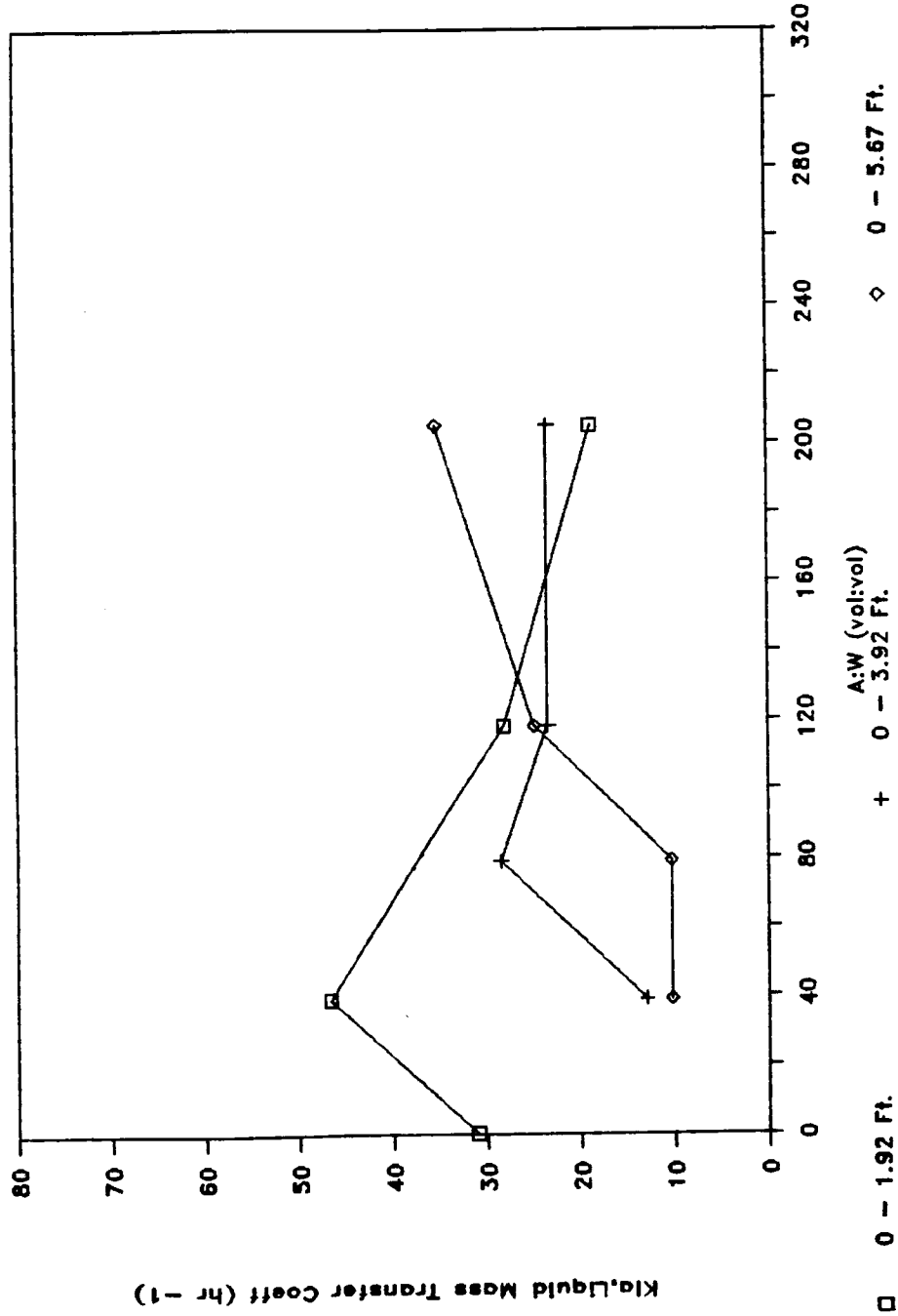


Figure 48. Experiments 10-14, Tripacks, 1,4 DCB, K_{La} Incremental Between Sample Ports vs. A:W.

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