3-24-2008

An Evaluation and Implementation Guide for Current Groundwater Mass Flux Measurement Practices

Jack G. Wheeldon III

AN EVALUATION AND IMPLEMENTATION GUIDE FOR CURRENT GROUNDWATER MASS FLUX MEASUREMENT PRACTICES

THESIS

Jack Grierson Wheeldon III, Major, USAF

AFIT/GEM/ENV/08-M23

DEPARTMENT OF THE AIR FORCE
AIR UNIVERSITY

AIR FORCE INSTITUTE OF TECHNOLOGY

Wright-Patterson Air Force Base, Ohio

AFIT/GEM/ENV/08-M23

AN EVALUATION AND IMPLEMENTATION GUIDE FOR CURRENT GROUNDWATER MASS FLUX MEASUREMENT PRACTICES

THESIS

Presented to the Faculty

Department of Systems and Engineering Management

Graduate School of Engineering and Management

Air Force Institute of Technology

Air University

Air Education and Training Command

In Partial Fulfillment of the Requirements for the

Degree of Master of Science in Engineering Management

Jack Grierson Wheeldon III,

Major, United States Air Force

March 2008

AFIT/GEM/ENV/08-M23

AN EVALUATION AND IMPLEMENTATION GUIDE FOR CURRENT GROUNDWATER MASS FLUX MEASUREMENT PRACTICES

Jack Grierson Wheeldon III, BS

Major, United States Air Force

Approved:

// Signed //	24 Mar 08
Dr. Mark N. Goltz (Chairman)	date
// Signed //	18 Mar 08
Dr. Alfred E. Thal Jr.	date
// Signed //	17 Mar 08
Major Sonia E. Leach	date

AFIT/GEM/ENV/08-M23

Abstract

Contaminant mass flux is an important parameter needed for decision making at sites with contaminated groundwater. New and potentially better methods for measuring mass flux are emerging. This study looks at the conventional transect method (TM), and the newer passive flux meter (PFM), modified integral pump test (MIPT), and tandem circulating well (TCW) methods. In order to facilitate transfer and application of these innovative technologies, it is essential that potential technology users have access to credible information that addresses technology capabilities, limitations, and costs. This study provides such information on each of the methods by reviewing implementation practices and comparing the costs of applying the methods at 16 standardized "template" sites. The results of the analysis are consolidated into a decision tree that can be used to determine which measurement method would be most effective, from cost and performance standpoints, in meeting management objectives at a given site.

The study found that, in general: (1) the point methods (i.e. the TM and PFM) were less expensive to use to characterize smaller areas of contamination while the pumping methods (the MIPT and TCW) would be more economical for larger areas, (2) the pumping methods are not capable of high resolution sampling, which may be required to characterize heterogeneous systems or to design remediations, and (3) when high resolution is required, the PFM is more economical then the TM. Finally, the study demonstrated that, arguably, test results of the newer methods indicate that their accuracy is as good as, or better than, the accuracy of the TM, the currently accepted method.

Acknowledgments

I would like to convey my gratitude to my thesis advisor, Dr. Mark N. Goltz, for guiding me through this challenging and rewarding experience. His guidance provided the vectoring for my unwieldy efforts. Without his insight, patience, and instruction this thesis would not be possible. Additionally, I would like to thank Dr. Alfred E. Thal Jr. for his teachings and lightheartedness which provided me with the tools and a voice of reason when under the stresses of graduate school. I would also like to thank Major Sonia Leach for her professional guidance and overall support.

My sincerest thanks to Drs. Michael C. Brooks and A. Lynn Wood of the United States Environmental Protection Agency for their professional assistance in this study.

This study was partially supported by (1) an interagency agreement between the Air Force Institute of Technology (AFIT) and the Ground Water and Ecosystem Restoration Division (GWERD) of the U.S. Environmental Protection Agency National Risk Management Research Laboratory, and (2) Environmental Security Technology Certification Program (ESTCP) Project CU-0318, Diagnostic Tools for Performance Evaluation of Innovative In Situ Remediation Technologies at Chlorinated Solvent-Contaminated Sites. The support of GWERD and ESTCP is gratefully acknowledged.

Most importantly, I am deeply thankful to my family for their unconditional love and support. My wife and children provided me the encouragement and inspiration to persevere when times were tough. I am also extremely grateful to my parents for their support and confidence that have shaped who I am today.

Jack Wheeldon

Table of Contents

Page

Page

Page

Page

List of Figures

Page

Page

List of Tables

Page

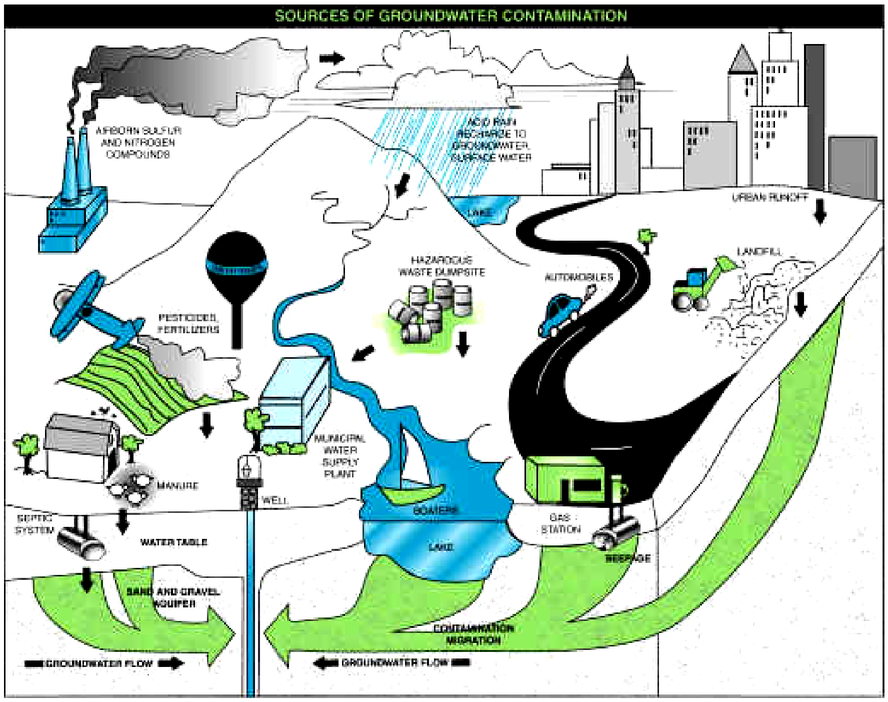
AN EVALUATION AND IMPLEMENTATION GUIDE FOR CURRENT GROUNDWATER MASS FLUX MEASUREMENT PRACTICES

Introduction

1.1 Motivation

In the United States, 46 percent of the population depends on groundwater for their drinking water supply. In fact, 83.2 billion gallons of groundwater is pumped daily from 15.9 million wells for public and private supply, irrigation, livestock, manufacturing, mining, and other purposes (NGWA, 2007). Clearly, groundwater is an important resource that can pose health and environmental risks if it is contaminated. Contamination of groundwater can result from a wide range of sources, such as landfills, neglected hazardous waste sites, leaking underground storage tanks, agricultural activities, and industrial spills (see figure 1.1). Protecting groundwater from contamination is both technologically and economically challenging. As an example of the immensity of the problem, the US Environmental Protection Agency reports more than 460 thousand confirmed releases of petroleum and hazardous materials from underground storage tanks (USTs) as of March 2007. Roughly 357 thousand of these releases have been cleaned, leaving over 100 thousand yet to be remediated, in addition to any new releases that are discovered (USEPA, 2008).

Figure 1.1 Groundwater contamination sources (Groundwater Foundation, 2007)



Clean-up of the releases costs millions of dollars every year, which is paid by responsible parties or covered by a $0.001 tax on every gallon of fuel sold (USGAO, 2007). With thousands of contaminated sites and limited funding, it is most advantageous to clean those sites that pose the greatest threat to human and environmental receptors first (Einarson and Mackay, 2001). However, identifying the sites which pose the greatest threat requires site characterization.

Critical to the site characterization effort is the ability to accurately measure the contaminant concentrations and movement (Kao and Wang, 2001; Einarson and Mackay, 2001). Einarson and Mackay (2001) suggest that contaminant flux rather than concentration is a more effective measure of risk. Basu et al. (2006) report a growing consensus among researchers and regulatory agencies that contaminant flux should be used as an alternate performance metric in site assessment and remediation design. Contaminant mass flux is defined as the total mass of contaminant passing a unit area of control plane that is perpendicular to the mean groundwater flow direction per unit time (Basu et al., 2006). Mass flux is a key parameter needed to characterize contaminant movement; it provides data that are essential to prioritizing contaminated sites for remediation (Einarson and Mackay, 2001; USEPA, 2007). Contaminant mass flux measurements integrated over a source area will produce estimates of the source strength and generate critical data for optimizing design and assessing performance of source remediation technologies (Annable et al., 2005)

Recent studies (Einarson and Mackay, 2001; USEPA, 2007; NRC, 2004; Basu et al., 2006) have shown that contaminant mass flux is an important parameter to quantify in order to assist remediation decision making at sites with contaminated groundwater. Mass flux measurements may be used to 1) prioritize contaminated groundwater sites for remediation, 2) evaluate the effectiveness of source removal technologies or natural attenuation processes, and 3) define a source term for groundwater contaminant transport modeling, which can be used as a tool to achieve the previous two objectives and to assist with remediation technology design (Goltz et al., 2007a).

Current methods for measuring contaminant mass flux include the transect method (TM) using multi-level sampling (MLS), the integral pump test (IPT, formerly known as the integral groundwater investigation method (IGIM)), and passive flux meters (PFMs). In addition to the above-mentioned methods, new groundwater contaminant mass flux measurement methods are in development. The newer methods include modified integral pump tests (MIPTs) and tandem circulating wells (TCWs). The introduction of new flux measurement methods offers the potential of increased accuracy and decreased cost and time. The development of the new methods has also created a knowledge gap between current field practices and the progress made in academic research. In order to facilitate transfer and application of an innovative technology, it is essential that potential technology users have access to credible information that addresses the capabilities, limitations, and projected expenses of the new technology (NRC, 1997).

1.2 Background

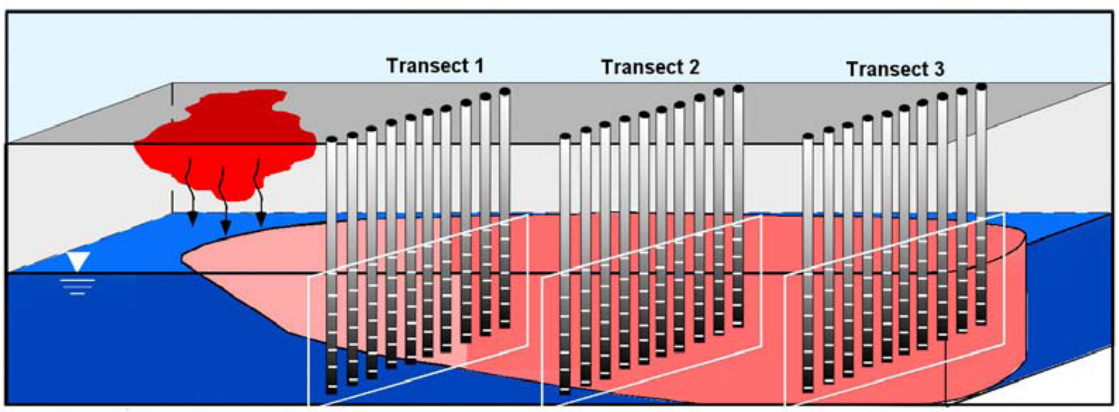
The innovative flux measurement methods may be categorized by how they measure flux. Some methods are so-called point methods, where the measurements are taken at particular locations at particular instants in time. Other methods employ time-averaging, where samples are taken at particular locations, but averaged over defined time intervals. Still other methods employ a volume-averaging approach, where pumping is used to estimate flux averaged over a large subsurface volume.

Depending on circumstances, each method has advantages and disadvantages in comparison to the other methods. A number of studies have been conducted that have looked at the performance of the new flux measurement technologies in both the laboratory and field (Hatfield et al., 2004; ESTCP, 2007b; Brooks et al., 2007; Goltz et al., 2007a; 2007b; ESTCP, 2006; SERDP, 2004). Costs of applying the new technologies have been addressed to a smaller degree. Not addressed at all is a side-by-side comparison of the different methods under varying hydrogeologic conditions. In addition, past studies have not looked at how the choice of measurement methodology is affected by the purpose of the measurement. As noted earlier, flux measurement is essentially done for two reasons: (1) assess risk in order to develop cleanup priorities or evaluate the efficacy of remediation and (2) design of a remediation technology. In the sections below, we briefly describe each mass flux measurement method.

1.2.1 Transect method (TM)

The conventional transect method uses multiple multilevel sampling (MLS) wells installed along control planes orthogonal to the mean groundwater flow direction. For this study, the term TM will refer to the transect method using the MLS. Groundwater samples are taken at various depths at each of the sampling points and the contaminant concentration [M/L^3] is measured. The general configuration of the transect method is shown in figure 1.2. Note that the transect method is essentially a point method, which estimates the flux at the particular location and instant in time where and when the groundwater sample is obtained. If the aquifer and contaminant distribution is heterogeneous, as is typical, a higher resolution of sampling may be needed to adequately characterize the flux.

Figure 1.2 Basic configuration of the transect method with multilevel sampling (API, 2003)



The TM requires a separate test to estimate the hydraulic conductivity (K) [L/T] of the aquifer, as well as the regional hydraulic gradient (i) [L/L]. By Darcy's Law, the product of the hydraulic conductivity and the hydraulic gradient is the Darcy velocity (also referred to as specific discharge or groundwater flux) (q_0) [L^3/L^2T]. To calculate the contaminant mass flux (J) [M/L^2T], the following equation can then be used:

$$J = q_0 C = -KiC \tag{1.1}$$

Slug and pump tests may be used to estimate K, while piezometers can be used to determine the hydraulic gradient i. Another method of determining K is through the use of a borehole dilution test (BDT). In this study, the method used in conjunction with the TM to determine groundwater flux is the BDT test. The BDT test is a method designed to estimate a time-averaged value of q_0 at a particular point. A tracer is injected into a well and the rate at which the tracer concentration is reduced due to dilution of the tracer by groundwater flowing through the well is monitored (Pitrak et al., 2007). A plot of tracer concentration versus time can be interpreted to produce an estimate of the Darcy velocity of the flowing groundwater.

1.2.2 Passive flux meter (PFM)

The PFM measures the time-averaged Darcy velocity and contaminant flux at a point in space. This innovative method, developed by University of Florida and Purdue University researchers (Klammler et al., 2007; Campbell et al., 2006; Annable et al., 2005; Hatfield et al., 2004; Lee et al., 2006) uses a sorbent (e.g., granular activated carbon or GAC) impregnated with known masses of so-called resident tracers. The sorbent is then packed in a cylindrical unit and placed down the wells of a control plane in an aquifer to be characterized. The control plane transect used for this method is similar to the TM used when applying the MLSs (figure 1-2) except that the packed sorbent units are placed downwell rather than using the MLSs to obtain samples at individual depths. As flowing groundwater passes by the PFM, the groundwater contaminant partitions into the sorbent while the resident tracer in the sorbent is released into the groundwater. After a specified time, the sorbent is removed from the well and analyzed to determine the mass of contaminant captured and the mass of tracer released. The amount of tracer released is used to estimate the groundwater Darcy velocity, averaged over the time the PFM was in the well. Similarly, the mass of contaminant captured is used, in conjunction with the Darcy velocity, to estimate the time-averaged contaminant concentration and contaminant flux (Hatfield et al., 2005).

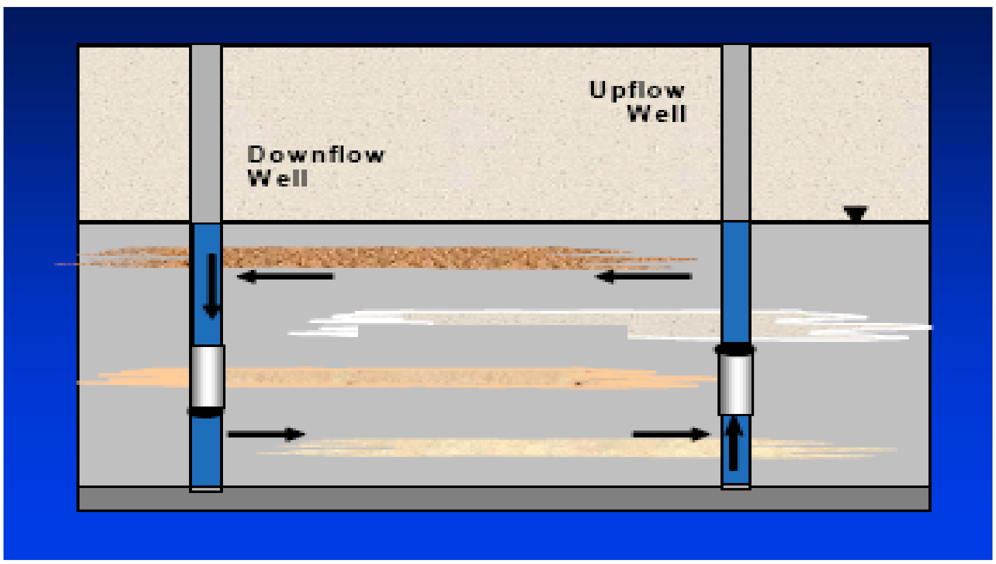
1.2.3 Modified integral pump test (MIPT)

The modified integral pump test is an innovative volume-averaging method for estimating Darcy velocity and contaminant flux. The MIPT is a variation of an earlier method, known as the IPT (Brooks et al., 2007; Bayer-Raich et al., 2006; Ptak et al., 2003). The IPT, formerly called the integral groundwater investigation method (IGIM), uses concentration-time series information from a series of pumping wells located across a control plane, along with a separately obtained estimate of Darcy velocity, to calculate contaminant mass flux (Bockelmann et al., 2001; Ptak et al., 2003; Bauer et al., 2004). The MIPT differs from the IPT in that the MIPT directly obtains volume-averaged estimates of Darcy velocity and concentration by pumping multiple wells and monitoring the hydraulic head at nearby piezometers (USEPA, 2007; Brooks et al., 2007). The differences in head that are measured as pumping is increased in steps can be used to determine an estimate of the Darcy velocity. This, in conjunction with sampling the wells during operation to determine the concentration of contaminant, will provide an estimate of contaminant flux (Brooks et al., 2007).

1.2.4 Tandem Circulating Wells (TCWs)

Tandem Circulating Wells measure hydraulic conductivity by developing a three dimensional circulation pattern between a pair of dual-screened wells (figure 1.3). The technique is an adaptation from earlier work, where TCWs were used to effect *in situ* bioremediation (McCarty et al., 1998). One TCW pumps groundwater upwards, extracting water through a well screen at a low elevation and injecting it through a well screen at a higher elevation while the second TCW, installed nearby, pumps water in the opposite direction.

Figure 1.3 Flow paths induced by TCW operation



Measurements of hydraulic heads at the two wells for various pumping rates can be used to obtain estimates of hydraulic conductivity. With the pumps turned off, the regional hydraulic gradient (*i*) can be obtained, and the Darcy velocity (q_0) estimated by applying Darcy's Law (Goltz et al., 2007a). With the Darcy velocity known, groundwater samples are taken to determine average contaminant concentrations, which can then be multiplied by the Darcy velocity to estimate contaminant flux (per equation 1.1). Goltz et al. (2007a) also describe how a tracer test can be conducted using TCWs in order to obtain an estimate of hydraulic conductivity (which can subsequently be used to determine Darcy velocity and contaminant flux as discussed above).

1.3 Research Objectives

In order to facilitate transfer and application of an innovative technology, it is essential that potential technology users have access to credible information that addresses the capabilities, limitations, and costs of the technology. The purpose of this work is to critically review the methods currently available to measure groundwater contaminant flux, and provide guidelines for the implementation and use of those methods in the field.

1.4 Methodology

This study will consist of a background investigation of the aforementioned flux measurement methods from both published and unpublished literature. When applicable, interviews of flux measurement researchers will also be used to develop the technology review. The review will include the following:

1. Laboratory, field, and commercial applications of conventional and innovative flux measurement technologies. Included in the review will be:
    a. A description of the measurement methods
    b. Technology implementation details and costs
    c. Quantification of measurements errors (accuracy, performance, etc.)
2. Information required by stakeholders to evaluate the applicability of a technology to facilitate decision making

3. Cost estimation of subsurface investigation methods, considering methods for extrapolating the results of small- and pilot-scale studies to predict full-scale costs.

Based on the literature review, we will prepare a critical analysis of the advantages and disadvantages, costs, and measurement errors of the different methods. The analysis will include a discussion of how hydrogeologic conditions and management objectives support application of one method over another.

Finally, considering the requirements of decision makers that were elicited from the literature review, we construct a “user-friendly” tool that can be used to facilitate information transfer to potential technology users. The tool will help the technology users decide which flux measurement method is most appropriate for given site conditions.

1.5 Study Limitations

1. Field testing
    a. To date no attempt has been made to implement the TCW in the field
    b. Modified IPT has not undergone validation in the field with known fluxes for comparison
2. Cost data are limited (especially for the newer methods)

II. Literature Review

2.1 Introduction

This chapter reviews the current literature on the different flux measurement methods briefly described in Chapter 1. We will start with a description and basic definition of flux along with the elements required for its calculation in section 2.2. Knowing these elements will allow us to explain the differences between the methods. Next, in section 2.3, we will systematically describe each method, to include implementation information and a description of each method's advantages and disadvantages. In addition, laboratory and field tests, cost, and regulatory issues will be reviewed for each method. In sections 2.4 and 2.5 comparisons of method performance and costs, respectively, which have appeared in the literature, will be examined.

2.2 Mass Flux

Mass flux is the rate at which a dissolved contaminant passes through a cross-sectional area perpendicular to the direction of flow (Basu et al., 2006). Flux therefore has units of mass per area per time. Mass flux (J) can be calculated as the product of contaminant concentration (C) and groundwater flux (or Darcy velocity) (q_0):

$$J = q_0 C \qquad (2.1)$$

Since the Darcy velocity is, in accordance with Darcy's Law, equal to the product of the hydraulic conductivity of the porous media (K) and the hydraulic gradient (i), we may also calculate mass flux as the product of concentration, hydraulic conductivity, and hydraulic gradient:

$$J = -KiC \qquad (2.2)$$

Mass flux is sometimes confused with mass discharge, which has units of mass per time. In fact, mass discharge is just the mass flux multiplied by the cross-sectional area through which the dissolved contaminant plume is moving. Both mass discharge and mass flux are recognized as important parameters to characterize contaminated groundwater and quantify the efficacy of subsurface contaminant remediations (Basu et al., 2006).

Equations 2.1 and 2.2 offer different approaches to estimating contaminant mass flux. Equation 2.1 uses q_0 directly while in equation 2.2 K, i, and C are individually determined. These parameters (q_0, K, i, and C) can either be measured at a point or averaged over space and/or time. A point measurement may not be representative of the larger cross sectional area being considered. To decrease uncertainty, more points can be used in multiple wells at varying depths (e.g., by using an MLS) to determine concentration and flux distribution across an area. Though costly, this approach can provide detailed measurements of concentration and flux at many points in a contaminant plume. A second approach is to obtain time- or space-averaged concentration and flux measurements. This approach, in general, is less costly than a point approach, particularly when concentrations and flux vary spatially (due, perhaps, to heterogeneity of hydraulic conductivity or contaminant distribution) or temporally (for example, due to hydrological fluctuations). The disadvantage of averaging is that the detailed spatial and temporal definition of point measurements is lost.

2.3 Flux Measurement Methods

2.3.1 Transect method (TM)

The TM is a point approach to measuring concentration. From here on, TM will refer to the transect method using the MLS unless otherwise stated. Additional tests are required to measure K and i to calculate mass flux. Constructing the transect involves placing a control plane perpendicular to the groundwater flow direction. Within the control plane, either multilevel samplers or single-screen wells can be employed. To accomplish a total mass flux calculation, it is imperative that the entire width and depth of the plume is captured by the control plane. A transect can be placed at any distance down gradient of a contamination source. Transects are normally placed to evaluate source strength, natural attenuation, or compliance at a control plane. Figure 1.2 is an example of multiple transects at different locations down gradient of a source of contamination.

The application of the transect method to determine contaminant mass flux and discharge is relatively straightforward. Once the concentration at each of the n points across the control plane is determined, the mass discharge, M_d [M/T], at each point can be calculated by applying equation 2.3 (API, 2003):

$$M_d = \sum_{j=1}^{n} C_j q_j A_j \qquad (2.3)$$

where C_j [M/L^3] is the concentration at individual measurement point j and q_j [L/T] is the specific discharge (Darcy velocity) through the flow area associated with measurement point j (A_j). The A_j for each measurement point j can be estimated by constructing Theissen polygons about the points. Theissen polygons are constructed by connecting points that are located halfway between measurement point j and adjacent measurement points. For sampling point j, the area of the Theissen polygon around j defines A_j (Bockelmann et al., 2003).

The Darcy velocity at measurement point j (q_j) can be determined directly or calculated by Darcy's Law:

$$q_j = -Ki \qquad (2.4)$$

where K is the hydraulic conductivity (L/T) and i is the hydraulic gradient (L/L). In section 2.3.1.1 below, we discuss how q_j, K, and i may be determined using the BDT, slug tests, and pump tests.

To convert the mass discharge to mass flux, simply divide the total mass discharge from equation 2.3 by the cross-sectional area of the plume across the control plane (A).

$$M_f = \frac{M_d}{A} = \frac{M_d}{\sum_{j=1}^{n} A_j} \tag{2.5}$$

Equation 2.5 gives the average contaminant flux of the plume across the control plane.

2.3.1.1 Calculating Darcy velocity

The transect method offers the ability to measure concentrations at various points along the transect. To complete the calculations for flux requires a value of Darcy velocity, either measured directly or determined by Darcy's Law (equation 2.4). The BDT provides a direct estimate of q_0 while the slug and pump test provide values for K. The hydraulic gradient is obtained from piezometers. The difference in head measured at piezometers along the direction of flow is divided by the distance between them to obtain the hydraulic gradient.

2.3.1.1.1The slug test

The slug test is a popular point method that is used to determine hydraulic conductivity in both soil and rock. Slug tests are implemented by removing, adding, or displacing a known quantity of water within a well and monitoring the changes in water level with respect to time (Nielsen, 2006). The time it takes for the water level to reach its initial level is representative of the horizontal hydraulic conductivity of the soil around the screened portion of the well. Data loggers can often be used to facilitate data collection under high hydraulic conductivity conditions where recovery takes place quickly (Nielsen, 2006). The simplest interpretation of recovery data for hydraulic conductivity is that of Hvorslev (1951). Additionally, the Bouwer and Rice (1976) method is also commonly used to calculate the hydraulic conductivity. For details on conducting the slug test, refer to Nielsen (2006) or Fetter (1994).

2.3.1.1.2 The pump test

The pump test is the most popular method for investigating hydraulic properties of water-bearing geological material (Nielsen, 2006). The test is volume-averaged and is typically conducted using a central pumping well and one or more nearby observation wells. The drawdown in the observation well(s) is monitored as the central well is pumped at either a constant or variable rate. The position and configuration of the wells will depend on the aquifer properties. Care should be taken to avoid proximity of the pumping well to boundaries (e.g. recharging river or impervious zones). This will allow drawdown measurements to be taken without the influence of the boundary (Nielsen, 2006). Before the pump test is started, initial head measurements are taken at the observations well(s). These measurements will provide a comparison for the drawdown measurements once pumping begins. Once pumping starts, the drawdown measurements are taken and plotted for each well on a time-verses-drawdown semi logarithmic chart. A horizontal line will indicate steady-state or near steady-state. Analysis of the data depends on the type of aquifer and conditions. At steady-state, graphical and analytical solutions can be applied to the well drawdown data to determine the hydraulic properties. Details for determining hydraulic conductivity from pump tests for different aquifer conditions may be found in Nielsen (2006).

2.3.1.1.3 Borehole dilution test (BDT)

The borehole dilution test is a time-averaged point method that is used to determine Darcy velocity directly. The BDT is conducted by injecting a known amount of tracer into a well and monitoring concentration levels as it dissipates due to the flow of groundwater through the well screen (Lile et al., 1997). Salts, such as potassium bromide, are commonly used as tracers. Concentrations of these salt tracers are easily measured by monitoring changes in electrical conductivity or by using a specific ion electrode (USEPA, 2007). The rate at which the tracer dissipates is a measure of the groundwater Darcy velocity. However, because the hydraulic conductivity of the well and borehole are different than the conductivity of the aquifer (typically the well and borehole conductivity is higher), flow lines will converge at the borehole. The impact of this must be accounted for when calculating the Darcy velocity from BDT data. Unfortunately, the degree of convergence (α) is difficult to quantify, resulting in either an over or underestimate of Darcy velocity (Bernstein et al., 2007). Drost et al. (1968) and Bidaux and Tsang (1991) have presented methods for approximating the convergence factor, α. Taking into account the convergence factor α, q_0 can be solved for directly by applying equation 2.6:

$$q_0 = -\frac{V}{\alpha A t} \ln\left[\frac{c}{c_o}\right] \qquad (2.6)$$

where $c(t)$ and c_0 [M/L^3] are the tracer concentrations within the well at time t and initially, V [L^3] is the specific well volume, and A [L^2] is the projected area of the well (Bernstein et al., 2007). Darcy velocity can be determined by a linear plot of log concentration vs. time.

Note that the BDT method measures the sum of advective flux and diffusive flux. Advective flux is due to the flow of the groundwater, and it is the measurement of interest. Diffusive flux is due to Fickian diffusion of the tracer, which results from the concentration difference between the tracer in the well and the tracer outside the well. In most cases, the advective flux is much larger than the diffusive flux and the diffusion effect can be ignored. However, in low conductivity and highly porous formations, where the advective flux is low, diffusion may have an important effect on the observed tracer dilution. Not accounting for diffusion under these low flow conditions may result in false readings of high groundwater velocities (Bernstein et al., 2007).

2.3.1.2 TM costs

The costs associated with using the TM/BDT at a hypothetical site were evaluated (ESTCP, 2007b). Using a transect of 10 wells at a depth of 10 feet and a vertical sampling resolution of one foot (100 points total), the study found that costs of implementing the TM/BDT method were $430 per linear foot of well. Details of the cost breakdown are in section 2.5. It is important to note that the cost analysis included conducting a BDT for each of the 100 points in order to quantify Darcy velocity. This amounted to a significant fraction of the overall cost ($160 of the $430/lf). Of course, these costs could perhaps be reduced if slug or pump tests were used instead of the BDT, or if the BDT was used at a lower resolution.

A cost analysis developed by Kim (2005) for a template site in a confined sand aquifer with a 200 m wide and 10 m deep plume, concluded that overall costs for implementing the TM using the pump test would be $157,000. This cost included the installation of 9 wells, 18 contaminant sample analyses, and $2,000 for conducting the slug test.

2.3.1.3 TM regulatory concerns

There are no regulatory issues associated with installing MLSs to implement the TM. However, use of BDTs or pump tests bring on regulatory concerns. The BDT typically involves injection of a salt tracer into an aquifer. This may be of concern as bromide salts may lead to the formation of bromates under certain conditions (Goltz et al., 1998). Recent studies have investigated the use of other tracers (e.g., the food color "Brilliant Blue") that could be used to avoid the concerns of using salt tracers (Pitrak et al., 2007). The pumping test will involve the extraction of potentially contaminated groundwater which will require proper handling and treatment.

2.3.1.4 TM Advantages and Limitations

The main advantages of the transect method are its flexibility and simplicity. Flexibility comes from the ability to increase or decrease the sampling resolution with little difficulty. Being able to decrease the number of sample locations offers cost savings while the ability to increase the number of locations allows for increased resolution or expansion of the transect width. The spacing of monitoring points is a critical aspect of the MLS method (Guilbeault, 2005; Kubert and Finkel, 2006). An increase in spatial resolution may be required for effective plume characterization since it has been noted that upwards of 75% of mass crossing the control plane can pass through only 10% of the transect area (Guilbeault et al., 2005). Expansion of the transect may be needed if it is found that more sampling wells are required to delineate the plume boundaries (Kim, 2005). Another advantage of the method is its simplicity. The method is well-understood and has been applied at numerous sites. Thus, finding personnel to implement the method is relatively easy. Regulators are familiar with the method and readily accept its results. When used in conjunction with the BDT or the slug test, the TM does not require extraction and treatment of large volumes of groundwater.

The limitations of the transect method are mainly due to the fact that the concentration measurements that are obtained using the TM are point measurements in space and time. Thus, temporal variability (e.g., weather events) may result in an over or under estimation of flux (Goltz et al., 2007b). Spatial heterogeneity can also cause difficulties in acquiring accurate and representative measurements. Heterogeneity of the porous media or the contaminant distribution will require a decrease in the spacing of the sampling points and therefore an increase in the number of points required (Kao and Wang, 2001; Guilbeault et al., 2005; Kubert and Finkel, 2006). Kubert and Finkel (2006) propose that the TM not be used where heterogeneity requires such a high resolution that use of an integrated pumping method would be a more cost-effective approach. We will look at this more closely in Chapter 4.

Other advantages and limitations of the TM are associated with the use of pump, slug, or borehole dilution tests to determine the Darcy velocity. The BDT can be conducted under a wide variety of conditions. The technique has been refined for use in geologies varying from unconsolidated to fractured consolidated material (Pitrak et al., 2007; Bernstein et al., 2007; Wilson et al., 2001; Lile et al., 1997). Depending upon site heterogeneity, the method may be applied at every MLS sampling location or at a lower spatial resolution. The accuracy of the BDT estimate of Darcy velocity is dependent upon the degree to which the borehole and well affect the groundwater flow field, as quantified by the convergence factor, α. Additionally, studies show that for low conductivity and high porosity media, diffusive flux of the tracer can be significant and must be accounted for. Ignoring tracer diffusion will result in the over estimation of groundwater flux (Bernstein et al. 2007; Arnon et al., 2005).

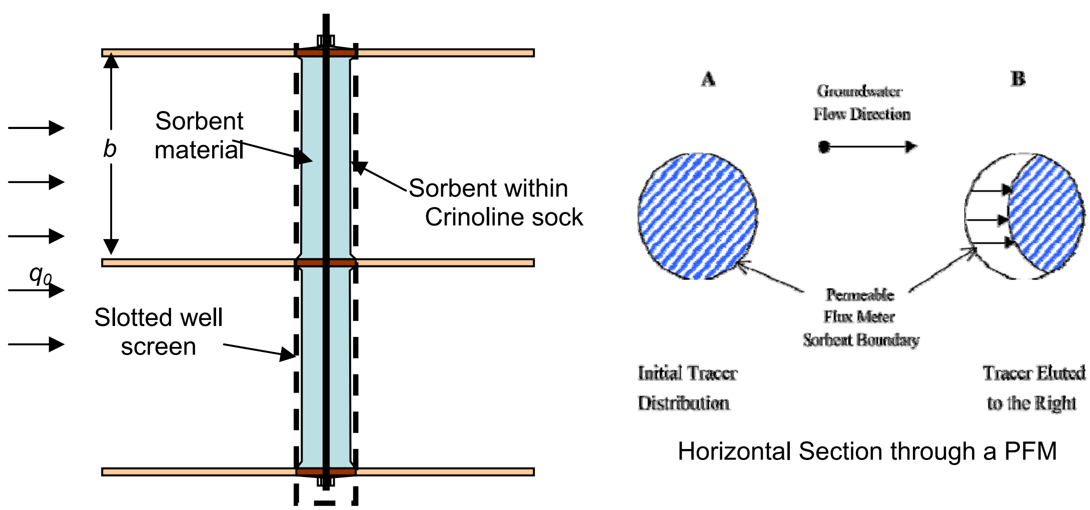
The slug test is also a common method for estimating hydraulic conductivity. The test is basically a point method, though the conductivity measurement is averaged over the relatively small volume of water interrogated during the test. One of the advantages of the slug test is that it is a short-term test. The test duration can be from less than a minute to several hours. A shortcoming is that for high conductivity formations, drawdown may be so fast that measurements would have to be recorded using electronic data loggers, and accuracy may be degraded Additionally, for high conductivity formations, well screen head loss may have a significant impact on the water-level measurements (Nielsen, 2006).

The pump test is the most common method for estimating conductivity, and it is well-understood and accepted as a good way to make conductivity measurements. The test can run from hours to days depending on conditions (Nielsen, 2006). In addition to the time it takes to run a pump test, the test requires extraction of potentially contaminated groundwater that will require treatment and disposal. This incurs additional costs.

2.3.2 Passive flux meters (PFMs)

A more recently developed device for measuring flux is the *in situ* passive flux meter. The PFM is a point method which time-averages the contaminant concentration and Darcy velocity. This method, like the transect method, requires establishing a control plane that intercepts the contaminant plume (Hatfield et al., 2005). In contrast to the TM, which must be supplemented by a pump test, slug test, BDT, or some other method of estimating Darcy velocity, the PFM method simultaneously measures contaminant concentration and Darcy velocity and therefore does not require a separate velocity measurement. The PFM is essentially a self-contained permeable unit properly sized to fit in a screened well or borehole (Hatfield et al., 2004). The unit is composed of a sorbent pre-equilibrated with a resident tracer and packed into a crinoline sock (figure 2-3). The sorbent can be a mixed medium of hydrophobic or hydrophilic permeable material which retains dissolved organic/inorganic contaminants in the groundwater (Hatfield et al., 2004). The PFM dimensions are determined by the well or borehole it is to be installed in (Hatfield et al., 2004; Annable et al., 2005). During construction of the PFM unit, the sorbent is weighed and sampled for initial concentrations of sorbed resident tracers. The PFM is then inserted into the well and is exposed to the groundwater flow for a period of time ranging from days to weeks. The duration of exposure in the well represents the time the flow and concentration is averaged.

Figure 2.1 Cross section showing PFM installation (after Hatfield et al., 2004)



While the PFM is in the well, the resident tracer desorbs into the flowing groundwater while the dissolved contaminant in the groundwater sorbs to the PFM sorbent. Similar to the BDT method, the extent of desorption of the resident tracer may be used to determine the Darcy velocity of the groundwater. When the PFM is removed from the well, the contaminant that sorbed to the sorbent, as well as the quantity of resident tracer that desorbed, are measured (Campbell et al., 2006). The sorbed contaminant mass and desorbed resident tracer are used to calculate a flux-averaged contaminant concentration and groundwater Darcy velocity, respectively, at the location of the PFM. Equation 2.1 is then used to determine mass flux. By using multiple PFMs across a control plane, an average mass flux and a total mass discharge over the control plane may be obtained (Hatfield et al., 2005).

To determine the average groundwater specific discharge (Darcy velocity) q_d [L/T] through the PFM, without accounting for convergence or divergence of flow, apply the equation (Hatfield et al., 2004):

$$q_d = \frac{1.67(1-\Omega_r)r\theta R_d}{t} \qquad (2.7)$$

where Ω_r is the mass fraction of residual tracer remaining on the PFM after the device has been exposed to flowing groundwater for time t, r is the radius of the borehole or well, θ is the volumetric water content of the sorbent, and R_d is the retardation factor for the resident tracer onto the sorbent.

It is possible for Ω_R values to fall outside the range of values (0.32-0.7) calculated theoretically by Hatfield et al. (2004) and Basu et al. (2006) as the upper and lower limits of residual tracer remaining to accurately calculate groundwater flux. However, it is proposed that exceeding this theoretical range of values can be done without significant loss in accuracy, so long as the remaining tracer mass is not excessively low (ESTCP, 2006; Hatfield et al., 2004)

To determine the ambient Darcy velocity, we must account for the convergence or divergence of groundwater around the PFM. Since q_d is linearly proportional to the ambient groundwater flux, q_o, then:

$$q_d = \alpha q_o \qquad (2.8)$$

where α is the convergence/divergence factor and is a function of the difference in hydraulic conductivity between the surrounding aquifer and that of the well and borehole. This is the same factor discussed earlier during our explanation of the BDT calculations. Applying Equation 2.9, the Darcy velocity can be determined:

$$q_o = \frac{1.67(1-\Omega_r) r\theta R_d}{\alpha t} \qquad (2.9)$$

Having used the resident tracer to measure the groundwater Darcy velocity, we now use the sorption of contaminant onto the PFM to estimate the flux-averaged dissolved contaminant concentration. It is first necessary to measure the sorbed contaminant mass, m_c. Then, the flux-averaged contaminant concentration C [M/L^3] can be estimated from Equation 2.10 (Hatfield et al., 2004):

$$C = \frac{m_c}{\pi r^2 b A_{RC} \theta R_{DC}} \qquad (2.10)$$

where b is the length of the sorbent material, A_{RC} is a dimensionless term which quantifies the fraction of sorptive matrix containing contaminant (which can be estimated based on how much resident tracer desorbed), θ is the volumetric water content of the sorbent, and R_{DC} is the retardation factor for the contaminant onto the sorbent. The time-averaged incremental contaminant mass flux (averaged over the aquifer area interrogated by the PFM (dA) is determined using equation 2.1. Spatial integration of the incremental measurements of contaminant mass flux, J, yields a total time-averaged contaminant mass discharge (M_d) [M/T] across the control plane of area A:

$$M_d = \int_A JdA = \int_A q_o CdA \qquad (2.11)$$

2.3.2.1 Laboratory and field testing of the PFM

To validate the PFM, multiple box-aquifer experiments were conducted using ethanol as a resident tracer (Hatfield et al., 2004). These experiments measured water flux (Darcy velocity) with relative errors of 4%. Subsequently, using 2,4 dimethyl-3-pentanol (DMP) as a contaminant, flux (*J*) was calculated within 5% of known fluxes (Hatfield et al., 2004). More recently, column tests of a new PFM configuration that used a granular anion exchange resin as the sorbent and benzoate as the resident tracer were conducted. The granular anion exchange resin was chosen in order to capture hexavalent chromium (Cr(VI)), an inorganic contaminant. The contaminant was selected for study due to its high mobility and toxicity. The test, which was also done to see if the PFM could determine the direction of groundwater flux as well as the magnitude, measured the Darcy velocity direction with an error of 3±14°. The errors in measuring groundwater flux magnitude and contaminant flux were -8% ±15% and -12% ± 23%, respectively (Campbell et al., 2006).

In 2004, the PFM method was applied at a former electronic-part manufacturing facility site that was contaminated with TCE. By using the PFM, it was determined that there were two distinct zones, one shallow and one deep, where the groundwater flux was considerably different: 2 cm/day and 15 cm/day in the shallow and deep zones, respectively. This finding was significant, as previous site characterizations had not identified these zones, and the selected remediation design did not account for them (Basu et al., 2006). Subsequent to the PFM application at the site, the TM was applied and similar flux measurements were obtained (Basu et al., 2006).

The PFM has been demonstrated at 25 sites in the US, Canada, Australia, and Wales in an effort to gain regulatory and end-user acceptance and to stimulate transfer and commercialization of this innovative technology (ESTCP, 2007). Field demonstrations and comparisons with the TM/MLS were conducted at Canadian Forces Base (CFB) Borden, NASA's LC-34, Cape Canaveral, Florida; Naval Base Construction Base, California; and Naval Surface Warfare Center at Indian Head, Maryland.

At CFB Borden, the PFM was compared to known (historical) groundwater fluxes as well as fluxes measured using the BDT. These "known" fluxes were between 5 and 8 cm/day. Deploying the PFM for 7.3 weeks, Darcy fluxes of 6.6 cm/day were measured. Also at Borden, an experiment was conducted within a three–sided box that was constructed in the aquifer using barrier technology. Groundwater flowed into the box at the upstream (open) end. At the downstream end of the channel was a pumping well, pumping at 203 mL/min. Methyl tertiary butyl ether (MTBE) was injected upstream. Within the box were three rows of three wells each. The first row consisted of three MLS wells. The second row had three PFMs, installed without sand packs. The third row had three PFMs with sand packs around the wells (Annable et al., 2005). Mass discharge was measured at 0.54, 0.47, and 1.1 g/day for the MLS, PFM without sand packs, and PFM with sand packs, respectively (Annable et al., 2005). Contaminant (MTBE) fluxes measured by the PFMs were compared to fluxes calculated by multiplying the induced specific discharge going into the channel by the average MTBE concentration measured at the extraction wells and the MLSs in the channel. The results for the PFM (no sand packs) were within 16.6% of the flux calculated from the MLS measurements and the PFM (sand pack) measurements were within 1.2% of the MLSs' (ESTCP, 2007).

NASA's LC-34 site is contaminated with TCE due to use of the solvent in support of rocket launch activities. An experimental flow cell was constructed at the site, consisting of three injection and three extraction wells, along with five MLS wells within the cell. Flux monitoring was conducted using PFMs and the MLSs before, during, and after a bioremediation experiment at the site. The bioremediation experiment consisted of injecting ethanol as an electron donor to stimulate biodegradation of TCE, followed by bioaugmentation, whereby chlorinated solvent biodegrading microorganisms were injected into the aquifer. Groundwater flux measured using the PFMs was compared to the groundwater flux that was calculated based on flow between the injection and extraction wells. Contaminant flux measured using the PFMs was compared to the contaminant flux estimated by applying the TM to the MLS concentration data. In addition, contaminant mass discharge was estimated by integrating the PFM-estimated fluxes over the flow cell cross-sectional area. This mass discharge estimate was compared to the mass discharge measured at the extraction wells.

Prior to the bioremediation experiment, the groundwater flux measurements by the PFM were within 6-19% of the calculated flux. After the injection of ethanol to stimulate biodegradation, accuracy decreased, though the difference between the PFM estimate and the calculated flux remained within 67%. Following bioremediation, the percent difference ranged between 4 and 30% (ESTCP, 2007). It is believed that the biological activity induced during the bioremediation resulted in some degradation of the ethanol resident tracer, resulting in the observed loss of accuracy in the PFMs estimates of groundwater flux, which depend upon the resident tracer (ESTCP, 2007).

The average contaminant flux prior to remediation was measured using both the MLS and the PFM. Averaging over the transect planes, the difference between the PFM- and MLS-measured fluxes prior to bioremediation ranged from 0 to 23%. During and after bioremediation, the difference between the two methods ranged from 17 to 190%. The difference between the mass discharge estimated by the PFM method with the "actual" discharge measured at the extraction wells ranged from 32 to 190%. PFM estimates of vinyl chloride (VC) and ethene fluxes (two compounds that are TCE degradation byproducts) were greater than the estimates obtained from the MLSs and extraction wells. Project investigators speculate that TCE degradation to VC and ethene might be occurring on the PFM sorbent, thus leading to overestimates of VC and ethene fluxes (and an underestimate of TCE flux) (ESTCP, 2007).

At Port Hueneme, there is a contaminated site located downgradient from a Navy Exchange Service Station. The service station reported a subsurface release of 10,800 gallons of gasoline containing MTBE in 1984 and 1985. PFM testing was conducted in well clusters which allowed for side-by-side comparison of PFM-measured groundwater discharges with measurements made by the BDT and slug test methods. Although the methods were expected to measure groundwater flux within 25%, the results varied by a factor of 2 to 3 (ESTCP, 2007). Only the PFM was used to measure MTBE flux. However, due to natural aquifer heterogeneities, contaminant flux measured in adjacent wells were not comparable (ESTCP, 2007)

The Indian Head site is located at the Naval Surface Warfare Center, Indian Head, Maryland. At this facility, solid rocket propellant, containing ammonium perchlorate, was cleaned out from devices such as spent rocket and ejection seat motors. This resulted in a perchlorate plume near the facility. The groundwater gradient was 0.023 and average hydraulic conductivity (based on slug testing) was 526 cm/day. Thus, groundwater flux based on the slug test and the hydraulic gradient was 12 cm/day (ESTCP, 2007). In addition, groundwater flux measured using a BDT was 3.5 cm/day (Lee et al., 2007). PFMs were installed in existing wells with 10 ft screens. Two transects, one near the source and one further downgradient within the plume, were established for deployment of the PFMs. Two separate deployments took place. The first was for 3 weeks and the second for 7.3 weeks. The test focused on demonstrating the applicability of surfactant modified silver impregnated GAC (SM-SI-GAC) in the PFM for use in measuring groundwater and perchlorate fluxes (Lee et al., 2007; ESTCP, 2007). The groundwater fluxes measured by the PFM for three wells were: 1.8 cm/day for MW1, 2.8 and 2.1 cm/day for MW4, and 7.6 and 4.9 cm/day for MW3. Wells MW1 and MW4 compared well to the BDT and MW3 was within a factor of 2. The perchlorate fluxes measured at the three wells ranged from 0.22 to 1.7 g/m^2/day. Uncertainties in estimating the groundwater flow divergence factor, as well as the proximity of well MW4 to an ongoing bioremediation study, may have contributed to the variations in the flux measurements. The field test demonstrated that SM-SI-GAC can be used as a sorbent for the PFM at sites with perchlorate concentrations ranging from 7 to 64 mg/L (ESTCP, 2007).

2.3.2.2 PFM costs

In their report, ESTCP (2007) set up a notional template site which consisted of 10 wells spaced 10 ft apart in a transect, with each well screened over 10 ft depth in the saturated subsurface. Thus, there was a total of 100 linear feet (lf) of screened well installed. It was assumed that 20 five-foot long PFMs would be deployed in these wells. The vertical sampling interval for the PFMs is assumed to be one foot; resulting in a total of 100 data points. The cost of using the PFM to measure groundwater and contaminant flux was compared to the costs of using the TM (with MLSs and a BDT at each MLS sampling point). The number of sampling points for the MLS and BDT were assumed to be the same as the PFM (i.e., 100 sampling points total). ESTCP (2007) estimated the PFM approach applied to the template site, using the one-foot vertical resolution described above, would cost \$303/lf, as compared to \$430/lf for the TM. The breakouts of the costs for the TM and PFM methods at the template site are shown in Tables 2.7 and 2.8, in section 2.5.

A cost analysis was conducted by Kim (2005) for a template site consisting of a 200 m wide by 10 m deep contaminant plume in a confined sand aquifer. His analysis concluded the costs for implementing the PFM method would be \$155,000 or 99% of the cost of implementing the transect method at the site (Goltz et al., 2007b). Note that the cost analysis for the PFM method has considerable uncertainty, as only cost data to construct the PFMs and analyze the tracer and contaminant concentrations are based upon research studies conducted at Purdue University and the University of Florida (ESTCP, 2007; Annable et al., 2005; Basu et al., 2006). It is unclear what costs will be once technology use is more widespread.

2.3.2.3 Regulatory concerns

Besides the concern that regulators may have with the accuracy of using this innovative technology to measure groundwater contaminant flux, the only other concern might be with the release of resident tracers into the groundwater. In some circumstances, approval to use the resident tracer might require a permit, with the associated costs in time and money (ESTCP, 2007).

2.3.2.4 Advantages and limitations

The PFM method has no requirement for on-site electric power, making the method very amenable for use at remote sites. PFMs are easy to install and little site labor is needed for the duration of the test (ESTCP, 2007). The PFM also offers the same flexibility as the MLS and becomes more economical when high spatial resolution monitoring is required (ESTCP, 2007). In addition to contaminant and groundwater flux, the PFM can also be used to determine the direction of groundwater flow (Campbell et al., 2004). The fact that the PFM provides a time-averaged flux is also useful (Basu et al., 2006). Since the test duration is days to weeks long, time-averaging avoids anomalies that may be seen from point sampling in time (Goltz et al., 2007b). Of course, this means that a PFM measurement requires weeks, as compared with the TM, that provides results quickly.

As a new technology, the PFM does require special expertise in selecting the appropriate sorbent and resident tracer. As noted in the discussion of the PFM evaluation at LC-34, both the sorbed contaminant and the resident tracer may be susceptible to bioactivity, which would skew results (ESTCP, 2007).

Like the BDT, the PFM requires estimation of the convergence and divergence factor (α). The error associated with estimating this factor directly affects the estimate of flux.

2.3.3 Tandem circulating wells (TCWs)

Goltz et al. (2007a) and Huang et al. (2004) proposed an innovative approach to measuring contaminant mass flux through the use of tandem circulating wells (Kim, 2005). The TCW technique uses two dual-screened wells (figure 1.3). One well extracts water from lower depths of the aquifer and pumps the water upwards, injecting it at shallow depths, while the other well operates in the opposite direction. This results in circulation of water between the two wells. The extent of circulation is a function of pumping rate, hydraulic conductivity, distance between the wells, well screen locations, and the regional hydraulic gradient. The TCW method offers the advantage of obtaining volume-averaged measurements of contaminant concentration (C) and hydraulic conductivity (K) by interrogating a large volume of subsurface water, without the need to extract water from the subsurface. Knowing C and K, along with an independent measurement of the regional hydraulic gradient (i) that can be determined by measuring the piezometric surface at the two TCW wells (with pumps turned off) and a third piezometer, contaminant mass flux can be calculated using equation 2.2 (Yoon, 2006; Goltz et al., 2007a). The details of obtaining volume-averaged contaminant concentration and hydraulic conductivity measurements are described below.

For concentration measurements, the samples are taken directly from the wells as water circulates through them, providing a volume-averaged concentration. Calculating the hydraulic conductivity using TCWs can be accomplished in one of two ways. One way is by application of the multi-dipole technique (TCW-MD), which requires that the extent of drawdown (for the downflow TCW) and mounding (for the upflow TCW) be measured for various TCW pumping rates (Yoon, 2006). The second approach, the tracer test technique (TCW-T), uses tracers injected in each of the two TCWs to quantify the extent of circulation (referred to as interflow) between the two TCWs (Goltz et al., 2007a).

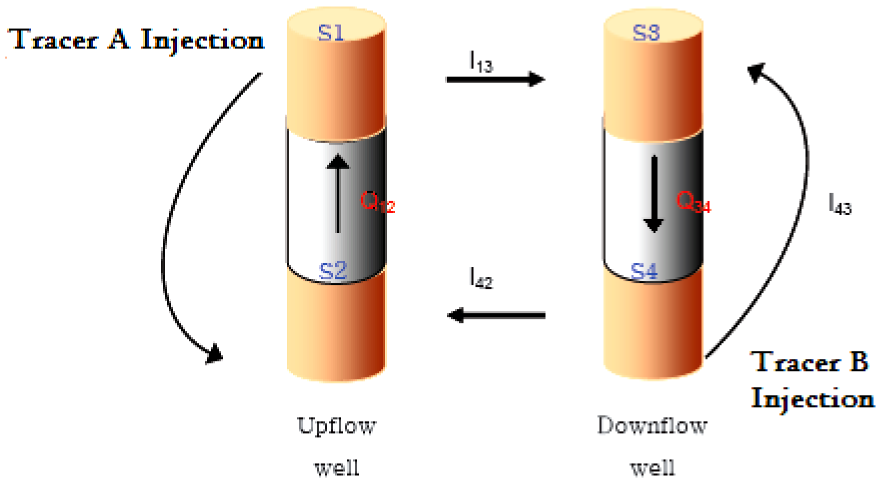
The multi-dipole technique is an adaptation of the single-well dipole flow test method to simultaneously measure both horizontal and vertical hydraulic conductivity that was developed by Kabala (1993) (Goltz et al., 2007a). To apply the method, the TCWs are operated at various flow rates and the drawdown and mounding are measured at the downflow and upflow TCW, respectively, for each flow rate. Goltz et al. (2004) presented a method that makes use of a genetic algorithm to determine values for horizontal and vertical hydraulic conductivities that result in a best fit of model-simulated (H_{calc}) and measured (H_{meas}) values of drawdown and mounding. Model-simulated values of drawdown and mounding may be obtained by either analytical (Goltz et al., 2007a) or numerical (Harbaugh and McDonald, 1996) modeling. The best fit value for hydraulic conductivity will maximize the objective function in Equation 2.12 below.

$$F_{obj} = \frac{1}{1 + \sum_{i=1}^{N} (H_{meas}^{i} - H_{calc}^{i})^2} \qquad (2.12)$$

where H_{meas}^{i} and H_{calc}^{i} indicate the measured and calculated hydraulic heads at the i^{th} flow rate, respectively, and N is the total number of head measurements.

For cases where drawdown and mounding induced by the TCW are difficult to measure, due to their relatively small magnitudes (which may occur in high hydraulic conductivity systems), the tracer technique is an alternative method. This technique is based on determining the fractional flow between the two TCWs by injecting and monitoring tracers at each of the two TCW injection screens (Goltz et al., 2007a).

Figure 2.2 Fractional flows between TCW screens



The fractional flow (I_{ij}) is defined as the fraction of water drawn into the j^{th} extraction screen which originated from the i^{th} injection screen (Figure 2.5). If a step concentration of tracer A is injected into the upflow well and a step concentration of tracer B is injected into the downflow well, when steady-state is achieved, by mass balance (Goltz et al., 2007a; Huang et al., 2007):

$$A_4 I_{43} + A_2 I_{23} = A_3 \qquad (2.13a)$$

$$B_4 I_{43} + B_2 I_{23} = B_3 \qquad (2.13b)$$

$$A_4 I_{41} + A_2 I_{21} = A_1 \qquad (2.13c)$$

$$B_4 I_{41} + B_2 I_{21} = B_1 \qquad (2.13d)$$

where A_i is the concentration of tracer A measured at the i^{th} well screen and B_i is the concentration of tracer B measured at the i^{th} well screen. Tracer concentrations (A_i and B_i) are measured at the injection and extraction screens over time (after steady-state has been achieved)

(Goltz et al, 2007). The measured concentrations may be averaged; Kim (2005) showed that the calculation is not sensitive to the averaging method that is used.

Having values for A_i and B_i, Equation 2.13 may be solved for the four fractional flows (I_{ij}^{meas}). The numerical model MODFLOW (Harbaugh and McDonald, 1996) can then be employed to calculate simulated values of fractional flow (I_{ij}^{calc}) for the TCW flow rate used in the test (Goltz et al., 2007a). The optimum hydraulic conductivity may then be found by using a genetic algorithm to maximize the following objective function (F_{obj}):

$$F_{obj} = \frac{1}{1 + \sum_{i=1}^{N_{inj}} \sum_{j=1}^{N_{ext}} (I_{ij}^{meas} - I_{ij}^{calc})^2} \tag{2.14}$$

where N_{ij} and N_{ext} are the number of injection and extraction well screens, respectively, and N is the total number of well screens.

2.3.3.1 Laboratory testing of the TCW

TCW laboratory experiments were conducted in Canterbury, New Zealand, in a large "meso-scale", homogeneous sand aquifer. The aquifer was 9.5 m long, 4.7 m wide, and 2.6 m deep and filled with sifted sand ranging in size from 0.6 to 1.2 mm in diameter. Constant head tanks at either end of the aquifer provided a "regional" hydraulic gradient (Goltz et al., 2007a). The target contaminant was chloride which was naturally present in the water. The actual mass flux was calculated by multiplying the chloride concentration in the influent water by the flow through the aquifer and dividing by the cross-sectional area. Both the tracer and multi-dipole approaches were tested, and the results were compared with the TM. The results of the experiments show relatively accurate results for the multi-dipole approach and for the tracer approach. Goltz et al. (2007a) report the multi-dipole approach estimated a hydraulic conductivity of 173 m/d. The measured hydraulic gradient was 0.00132 and contaminant concentration was 10.5 g/m^3. Applying equation 2.2, the estimated mass flux was 2.39 $g/m^2/d$. This compares very well with the actual flux of 2.41 $g/m^2/d$.

The tracer test used bromide injected into the injection screen of the upflow well and nitrate injected into the injection screen of the downflow well. The bromide and nitrate injections continued for 240 and 336 hours, respectively, until steady-state concentrations were reached at the extraction wells (Goltz et al., 2007a). Values for the hydraulic conductivity were obtained by applying equations 2.13 and 2.14. The TCW tracer test resulted in a flux estimate of 2.53 $g/m^2/d$, 13% greater than the actual flux. Unfortunately, although TCWs have been used in the field to implement *in situ* remediation (McCarty et al., 1998), TCWs have never been implemented to measure contaminant mass flux in the field.

2.3.3.2 TCW Costs

Kim (2005) estimated the costs of measuring flux at a template site using the TCW (see template site description in Section 2.5). He calculated that the multi-dipole approach would cost $84,000 and the tracer approach would cost $92,000. The cost estimates assumed installation of two 8-inch wells plus a single 2-inch piezometer. The tracer test was conducted for 12.5 days. The cost of the multi-dipole and tracer approach equated to 54% and 59%, respectively, of the costs associated with implementing the TM with a single pump test for estimating hydraulic conductivity.

2.3.3.3 TCW regulatory concerns

The TCW tracer method requires the use of tracers which may raise some regulatory concerns (Goltz et al., 2007b) and may require permitting. There is also the issue of possible contamination of clean groundwater due to circulating water between the lower and upper depths of an aquifer. Additionally, the fact that the technology has never been field-tested will probably lead regulators to question its accuracy.

2.3.3.4 TCW Advantages and limitations

The inherent advantage of the TCW method is that it is a volume-averaged approach, eliminating the need for the extensive sampling (and associated costs) that is required when applying point measurements in heterogeneous systems (USEPA, 2007). The TCW method avoids the extraction and treatment of contaminated groundwater (Goltz et al., 2007a). Since the TCW method does not require that groundwater be pumped to the surface, the test is economical when characterizing deep plumes (Goltz et al., 2004). The TCW method can be of greater value if conditions do not allow for the installation of many wells for a large control plane due to geologic conditions or surface obstructions.

Although the TCW technique has been used in the past for remediation of contaminant plumes, it has not been applied in the field to measure flux (Yoon, 2006). Depending on hydraulic conductivity, pumping rate, and distance between the TCWs, the tracer technique may require a relatively long time (perhaps weeks) to establish steady-state tracer concentrations (USEPA, 2007). Another disadvantage is that the calculations for determining the flux (e.g., application of a genetic algorithm to estimate hydraulic conductivity) can be rather complex (USEPA, 2007).

2.3.4 Modified Integral Pump Tests (MIPTs)

Like the TCW method, the MIPT method obtains a volume-averaged measurement by interrogating a large volume of subsurface water (Brooks et al., 2007). The MIPT is a simplified version of the IPT, requiring less time to implement and therefore having the potential to reduce costs associated with labor and disposal of extracted water.

The original IPT was introduced as a flux measurement technique by Teutsch et al. (2000). To apply the IPT to measure flux, one or more pumping wells are installed along a control plane down gradient of a contamination source and orthogonal to the regional groundwater flow direction (Brooks et al., 2005; SERDP, 2004). Enough wells are installed to ensure capture of the entire contaminant plume (Bockelmann et al., 2001). The wells are pumped simultaneously or sequentially for several days and contaminant concentration measured as a function of time (Ptak et al., 2003). This concentration-time series information is used to estimate average contaminant concentration based on a number of assumptions: (1) flow towards the extraction wells is radially symmetric and regional flow can be ignored during the pumping test, (2) the aquifer is homogeneous with regard to porosity, hydraulic conductivity, and thickness, and (3) the concentration does not vary significantly within each streamtube at the scale of the well capture zone (Goltz et al., 2007b; Bockelmann et al. 2001). Hydraulic conductivity and gradient are measured separately, and Equations 2.2 and 2.3 are then used to estimate contaminant mass flux and mass discharge, respectively (Bockelmann et al., 2003).

The IPT has had several field implementations at various European industrial areas (megasites) to include Strausbourg, Linz, Stuttgart, and Milano. It was concluded that the IPT was capable of quickly and with certainty estimating the average contaminant concentration, spatial distribution of concentration, and mass discharge of contaminant down gradient of a source (Ptak et al., 2003).

The MIPT method also uses pumping wells, along with monitoring wells, to measure flux. The basis of the MIPT method is to determine the regional Darcy velocity, q_0, by measuring the head difference between the pumping wells and the monitoring well(s), when the pumping wells are pumped at different flow rates. If we assume a homogeneous confined aquifer of saturated thickness B and steady flow, the head difference between the pumping well at the origin and the monitoring well a distance Δx directly downgradient (Figure 2.5) can be expressed as a function of the sum of the flows in each of N pumping wells (Q_i as i goes from 1 to N) (Yoon, 2006; USEPA, 2007).

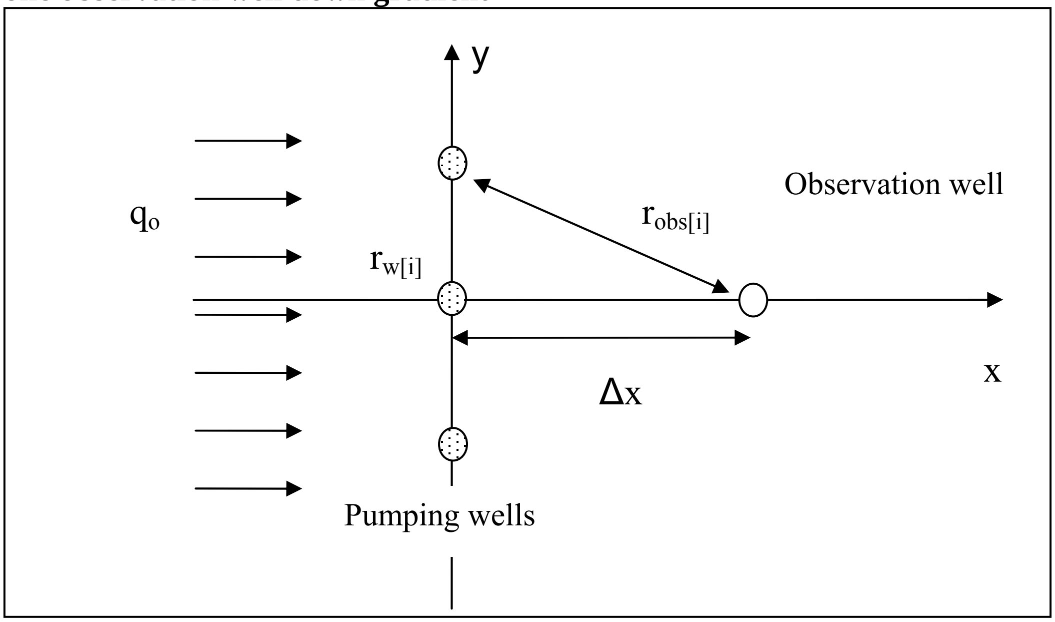
$$\Delta h = -\frac{q_0 B}{T}\Delta x + \frac{1}{4\pi T}\sum_{i=1}^{N} Q_i \ln \frac{r_{obs[i]}{}^2}{r_{w[i]}{}^2} \qquad (2.15)$$

where $r_{obs[i]}$ is the distance to the observation well from the i^{th} pumping well, $r_{w[i]}$ is the distance between the i^{th} pumping well and the origin, and T is the aquifer transmissivity, which is the hydraulic conductivity multiplied by the aquifer saturated thickness. The head differences are measured at different pumping rates, and plotted against the summation term on the right-hand-side of Equation 2.15. A best-fit line is constructed from the Δh versus $\sum_{i=1}^{N} Q_i \ln \frac{r_{obs[i]}{}^2}{r_{w[i]}{}^2}$ data. The line crosses the x-axis when $\Delta h = 0$. The value of $\sum_{i=1}^{N} Q_i \ln \frac{r_{obs[i]}{}^2}{r_{w[i]}{}^2}$ at the x-axis intercept allows calculation of the regional Darcy velocity, q_0, using equation 2.16:

$$q_0 = \frac{1}{4\pi B \Delta x}\left[\sum_{i=1}^{N} Q_i \ln \frac{r_{obs[i]}{}^2}{r_{w[i]}{}^2}\right]_{x=0} \qquad (2.16)$$

Now knowing the Darcy velocity (q_0) the contaminant mass flux can be calculated from the product of Darcy velocity and the average concentration measured at each of the pumping wells.

Figure 2.3, An example of the MIPT approach with multiple pumping wells and one observation well down gradient



It is important to capture the entire contaminant plume by the transect of pumping wells. The pumping rate required to capture the entire flow across the control plane can be determined by:

$$\sum_{i=1}^{N} Q_i = wq_0 B \qquad (2.17)$$

where *w* is the width of the plume captured by the pumping wells.

2.3.4.1 Laboratory and Field Testing of MIPT

The MIPT underwent testing at the three dimensional meso-scale confined aquifer in Canterbury, New Zealand, described in Section 2.3.3.1. The MIPT was used in three experiments. In each experiment, the pumping wells were pumped at 5 different rates and head measurements made for each rate. Experiments 1 and 3 employed a single pumping well and experiment 2 employed three pumping wells. Experiments 1, 2, and 3 under estimated the actual flux by 48, 42, and 61% respectively (Goltz et al., 2007b).

The MIPT was fielded at a contaminated site at Hill AFB, Utah, where degreasing solvents, primarily TCE, had been released (Brooks et al., 2007). The site has been undergoing remediation since the early 1990s. Before and after a source remediation effort in July 2002, the MIPT was used to measure the contaminant flux change that resulted from the remediation. Prior to the remediation effort, a transect of 10 wells spaced 10 feet apart was installed. The primary contaminants in the groundwater were TCE and DCE. During the pre-remediation test, 4 pumping steps were applied; for the first post remediation test, three steps were used; while for the second post remediation test, 5 steps were used (Brooks et al., 2007). Groundwater samples were taken at 3 to 6 hour intervals during each MIPT. Hydraulic gradients were measured before all the tests. Using historical data from previous studies (Meinardus et al., 2002; Rao et al., 1997), the hydraulic conductivity was estimated at 17 m/d (Brooks et al., 2007). Applying Equation 2.4, Darcy velocities of 3.4, 3.4, and 0.9 cm/day were estimated for each of the three tests. Although the actual contaminant mass flux was not known, the test results were compared to results obtained from the TM, which used previously measured values for hydraulic conductivity and gradient (see equation 2.2). Based on the TM, transect-wide average Darcy fluxes of 2.9, 1.7, and 3.2 cm/day were measured for the pre-remediation and two post-remediation tests, respectively. This compares well with the groundwater flux estimated from previously reported hydraulic conductivity and hydraulic gradient measurements. Based on the MIPTs and measured hydraulic gradients, the following values of hydraulic conductivity were calculated for each of the three tests: 19.0, 12.0, and 18.0 m/day. These three estimates of hydraulic conductivity are averaged to obtain a value of 16 m/day, with a standard deviation of 0.40 m/day. This compares reasonably well with previously published estimates (Brooks et al., 2007).

TCE mass discharge measurements using the MIPT yielded values of 76, 3.9, and 7.1 g/day for pre-remediation, first post-remediation, and second post-remediation tests, respectively. Also measured was DCE mass discharge of 2.0 and 20 g/day for the first and second post-remediation tests, respectively. This compares with TCE mass discharge measurements using the transect approach of 78, 7.2, and 1.3 g/day for the pre-remediation, first post-remediation, and second post-remediation tests, respectively, and DCE mass discharge measurements of 3.8 and 4.2 g/day for the two post-remediation tests (Brooks et al., 2007). The PFM was also tested at the site. Comparison of the PFM and MIPT methods is discussed in Section 2.4.

At the Fort Lewis, Washington, East Gate Disposal Yard, a release of chlorinated solvents (mostly TCE) had occurred from drums. The TCE plume is well established and extends north for approximately 4 kilometers. A pump-and-treat system was installed in 1995 for hydraulic control purposes and the drums were removed in 2000 and 2001 (Brooks et al., 2007). In 2003, thermal treatment was used to treat the source area. Before (Phase I) and after (Phase II) the thermal treatment, the MIPT was used to measure flux. Hydraulic gradients were measured during each of the two MIPTs. The MIPT used 10 wells spaced along a 60 m transect. During Phase I, the MIPT pumping rates were not large enough to stress the aquifer for sufficient drawdown, so groundwater flux was not directly measured. However, after Phase II was complete, the Phase I flux was estimated by scaling the Phase II estimated groundwater flux by using the ratio of hydraulic gradients of the two phases. The resulting groundwater fluxes were then estimated to be 32 and 18 cm/day for Phases I and II, respectively (Brooks et al., 2007). This compares well with the 34 and 19 cm/day estimates that were made based on multiplying the measured hydraulic gradients by the average hydraulic conductivity at the site.

Based on the MIPT-measured groundwater fluxes, the TCE mass discharge was calculated at 536 and 2.2 g/day for pre- and post-remediation tests, and the DCE discharge calculated at 257 g/day for the pre-remediation test (DCE was not detected in the post-remediation sample analysis). These mass discharge estimates compared well with pre-remediation estimates made using the TM of 688 g/day for TCE and 288 g/day for DCE. The post-remediation measurement from the MIPT for TCE (2.2 g/day) also compared well with the estimate made using the TM (2.8 g/day) (Brooks et al., 2007). A summary of the mass discharge measurements at Fort Lewis may be seen in Table 2.1 in section 2.4.

Field tests of the MIPT method have also been conducted at a tetrachloroethylene (PCE)-contaminated dry-cleaning site in Jacksonville, FL, and a dense nonaqueous phase liquid (DNAPL) spill site at Canadian Forces Base Borden. However, neither the Jacksonville nor Borden site data have been published.

2.3.4.2 MIPT costs

There were no cost data found in literature. In Chapter 3, we develop a cost estimation methodology for the MIPT method, which is subsequently applied in Chapter 4.

2.3.4.3 MIPT Regulatory issues

The extraction of potentially contaminated groundwater is of regulatory concern.

2.3.4.4 MIPT Advantages and limitations

Like the TCW method, the MIPT method obtains a volume-averaged measurement, thus eliminating the need for the large number of sampling points which would be required to characterize a heterogeneous site (SERDP, 2004). The fact that the MIPT is an active measurement method enables varying the pumping rates to accommodate seasonal variations in groundwater flow (Brooks et al., 2007). MIPT measurements have been shown to be consistent with the PFM and TM methods (Brooks et al, 2006; SERDP, 2004).

In the MIPT, it is assumed that hydraulic steady-state is attained after pumping for 24 hours. The validity of this assumption will be site dependent. In addition to labor costs, the extracted contaminated water must be treated and properly disposed of. This could increase costs significantly; depending on location and site conditions, disposal costs can be as high as $1/gal of extracted water (USEPA, 2007). Additionally, in order to obtain a steady concentration measurement, pumping on the order of weeks to months may be required (USEPA, 2007). Recently, modifications to the MIPT are being tested, with the goal of reducing the duration of pumping (and therefore, reducing labor costs, as well as the costs associated with treating and disposing of potentially contaminated groundwater) (SERDP, 2004).

2.4 Method comparison

Some comparisons where the different methods were applied at the same location and time have appeared in the literature. Brooks et al. (2006; 2007) compared the TM with the MIPT and PFM methods at Fort Lewis and Hill AFB. Although the actual flux was unknown, application of the MIPT and PFM methods resulted in estimated fluxes that were within 8 to 38% of the TM estimate at Fort Lewis and within 15 to 40% of the TM estimate at Hill AFB. In reference to the accuracy of application of the MIPT method at Hill AFB (see Section 2.3.4.1), Brooks et al. (2006) has stated:

> … it can be generally concluded that the uncertainty estimates as presented [Hill AFB] are comparable. Furthermore, in the work by Kubert and Finkel (2006), for a system comparable to the Hill AFB, transect error estimates associated with mass discharge were in the range of 15 to 40%. …Thus the conclusion that these methods are in general agreement is valid.

Table 2.1 reports results from the Hill AFB and Fort Lewis field tests, where the PFM, MIPT, and TM were applied to measure contaminant mass discharge.

Table 2.1 Comparison of mass discharge measurement methods at Hill AFB and Ft Lewis (after Brooks et al., 2007)

Summary of contaminant (TCE and DCE) mass discharge rates (g/day) as estimated using PFM and MIPT results, and comparison with corresponding estimates based on the Transect Method (TM). (Brooks et al., 2006)

Contaminant	Method	Hill AFB		Ft. Lewis	
		Phase I (Pre-Remediation) May-02	Phase II (First Post-Remediation) Jun-03	Phase I (Pre-Remediation) Oct-03	Phase II (First Post-Remediation) Jun-06
TCE (g/day)	PFM	76	6	743	3.4
	MIPT	76	3.9	536	2.2
	TM	78	7.2	688	2.8
DCE (g/day)	PFM	[1] -	3	155	5.7
	MIPT	[2] -	2	257	[2] -
	TM	[2] -	3.8	288	[2] -

Notes:
[1]DCE concentrations in SI-GAC extracts were below the level of quantification.
[2]DCE was not included in the analysis of all samples.

The different flux measurement methods have been compared using criteria beyond cost and accuracy (ESTCP, 2007; Kim, 2005; USEPA, 2007; Goltz at el, 2007b; Brooks et al., 2007). Evaluation of the methods took a broad view and included such aspects as ease of implementation, complexity of analysis, ease of data interpretation, regulatory issues, method availability, and other concerns. Depending on the reference, the methods were ranked numerically or descriptively (from "good" to "poor"). These comparisons are shown in Tables 2.2 through 2.5.

Table 2.2 Comparison of methods of measuring Darcy velocity (after USEPA, 2007)

Applicability	Method	Ease of Field implementation		Complexity of analysis and preparation	Data Interpretation for q_D	Data Interpretation of q_0		Concerns
		Ranking						
Point Scale	BDT	Requires Pumps and in situ monitors or time series sampling	2	1	2	1	Very Simple modeling	Values obtained may be precise but not accurate due to difficulty in obtaining α
	PFM	Relative easy pre-prepared sorbent sampled and installed then removed and resampled	1	4	1	2	Very Simple Calculation	Values obtained may be precise but not accurate due to difficulty in obtaining α.
Point Scale	MIPT	Requires pumps, head monitoring equipment, and in situ monitors or time series sampling	4	2	n/a	3	Easy Calculation	Large volumes of potentially hazardous water must be treated. May be difficult to maintain constant low over the range of desired flow rates. Small differences in
	TCW	Complex pumping and in situ monitoring or time series sampling	3	2	n/a	4	More difficult modeling and inversion	Short circuiting of flow near well screen may reduce precision of the local scale parameters and may limit valid data to point scale values.

Table 2.3 Comparison of mass flux measurement methods (after USEPA, 2007)

Applicability	Method	Ease of Field implimentation		Complexity of analysis and preparation	Data Interpretation for Mass Flux	Data Interpretation of Mass Discharge		Concerns
		Ranking						
Point Scale	PFM	Relative easy pre-prepared sorbent sampled and installed then removed and resampled	2	4	2	3	Easy Calculation	Values obtained may be precise but not accurate due to difficulty in obtaining α. Short circuiting in wire rapped well screens.
Local Scale	MIPT	Require pumps, head monitoring equipment, and in situ monitors or time series sampling and analysis	4	2	1	2	Easy Calculation	Requires very accurate survey data to resolve small head differences to determine q0. Large volumes of water produced that may need treatment
	TCW	Complex pumping and in situ monitoring or time series sampling and analysis	5	2	5	6	Requires complex inverse modeling	Complicated numerical inversion required to interpret data
Tables 2.2 and 2.3 (EPA, 2007) rank the different methods 1-6 with 1 being best/easiest and 6 being worst or most difficult.								

Table 2.4. Comparison of flux measurement methods (after Kim, 2005)

	TM	PFM	MIPT	TCW
Cost	Poor	Poor	Moderate	Good
Simplicity/Implementability	Good	Moderate	Moderate	Poor
Regulatory Considerations	Good	Moderate	Good	Poor
Availability	Good	Poor	Moderate	Poor

Table 2.5 Comparison of groundwater contaminant flux measurement methods (after Goltz et al., 2007b)

	TM	PFM	MIPT	TCW
Cost	4	4	2	1
Simplicity/Implementability	1	3	1	4
Regulatory Considerations	1	1	3	3
Availability	1	2	1	4
Note: 1 is the best and 4 the worst				

From Tables 2.2-2.5, there would seem to be general agreement that:

1. The TM is readily available and the simplest method to implement. When applied in conjunction with the BDT, accuracy is affected by the need to estimate the convergence/ divergence factor, α.
2. The PFM is less available than the TM and MIPT, requires somewhat complex or difficult preparation, and has regulatory concerns associated with the use of the resident tracers.
3. The MIPT is readily available and simple to implement. However, there are concerns associated with managing extracted groundwater.
4. The TCW is less available than the other methods, more difficult to implement (complex calculations and well construction), and it has some regulatory concerns.

2.5 Cost studies

The different flux measurement methods have considerable variation in cost and comparing costs between methods is not straightforward. According to the NRC (1997), for a variety of reasons, developing accurate cost comparisons is complicated and extremely difficult. The leading reasons for these difficulties are:

1. Costs reported under a set of conditions at one site are very difficult to extrapolate to another.
2. Vendors may report costs using a variety of metrics that cannot be directly compared.
3. Technology providers do not always report all costs (such as the costs for permitting and mobilization). They tend only to report the costs associated with operating the system.
4. The category of cost that is reported can vary (e.g. reporting life cycle cost versus initial capital cost).

The inherent uncertainties associated with investigating the subsurface present challenges to comparing costs. Investigating sites is expensive and site characterization and selection of a remediation technology can exceed the cost of the actual remediation (RS Means, 2003). To overcome these challenges, certain steps can be employed to help evaluate and compare technology options. One such step is to develop site templates that can be used to compare the costs of implementing different site characterization or remediation technologies (NRC, 1997). Another is to report costs in comparable units. The NRC (1997) suggests using template sites that describe a broad range of aquifer conditions. The aquifer conditions differ by varying the values of aquifer conductivity, depth, and groundwater flow rate. Additionally, the methods can be applied to plumes of different widths. By applying a site characterization technology under the different conditions, a range of costs is obtained (NRC, 1997). This basic approach to cost estimating and reporting is also described in Nielson (2006), who uses three separate "Unit Model Scenarios" to evaluate technology performance and cost.

As noted in earlier sections, Kim (2005) used a template site consisting of a confined sand aquifer contaminated by a 200 m wide by 10 m deep plume to compare the costs of implementing various flux measurement technologies. He did not consider common costs shared among the technologies, so the estimates are relative, not absolute. The results of the analysis are shown in Table 2.6.

Table 2.6. Costs of applying various flux measurement methods to characterize a template site (after Kim, 2005; Goltz et al., 2007b)

<table>
<tr><th colspan="2">Method / approach</th><th>Relative Cost</th></tr>
<tr><td colspan="2">TM with Pump test</td><td>$157,000</td></tr>
<tr><td colspan="2">PFM</td><td>$155,000</td></tr>
<tr><td colspan="2">IPT</td><td>$123,000</td></tr>
<tr><td rowspan="2">TCW</td><td>Multi-dipole approach</td><td>$84,000</td></tr>
<tr><td>Tracer test approach</td><td>$92,000</td></tr>
<tr><td colspan="2">MIPT</td><td>$104,000</td></tr>
</table>

As discussed in Section 2.3.2.2, the ESTCP (2007) report compared costs of the PFM and the TM with the BDT at a template site. Since both methods required installation of the same set of wells, well installation costs were omitted from the analysis. Detailed deployment (equipment, training, and planning) and operation costs for the PFM and TM methods are reported in Tables 2.7 and 2.8, respectively. It was determined that total costs per linear foot were $303 for the PFM and $430 for the TM with the BDT.

Table 2.7. Detailed costs for PFM deployment* (after ESTCP, 2007)

COST CATEGORY	Subcategory (10 wells – 100 linear ft)	Costs ($)
FIXED COSTS		
CAPITAL	Operator Training—for passive flux meter installation and sampling.	$500
COSTS	Cost of $2,500 per person. Amortize over 10 deployments.	
	Planning/preparation (assume 8 hours, $80/hr) Organizing supplies, site access, deployment duration, sorbent/tracers selection and approval	$640
	Equipment: Sorbent preparation mixing equipment and PFM packing equipment ($10,000 capital) amortize over 10 major deployments	$1,000
	Environmental safety training ($1,000/yr/person). Amortize over 10 deployments for two people	$200
Subtotal $2,340		
VARIABLE COSTS		
OPERATING COSTS	Operator labor—two people required to construct and install passive flux meters and to collect, prepare, and ship samples. One day for deployment and a second day for retrieval. (8hr/day X 2 people X 2 days X $80/hr)	$2,560*
	Mobilization/demobilization—assumes two trips to and from the site, each requiring 0.5 days of travel plus travel costs for two people. $80/hour labor, air fare, travel costs up to ~$800 per person.(4 trips X 4hrs/trip X 2 people X $80/hr +4 X ~$800)	$5,760*
	Hotel for 2 people for 2 nights during PFM deployment and 2 nights during PFM retrieval assuming $150/night per diem. (4 nights X 2 people X $150/night)	$1,200
	Raw materials—sorbent and resident tracers ($166.70/well)	$1,667
	Consumables, supplies—sorbent, socks, ancillary components of the PFM, and sample vials ($183.33/well)	$1,833
	Residual waste handling—consumed sorbent and socks ($333.33/well)	$3,333
	Sampling and analysis for contaminants and resident tracers retained on passive flux meter sorbent ($100/sample or $1,000/well)	$10,000*
Subtotal		$26,353
OTHER COSTS	Data analysis—6 hours required ($160/well)	$1,600
Subtotal		$30,293
TOTAL TECHNOLOGY COST		$30,293
Unit cost per linear foot (ft)		$303/ft

*Capital cost of well installation not included.

Table 2.8 Detailed costs for TM with BDT deployment* (after ESTCP, 2007)

COST CATEGORY	Sub Category (10 MLS – 100 samples)	Costs ($)
IXED COSTS		
CAPITAL COSTS	Operator Training for BDT ($5,000). Amortize over 10 sampling events	$500
	Planning/Preparation (assume 8 hours, $80/hr)—organizing supplies, site access, deployment duration, sorbent/tracers selection, and approval	$640
	Equipment—borehole dilution, MLS sampling equipment, and PFM packing equipment ($5,000). Amortize over 10 sampling events.	$500
	Environmental safety training ($1,000/yr/person) Amortize over 10 sampling events.	$200
Subtotal $1,840		
VARIABLE COSTS		
OPERATING	Operator labor—two people are required to sample the MLS network 15 min per sample per person (100 samples X 1/4 hr X $80/hr) or ($200/well)	$2,000*
COSTS	Mobilization/demobilization—assume 1 trip to site each 0.5 days of travel plus travel costs for two people. $80/hour labor, air fare, travel costs up to ~$800 per person. (2 trips X 4 hrs X 2 people X $80 +2 X ~$800)	$2,880*
	Hotel for 2 people for 16 nights for BDTBDTs assuming $150/night per diem. Total costs = (number of nights in a hotel X $150/night). Number of nights in a hotel = (2+number of wells X 1.4 days of BDT/well) X 2 people. For 10 wells, this is 16 nights. Thus, (16 nights X 2 people X $150/night)	$4,800
	Conduct BDTBDTs at 100 locations. Each test requires approximately 2 hours. (100 locations X 2 hrs X $80/hr or $1,600/well)	$16,000
	Consumables, supplies—sample vials, gloves, tracers ($66.70/well)	$667
	Residual Waste Handling Purge water for MLS sampling ($333/well)	$3,333
	Sampling and analysis for contaminants in water samples ($100/sample)	$10,000*
Subtotal		$39,680
OTHER COSTS	Data analysis. ($160/well)	$1,600
Subtotal		$43,120
TOTAL TECHNOLOGY COST		$43,120
Unit cost per linear foot (ft)		~$430/ft

*Capital cost of well installation not included.

The cost breakdowns in Tables 2.7 and 2.8 support the evaluation of PFM and TM costs reported in Tables 2.4 – 2.6; that is, that the costs are very similar. Especially note that the TM cost reported in Table 2.8 assumes an extremely high resolution of BDT measurements. If the costs for the BDTs are reduced, by reducing the resolution of the BDTs, the overall cost difference between the TM and PFM methods will also become significantly less. Not available in the literature is a comprehensive comparison of all four flux measurement methods that includes estimation of capital and operating costs over a variety of site conditions.

III. Methodology

3.1 Introduction

The purpose of this chapter is to present a methodology to objectively compare flux measurement method costs, as well presenting an approach that may be used to assist a remedial site manager in deciding which flux measurement method is most appropriate for use at his or her site. In Section 3.2, the assumptions used in the cost analysis are presented. The details of the cost analysis methodology itself are discussed in Section 3.3. Finally, in section 3.4, we present the approach to developing a decision tree that may be used by stakeholders to select an appropriate flux measurement method based on site characteristics and management goals.

3.2 Cost Analysis Assumptions

Comparing technologies is difficult for a variety of reasons. An underlying problem is that sites may have very different geological conditions and contaminant distributions, resulting in widely varying site characterization costs. Other issues are the metrics used to report costs and the sources from which cost data are derived (NRC, 1997). Ultimately, to compensate for varying site conditions, cost reporting methods, and data sources, it is necessary to develop uniform conditions for comparison of the different technology costs. When possible, costs common to each of the methods will be assumed equal and excluded from the analysis. Costs associated with such things as a new road for site access or an on-site facility for communications, equipment assembly, and sample packaging would fall into this category. It is also assumed that the contractor implementing the method is local and perdiem will not be necessary. When applicable, innovations that can reduce labor hours, such as auto sampling and remote monitoring, will be included in the analyses. Initial site investigation will be considered accomplished and data collected prior to selecting or implementing a flux measurement technology. Data such as aquifer depth, plume size (required transect width), and source characteristics will be generally defined. These data would also include an estimate of aquifer heterogeneity, hydraulic conductivity, hydraulic gradient, source strength and groundwater flow magnitude and direction.

In the cost analysis, we assume that each measurement method is capable of measuring contaminant mass flux at the site under consideration and is not excluded due to any regulatory or site constraints. Costs will be estimated for application of each method at various template sites. The site templates are designed to provide a range of conditions, by varying the following:

1. Size of the control plane, which is a function of the aquifer depth and plume width.
2. Required sampling resolution, which is driven by management objectives and/or site heterogeneity.
3. Hydraulic conductivity, which may affect test duration and the volume of aquifer that's interrogated and water that's extracted.

It is anticipated that depending on conditions, the costs of point and volume-averaged methods will be very different. Thus, a survey of the methods using various template sites will serve to give stakeholders a better picture of which method may be most applicable to their specific management objectives and site conditions.

3.3 Cost Analysis Methodology

3.3.1 Development of template sites

The template sites will enable an evaluation of costs for different measurement methods under varying site conditions. Applying the site templates suggested by the National Research Council (1997) and Nielsen (2006) will allow for cost analyses to be done over a range of site conditions. The NRC suggested template sites are shown in Table 3.1.

Table 3.1 Template Site Characteristics (after NRC, 1997)

Template Number	Depth to Water Table (m)	Aquifer Thickness (m)	Hydraulic Conductivity (cm/sec)	Ground Water Flow Rate (m/year)
1	4.6	7.6	1.90E-03	3
2	4.6	7.6	9.50E-02	150
3	4.6	21	1.90E-03	3
4	4.6	21	9.50E-02	150
5	7.6	7.6	1.90E-03	3
6	7.6	7.6	9.50E-02	150
7	7.6	21	1.90E-03	3
8	7.6	21	9.50E-02	150
NOTE: Soil porosity is assumed to be 25 percent, and hydraulic gradient is assumed to be 0.005 cm/cm for all eight cases.				

From Table 3.1 and Nielsen (2006), 16 different templates were built to encompass a range of differing site and contaminant conditions. These templates are detailed in Table 3.2. Having multiple template sites allows a site manager to better relate his or her site characteristics to one of the template sites. For simplicity, all template sites are assumed homogeneous and isotropic, with an effective porosity of 0.25, hydraulic gradient of 0.005 cm/cm, and depth to the water table of 4.6 m. Fractured consolidated systems were not considered in developing the templates as this is a special circumstance requiring site-specific analysis (NRC, 1997).

Table 3.2, Template sites used in this study

Template Number	Depth of Aquifer (m)	Width of plume (m)	Control plane Area (m^2)	Hydraulic Conductivity (cm/sec)	Darcy Velocity (m/s)
1	7.6	7.6	57.76	1.90E-03	9.51E-08
2	7.6	15.25	115.9	1.90E-03	9.51E-08
3	7.6	30.5	231.8	1.90E-03	9.51E-08
4	7.6	45.7	347.32	1.90E-03	9.51E-08
5	22.86	7.6	173.736	1.90E-03	9.51E-08
6	22.86	15.25	348.615	1.90E-03	9.51E-08
7	22.86	30.5	697.23	1.90E-03	9.51E-08
8	22.86	45.7	1044.702	1.90E-03	9.51E-08
9	7.6	7.6	57.76	9.51E-02	4.76E-06
10	7.6	15.25	115.9	9.51E-02	4.76E-06
11	7.6	30.5	231.8	9.51E-02	4.76E-06
12	7.6	45.7	347.32	9.51E-02	4.76E-06
13	22.86	7.6	173.736	9.51E-02	4.76E-06
14	22.86	15.25	348.615	9.51E-02	4.76E-06
15	22.86	30.5	697.23	9.51E-02	4.76E-06
16	22.86	45.7	1044.702	9.51E-02	4.76E-06
NOTE: Soil porosity is assumed to be 25 percent, and hydraulic gradient is assumed to be 0.005 cm/cm for all 16 cases.					

3.3.2 Reporting metrics

In addition to the template sites, the NRC (1997) recommends that costs be reported using a standardized format. For the flux measurement methods, well installation, initial deployment costs, and operation and labor costs are reported per event; that is, per measurement of flux at all the sampling points/volumes deemed necessary to characterize a site. This division in cost categories was chosen as the best means to provide a potential technology user useful data for flexibility in forecasting expenses associated with implementing the technology.

Well installation costs include the costs associated with mobilization of drilling rigs, labor, disposal fees, soil sampling and analysis, etc. (Herriksen and Booth, 1995). Table 3.3

reflects those costs categorized as well installation costs. Costs common to all methods, such as those associated with drilling permits, site access, and removal of obstructions were not considered, as these costs are assumed to be common to all methods. For each of the methods, aquifer depth, plume width, and the required number of wells are important cost drivers, and they will vary significantly for the different site templates.

Table 3.3 Well installation cost categories (after Herriksen and Booth, 1995)

Item
Drill Rig
Grouting
Sampling
Lab/Chem Analysis
Lab/physical Analysis
Mobilization-demobilize
Standby Labor
Decontamination labor
Drilling waste disposal
Drilling waste disposal (solid)
Drilling waste disposal Water)

Deployment costs include the expense of purchasing equipment, pre-deployment planning and packaging of equipment, and training on safety and environmental issues. Deployment costs will include those costs that are not included as well installation costs or part of the operation costs. Deployment costs include estimates for all specialized equipment and training. A stakeholder intending to implement a new technology will thus be aware of specialized capital equipment purchases that might be required. If the equipment is already available, those costs can be deducted from the overall cost estimate.

The operation costs include costs for sample testing, raw materials, waste handling, consumed supplies, labor for site monitoring, and specific method operation requirements. This cost will provide a potential technology user an estimate of costs associated with conducting the method for one test episode. The labor for all monitoring, data analysis, and general site labor is assumed to be $80 dollars per hour for all labor types. Raw material costs only pertain to the PFM where the sorbent material is considered a raw material. Not included in operation costs is electrical power. This cost is highly variable due to such factors as location and hydrogeological conditions (depth to the water table, hydraulic conductivity). Additionally, in comparison with other costs, power costs are assumed to be relatively small.

3.3.3 Application of measurement methods to the template sites

The cost to apply a flux measurement method to a particular template site depends on more than just the site parameters listed in Table 3.2. In particular, the cost of method application will depend greatly upon site heterogeneity as well as upon the management objective in making the measurement. For instance, if flux measurements are being made in a heterogeneous system with the objective of locating source zone "hot spots" (zones of high contaminant concentration or flux), a very high spatial resolution for sampling will be required. Conversely, for a homogeneous system, where the goal is to measure overall flux (perhaps to determine the efficacy of natural attenuation processes in reducing contaminant concentration and flux), a much lower sampling spatial resolution is needed.

In this study, three levels of spatial resolution, low, medium, and high, will be applied at the template sites. Note that the impact of spatial resolution on cost will vary, depending on whether a method is a point method or a volume-averaged method. For point methods, there is a clear relation between resolution and cost. Additionally, point methods can be installed such that resolution varies both vertically and horizontally. Variations in the resolution of volume-averaged methods, where pumping wells are installed, are practically limited to the horizontal direction. Examples of when low, medium, or high resolution sampling will be required are given below.

1. Low spatial resolution may be used for initial site characterization, or to characterize a relatively homogeneous aquifer.
2. Medium spatial resolution may be used when evaluating flux in a "typical" aquifer.
3. High spatial resolution may be appropriate when attempting to locate a source zone "hot spot" or evaluating the efficacy of a measurement method.

Quantitative definitions of the spatial resolution levels are included in Table 3.4. It was assumed that research testing of the various methods was conducted at a high spatial resolution. Therefore, the high spatial resolution values in Table 3.4 were based upon descriptions of those tests (ESTCP, 2007; USEPA, 2007). The medium and low spatial resolution values were developed based upon discussions with Wright-Patterson Air Force Base Environmental Management personnel who oversee multiple contaminated site investigations.

Table 3.4 Spatial resolution defined in m^2 of control plane per sampling point

		Range	
Spatial Resolution	**Average**	**High**	**Low**
Low	21.00	13	26
Medium	7.00	5	7.5
High	4.00	3	4.5
Values in m^2 of control plane/sampling point			

Costs that account for the specialized aspects of each of the methods will be estimated using data from various publications on the flux estimating methods (ESTCP, 2007; Brooks et al., 2007; Goltz et al., 2007b). Prices that are common to site characterization were obtained from RS Means (2006; 2007) when available.

Some equipment costs (e.g., for automated sampling and monitoring) were obtained directly from distributors, who provided costs from several manufacturers. When a range of costs was available, average costs were used in the analysis. Values for labor costs to prepare and ship samples were obtained from investigators who have conducted field tests of the technologies or from published reports.

3.3.4 Costs for each method

Each method cost was broken out into the three categories: well installation, deployment, and operation and maintenance costs. Additionally, within each category, each item was assigned a unit cost. The unit cost breakdown will enable the methods to be directly applied to the changing conditions of the 16 different site templates

3.3.4.1 TM with BDT

The transect method using the MLS is the conventional measurement method that can be used as a standard against which the other innovative methods can be compared for performance and for cost (Goltz et al., 2007b). As described in Chapter 2, the method requires the installation of multiple wells for the MLS and BDT test. The costs associated with well drilling are included as a well installation cost (Table 3.3). Labor hours for the TM/BDT method were obtained from ESTCP (2007). Note, however, that ESTCP (2007) assumed a BDT would be conducted at every MLS point. In the current study, we assume only a single BDT is conducted and the results were used to estimate the regional Darcy velocity over the entire template site. Sample costs were assumed to be $100/sample. This includes shipping, analysis, and reporting for each sample. Collection and packaging of the samples are captured in the labor costs. The unit costs used in the cost analysis of the TM/BDT method are shown in Table 3.5.

Table 3.5 TM/BDT Deployment and operation costs (after ESTCP, 2007)

	Deployment costs		
	Item	**Cost ($)**	**Unit**
1	Operator Training (TM/BDT)	$2,000	per person
2	Planning/preparation/equip shipment	$640	per episode
3	Equipment	$5,000	initial capital
	Operation Costs		
4	Sample cost	$100	sample
5	BHD test	$200	test
6	Operator costs	$80	man-hr
8	Waste handling fee	$33	per LF of well
9	consumables	$7	per LF of well

The number of wells assumed to be required for a given spatial resolution and control plane size is tabulated in Appendix A. Thus, for a given template site, using Table 3.5 and Appendix A, total costs for applying the TM method at the site can be estimated.

3.3.4.2 PFM

The installation costs of the PFM method are quite similar to the costs to install the TM. Differences between the TM and PFM costs are due to differences in the labor requirements of the two methods and the need to conduct a separate BDT (or slug or pump tests) to estimate the Darcy velocity as a component of the TM. Recall that the PFM method allows calculation of the regional Darcy velocity by analyzing the resident tracer, and no separate test is required. Note, however, that the PFM method does have higher equipment expenses and requires more specialized training. Additionally, expenses associated with the PFM's sorbent materials (categorized as raw materials) are greater than the TM costs. The unit costs used in the cost analysis of the PFM method are shown in Table 3.6.

Table 3.6 PFM Deployment and operation costs (after ESTCP, 2007)

Deployment costs			
	Item	**Cost ($)**	**Unit**
1	Operator Training	$2,500	per person
2	Planning/preparation/equip shipment	$640	per episode
3	Equipment	$10,000	per episode
Operation Costs			
5	Sample cost	$100	per sample
6	Operator costs	$80	per man-hr
7	Raw materials	$17	per- LF of well
8	Waste handling fee	$33	per- LF of well
9	Consumables	$7	per- LF of well

The number of wells assumed to be required for a given spatial resolution and control plane size is tabulated in Appendix A. The analysis of the PFM is assumed to be $100/sample. However, at the current time, the only facilities capable of performing the analysis are at the University of Florida and Purdue University. Considering this, the cost associated with analysis of the sorbent may depend greatly on where the analysis can be accomplished, what contaminant and resident tracer is being tested, and what special expertise may be required to perform the analysis.

3.3.4.3 MIPT

The unit costs used in the cost analysis of the MIPT method are shown in Table 3.7.

Table 3.7 MIPT Deployment and operation costs

	Item	Cost ($)	Unit
	Deployment Costs		
1	Operator Training (MIPT)	$1,000	per person
2	Planning/preparation/ Equip shipment	$1,280	per episode
3	Equipment (Automated Monitoring)	$1,537	per well
4	Equipment (Pumping/sampling)	$1,172	per well
5	Tubing	$3	per lf well
	Operation Costs		
6	Site setup labor	9.6	hr/well
7	Labor-Sampling	1.25	hr/well/5xday
8	Labor-flow monitoring	1.25	hr/well/5xday
9	Sample pack/ship	4	hr /day
10	Consumables, supplies	$200	per episode
11	Waste handling—contaminated water disposal	$264	per m^3
12	Sampling analysis-contaminants / tracers	$100	per sample

An increase in the number of pumping wells will allow for better horizontal (though not vertical) resolution. In addition, depending on aquifer conductivity, more wells may be needed to ensure plume capture. Of course, the greater the number of wells, the greater the capital costs to install the MIPT method. The number of wells assumed to be required for a given spatial resolution and control plane size is tabulated in Appendix A.

To determine the volume of extracted water, two methods were used. The first method involves calculating the pump rate and duration required to capture the entire plume width. This was dependent on the number of wells used, the spacing of the wells, and the hydrologic properties of the aquifer. Assuming the duration of the test would be five days, pump rates can be determined from the hydrologic properties specified by the site templates. Results of this method were within 1% of the results calculated using the second method. The second method determined the volume extracted for a typical MIPT method application; pumping for 24 hours at each of five predetermined rates. Extracted water disposal costs vary depending on proximity to a treatment facility, permit requirements if any, and the particular contaminant. For the purpose of this study, it will be assumed that the cost of contaminated water disposal is \$1/gal or \$264/m^3. This cost includes the cost for onsite storage, transportation, treatment, and disposal. The value of \$1/gal is a conservative estimate, encompassing costs for treating and disposing of most non-radioactive materials. USEPA (2008) reports that they will fund up to \$0.72/gal for disposal, but this cost does not take into account the costs for on-site storage.

3.3.4.4 TCW

The TCW method is similar to the MIPT method in that it is a method that obtains a volume-averaged measurement. The key advantage of the TCW method over the MIPT method is that the TCW method does not require extraction of contaminated water from the subsurface, thereby avoiding treatment, transport, and disposal costs. The costs for the TCW in Table 3.8 were largely obtained from a report that included a cost estimate for use of TCWs to treat contaminated groundwater (AFRL, 1998). The unit costs used in the cost analysis of the TCW method are shown in Table 3.8.

Table 3.8 TCW Deployment and operation costs

	Deployment Costs		
	Item	**Cost**	**Unit**
1	Operator Training (TCW)	$ 1,000	per person
2	Planning/preparation/ Equip shipment	$ 1,280	per episode/well set
3	Flow sensors and controllers	$ 3,750	per episode
4	Static Mixers	$ 1,446	per episode
5	Pumps and ancillary Equipment	$ 13,439	per episode
6	Tubing and connectors	$ 2,404	per episode
7	Valves and fittings	$ 1,165	per episode
	Operation Costs		
6	Site setup labor	9.6	hr/well
7	Labor-Sampling	1.25	hr/well/5xday
8	Labor-flow monitoring	1.25	hr/well/5xday
9	Sample pack/ship	4	hr /day
10	Consumables, supplies	$200	per episode
11	Sampling analysis-contaminants	$100	per sample
12	Sampling analysis- tracers	$85	per sample

To apply this technique under the conditions at the template sites, it is necessary to determine the maximum pumping rate (Q_{max}) that can be sustained for the specified hydraulic conductivity (K) and aquifer thickness (B) at the site. Q_{max} was calculated using equation 3.1 (Bear, 1979):

$$Q_{\max} = \frac{2\pi KBS_w}{\ln\left(\frac{R}{r_w}\right)} \qquad (3.1)$$

where r_w is the radius of the TCW, R is the radius of influence, and S_w is the maximum allowable drawdown (Mandalas et al, 1998). For this study, the r_w is assumed to be 0.1 m (for an 8 inch well), S_w is 1/3 of the aquifer thickness, and R is set equal to $3000{*}S_w{*}K^{0.5}$ (R in meters and K in m/sec) (Bear, 1979). For the specified aquifer conductivity and thickness, pumping at a rate Q_{max} or less ensures that the maximum allowable drawdown will not be exceeded. Equation 3.1 only accounts for extraction at the TCW, not injection. Thus, the calculation is conservative, as the injection serves to minimize drawdown. Equation 3.2 makes use of the Q_{max} calculated in equation 3.1 to determine the minimum numbers of well pairs.

$$\frac{N}{2} = \frac{KBi(PW)(1+f)}{Q_{\max}} \qquad (3.2)$$

where $N/2$ is the number of well pairs for single extraction and injection wells, PW is the capture zone width, Q_{max} is the max allowable pump rate, and f is the fraction of recycle between wells. From the Q_{max} calculated for the conditions in the different templates, it was determined that only a single pair of wells is required to meet the template capture zone widths.

To calculate the duration of run time for the TCW test, equation 3.3 is used to calculate the minimum time for a tracer to move from one TCW to the other (Kim, 2005; Cunningham et al., 2004):

$$t_{\min} = \frac{4}{3}\pi\frac{a^2 H n_e}{Q} \qquad (3.3)$$

where a, H, n_e, and Q represent the half distance between the injection/extraction wells, length of screened section of wells, aquifer porosity, and the well pumping rates, respectively. To determine the time required to reach steady-state, t_{min} will be multiplied by 20. This approximates the time for the tracers to reach steady-state at the extraction screens (Kim, 2005). For this study the variables a and H were held to 16.5% of the plume width and 12% of the aquifer thickness, respectively, across multiple site templates. This is consistent with Kim's (2005) experiment design. In addition, the well pumping rate, Q, was held constant at $\frac{1}{2}Q_{max}$.

3.4 Decision tree development

We want to develop a decision tree that will serve as a “user-friendly” tool that can be used to facilitate information transfer to potential technology users. Development of a chart that requires only initial site investigation data and management objective criteria to determine one or two preferred flux measurement methods will greatly facilitate the decision making process. The decision tree will be formatted as a flow chart. With an input of management objectives and initial site estimates of control plane size, required spatial resolution of the measurements, and hydraulic conductivity, the decision tree will output the two most economical methods of flux measurement for application at the site. The user can then chose one of the two most economical methods based on other subjective factors that have been discussed in this study (e.g., regulatory acceptability, implementability).

IV. Results and Discussion

4.1 Introduction

In section 4.2, we will use data obtained from the literature review on the performance, advantages, and limitations of the various flux measurement methods to present findings on performance and ease of implementation. The costs associated with measuring mass flux using the different methods at the various template sites will be presented in section 4.3. In Section 4.4, we will develop a decision tree and discuss the results obtained from application of the decision tree methodology.

4.2 Published Data

4.2.1 Performance

To various degrees, all the innovative measurement methods have been tested and evaluated. From the review in sections 2.3 and 2.4, we may conclude that all the methods have the potential to effectively measure contaminant mass flux and are viable candidates for commercial application. Table 4.1 displays comparable information on laboratory and field test data.

Table 4.1 Summary of field and laboratory evaluations of accuracy for the different flux measurement methods

Method	Laboratory tests	Field test locations	Compared to: Known flux	Compared to: Transect method	Result difference	Reference
TM	Boxed aquifers	CFB Borden	X		11-40%**	ESTCP, 2007
TM	Meso- scale	New Zealand lab	X		(-33) to 50%	Kim, 2005
PFM	Boxed aquifers	CFB Borden	X		9-32%**	ESTCP, 2007
PFM		CFB Borden		X	13-18%	Annable et al., 2005
PFM		Hill AFB		X	2-17%**	Brooks et al., 2007
PFM		Ft Lewis		X	16-300%*	Brooks et al., 2007
PFM		NASA's LC-34		X	17-190%*	ESTCP, 2007
MIPT		Hill AFB		X	3-81%**	Brooks et al., 2007
MIPT	Meso- scale	New Zealand lab	X		36-60%**	Yoon, 2006
MIPT		Ft Lewis		X	13-17%**	Brooks et al., 2007
TCW-T	Meso- scale	New Zealand lab	X		15-44%*	Goltz et al., 2007a
TCW-MD	Meso- scale	New Zealand lab	X		< 2%**	Goltz et al., 2007a

* indicates method being evaluated overestimates known or TM-measured flux
** indicates method being evaluated underestimates known or TM-measured flux

In table 4.1, where the various methods were used to measure a known flux, we see that the PFM, MIPT and TCW methods have accuracies comparable to the TM. As the TM is generally considered to provide "acceptable" results, from a regulatory point of view, it may be argued that the accuracy of the other more innovative methods should also be acceptable. We also see from the literature review that each method has its own distinct characteristics that may be advantageous (or not) to the stakeholder under particular circumstances.

4.2.2 Advantages and limitations

From section 2.3, the advantages and limitations of the different methods can be consolidated into the following list. We will later use these characteristics to develop the decision tree that will guide selection of a measurement method for given management objectives and hydrogeologic conditions.

Transect Method

- Advantages
  - Able to provide high resolution definition of contaminant plume in vertical and horizontal directions
  - Low costs under homogeneous conditions where high resolution is not required
  - Ease of use, simplicity of calculations, and short duration
  - Flexibility--additional wells can be added to locate "hot spots"
  - Minimal waste disposal
- Limitations
  - Requires additional tests to estimate flux (BDT to estimate q_0 or pump/slug test to estimate hydraulic conductivity along with piezometer soundings to estimate hydraulic gradient)
  - High costs under heterogeneous conditions that require high resolution
  - Point sampling in time which can't account for temporal hydrologic changes (e.g. rain events, seasonal variation)
  - Potential to miss "hot spots" if sampling resolution is insufficient

Borehole Dilution Test (BDT)

- Advantages
  - Time-averaged measurement of q_0 at a point in space
  - Low costs under homogeneous conditions where high resolution is not required
  - The test is of relatively short duration and can be applied under a variety of soil conditions from unconsolidated to fractured consolidated material
- Limitations
  - Very sensitive to divergence and convergence factor (α), which is difficult to quantify
  - In the cases where low permeability and high porosity media are present, diffusion can dominate and must be accounted for
  - Requires the use of tracers which may raise some regulatory concerns
  - Must be done in conjunction with another test (typically the transect method) to measure concentration

Passive Flux Meter

- Advantages
  - Time averages both concentration and q_0
  - Able to provide high resolution definition of contaminant plume in vertical and horizontal directions
  - Can also provide the direction of groundwater flow in addition to groundwater flux magnitude
  - Doesn't require electrical power on site, even for deep aquifers; attractive for austere conditions
- Limitations

- Very sensitive to divergence and convergence factor (α), which is difficult to quantify
- Sensitive to biological activity which can degrade resident tracers (may be especially problematic if engineered bioremediation is being used as a cleanup remedy)
- Requires special knowledge to deploy method and interpret results
- Deployment of method requires weeks
- Requires the use of tracers which may raise some regulatory concerns

Tandem Circulating Wells

- Advantages (Tracer)
    - Volume averages both concentration and hydraulic conductivity measurements
    - Avoids the need to extract and treat contaminated groundwater
    - Can be conducted for a long time period, allowing for both temporal and spatial averaging
    - Can interrogate deep plumes (does not require pumping to surface)
    - For high hydraulic conductivity systems, small number of wells can interrogate wide plumes
- Limitations (Tracer)
    - May require a relatively long and expensive tracer test (days-weeks)
    - Does not provide the spatial resolution of the point methods, which may be required for site characterization
    - Calculations for determining the flux can be rather complex
    - Has not been field tested
    - Requires the use of tracers which may raise some regulatory concerns
    - If contaminant is primarily in the upper or lower region of the aquifer, groundwater circulation has the potential to spread contamination
- Advantages (Multi- dipole)
    - Potential for very short test durations
    - Volume averages both concentration and hydraulic conductivity measurements
    - Avoids the need to extract and treat contaminated groundwater
    - Can be conducted for a long time period, allowing for both temporal and spatial averaging
    - Can interrogate deep plumes (does not require pumping to surface)
    - For high hydraulic conductivity systems, small number of wells can interrogate wide plumes
- Limitations (Multi-dipole)
    - Small head differences may be difficult to measure
    - Does not provide the spatial resolution of the point methods, which may be required for site characterization
    - If contaminant is primarily in the upper or lower region of the aquifer, circulation pattern has the potential to spread the contamination

Modified Integral Pump Test

- Advantages
  - Volume averages concentration and q_0 measurements
  - Variable pumping rates allow flexibility in accommodating seasonal flow events as the wells within the transect can be pumped at varying rates to accommodate changing conditions
  - Flexible in that additional wells can be placed without incurring large costs
  - Duration of test is relatively short (about a week)
  - Easy calculations
- Limitations
  - Extracts groundwater and therefore requires the treatment of contaminated water
  - May be difficult to measure small head differences
  - Does not provide the spatial resolution of the point methods, which may be required for site characterization, although emplacement of additional wells will allow for increased resolution horizontally

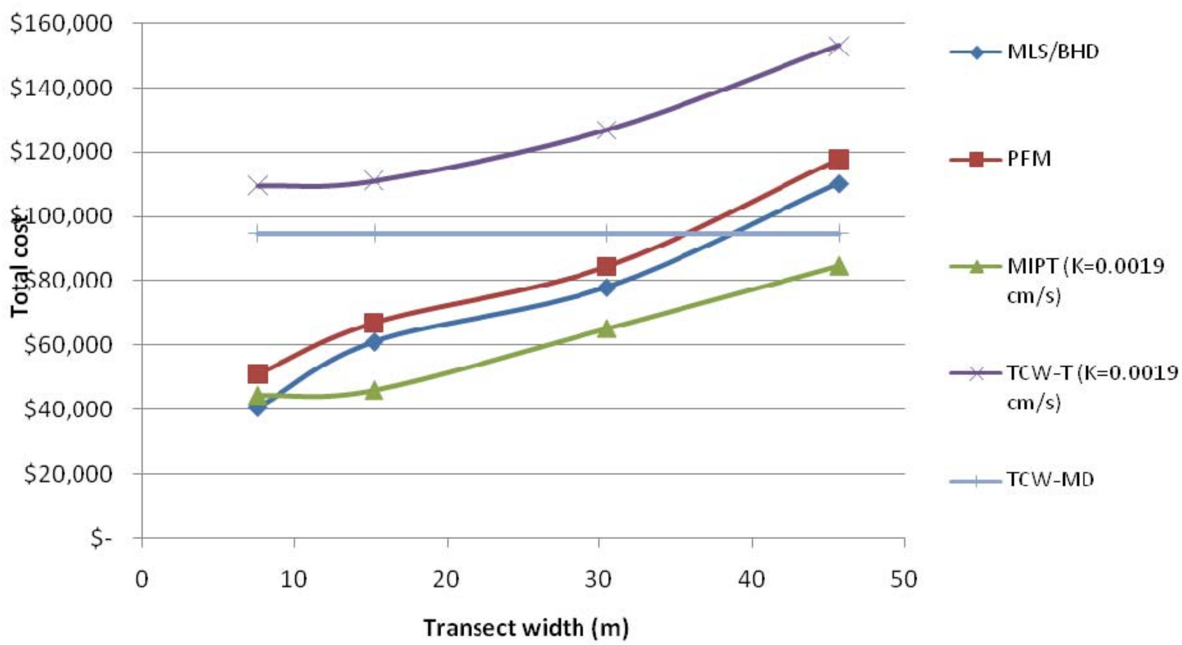
4.3 Cost Comparison

4.3.1 Cost versus size of control plane

Holding the hydraulic conductivity, depth of aquifer, and spatial resolution constant, a cost versus transect width plot shows how cost increases with increasing control plane width (Figure 4.1). This analysis was conducted for all three spatial resolutions (low, medium, and high). In addition, the analysis was conducted for different aquifer thicknesses. As would be expected, costs generally increased with an increase in the width of the control plane or aquifer thickness.

Cost increased for the point methods as width increased, as these methods require more wells and sampling points for a given spatial resolution (Figure 4.1). This is not surprising, as we assume that to obtain the same spatial resolution, these two point methods require the same number of wells and the same number of samples. However, there are cost differences between the two methods, though these differences lead to similar cost increases as transect width increases. At increased transect width, the TM incurs a relatively small additional cost for adding sampling points, but the increased labor costs associated with additional sampling are significant. Conversely, the PFM incurs relatively small labor costs, but significant costs for materials, when the transverse width increases.

Figure 4.1 Cost comparisons of methods with varying transect width for 21.3m thick aquifer at low resolution



From figure 4.1 we see that nearly all method increase in cost as the transect width increases. Only the TCW-MD remains unaffected by the increase in control plane size. This is a result of the TCW-MD technique is conducted for a predetermined duration and therefore no additional costs for labor or testing.

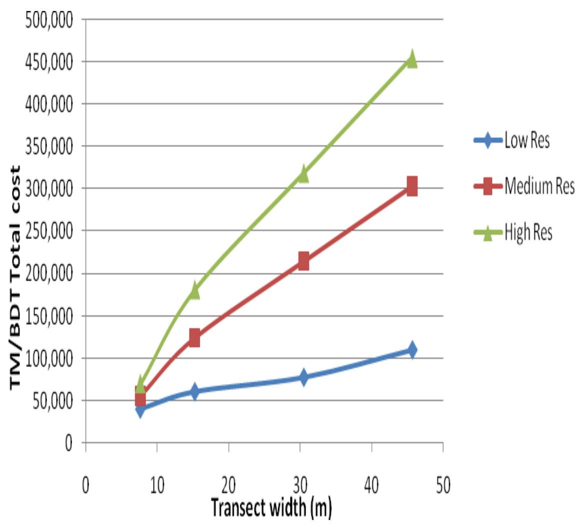
The MIPT costs also showed an increase with increasing transect width. The increase in MIPT costs were a direct result of the costs associated with treating and disposing of the extracted groundwater. This assumes that any increase in transect width or aquifer depth will increase the control plane cross-sectional area and therefore increase the total discharge that must be captured. The cost for treatment and disposal of the contaminated water was estimated at \$1/gal or \$264/m^3. An increase or decrease in this treatment and disposal cost will directly affect the slope of the MIPT cost versus transect width plot. Results showing MIPT cost versus transect width for medium spatial resolution can be seen in figures 4.6 and 4.7. Costs for low and high spatial resolutions are reported in appendix A

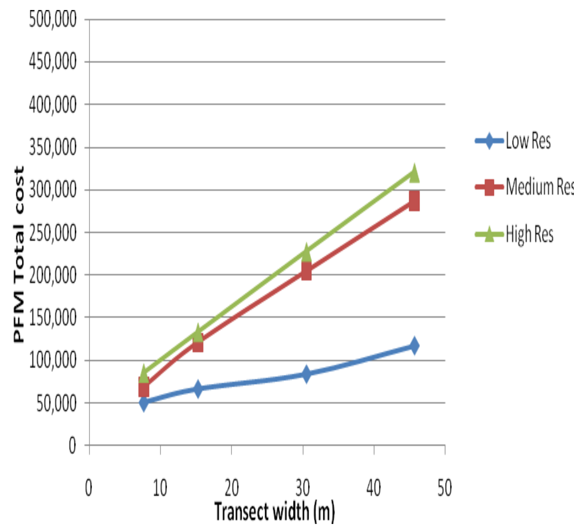
For the TCW method, assuming one well pair is sufficient to implement the method, an increase in transect width has only a minimal increase in costs. The small cost increase is directly associated with the increased labor needed to conduct a longer duration tracer test, as a result of the greater distance between the wells. If an additional set of wells must be installed, costs will increase based on the additional costs associated with installing and operating the additional wells. For the 16 template sites considered in this study, only a single set of wells was required. Results showing TCW cost versus transect width for medium spatial resolution can be seen in figures 4.6 and 4.7. Costs for low and high spatial resolutions are reported in appendix A.

4.3.2 Cost versus required spatial resolution

Application of each method was evaluated at the 16 template sites for each of the three required spatial resolutions: low, medium, and high. Costs associated with the PFM and transect methods for the template sites are shown in appendix A. As noted earlier, both of the two point methods have essentially the same costs at low and the medium resolutions, though at high resolution, the PFM is more economical (see Figure 4.2).

Figure 4.2 Point method costs for 21.3m thick aquifer of varying transect length

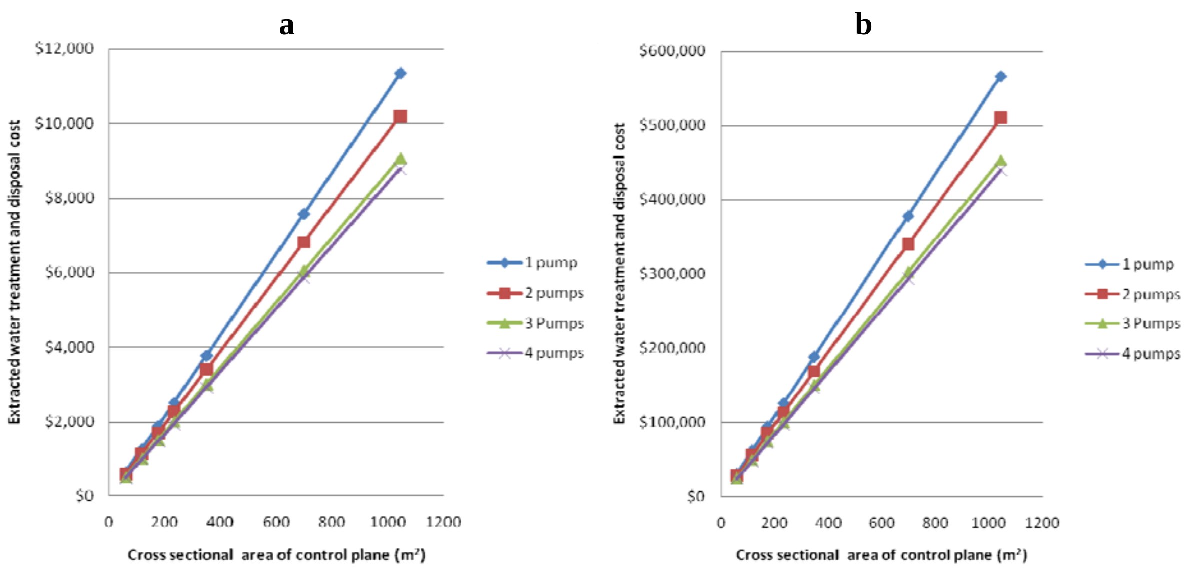




The difference in costs between the two methods at high resolution can be attributed to the increased labor costs associated with the sampling requirements of the TM. While both methods require increased sample analysis at high resolution, the labor costs of collecting samples are much greater for the TM than for the PFM method. Overall, the cost for implementing the TM using the BDT (at low and medium resolution) is nearly identical to the PFM for the same transect widths. Only at high resolution does one become more economical than the other.

The volume-averaging methods offer only limited ability to increase spatial resolution. For the template sites, doubling the resolution through the installation of an additional set of TCW wells could add an additional 40-90% to total cost. However, the MIPT offers the flexibility to add more wells to the transect with limited additional cost. Spacing extraction wells more closely along the transect will incur more capital and installation costs, but these added costs will be reduced by cost savings in disposal. The more pumps along the transect, the less volume of water needs to be extracted to capture the plume. The reduction in disposal cost for the 16 different site templates for the addition of a single well ranged from 5-10% with efficiency effects decreasing with increase of total number of wells. The addition of more wells along the transect will permit greater horizontal resolution. Vertical spatial resolution would not be increased for fully screened wells, such as are used in the MIPT method. Figure 4.3 shows the influence on cost of adding wells to a control plane for the MIPT.

Figure 4.3 Effect of additional MIPT wells on cost of treating and disposing extracted water (a) K = 0.002 cm/s and (b) K = 0.095 cm/s (<u>note different y-axis scales</u>)



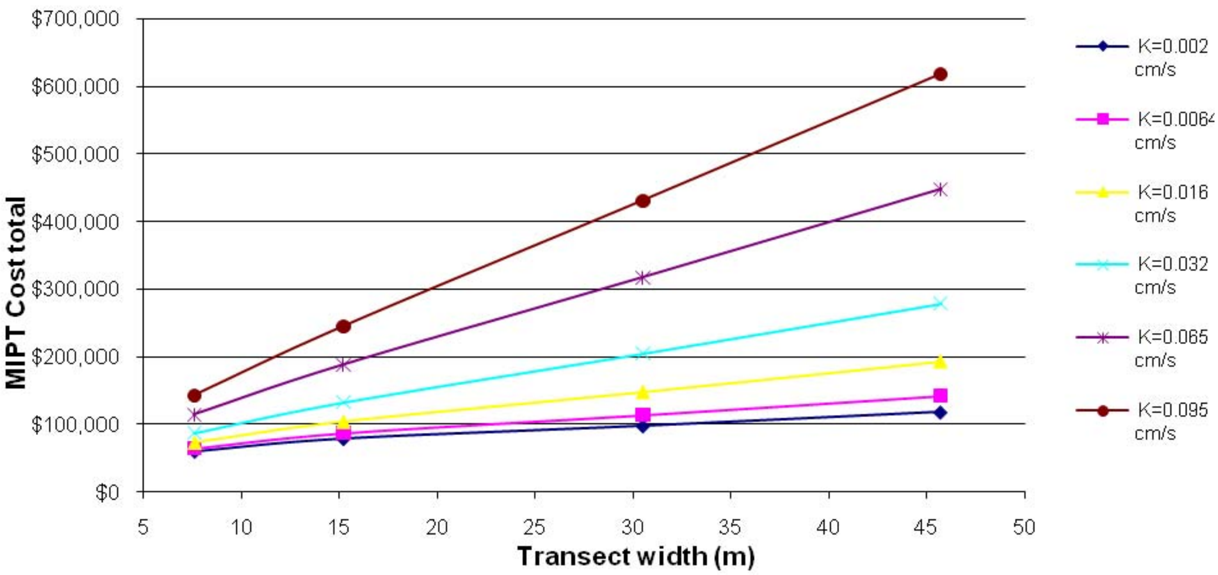
Where well installation to depths of 12 and 26 meters is roughly \$4,000 and \$9,000 per well, respectively, it seems from Figure 4.3 that only at higher hydraulic conductivities and larger control plane areas is the costs savings obtained by lower water treatment and disposal costs great enough to compensate for the cost of installing additional wells. At lower hydraulic conductivities and smaller control plane areas, stakeholders will have to balance the benefit of increasing horizontal resolution with the costs incurred from installing additional wells.

4.3.3 Cost versus hydraulic conductivity

The hydraulic conductivity values used at the template sites range from 0.002 and 0.095 cm/s. The site hydraulic conductivity has little to no effect on the point methods other than the length of time needed when applying the BDT and the PFM time-averaging methods. Note that for the PFM method, no additional costs are incurred at lower conductivities, since there are no labor costs associated with leaving the PFMs in the ground longer. There are labor costs associated with running longer BDTs at low conductivity sites, but these costs are relatively small (0.05-0.4% of total costs).

The influence of hydraulic conductivity on the costs of applying the two volume-averaging methods is significant. For the MIPT method, the cost for conducting flux measurements increases with the increase in conductivity. This is because at a higher conductivity, more water needs to be extracted (and treated) for plume capture. Figure 4.4 shows an example of the increased costs required to conduct an MIPT that result from an increase of hydraulic conductivity.

Figure 4.4 Total cost for measuring contaminant flux for a 21.3m thick aquifer at different values of hydraulic conductivity



Interestingly, TCW tracer technique costs decrease with increasing conductivity. This is because a significant fraction of the cost of implementing the TCW method is the labor and analytical costs associated with conducting the tracer test. The increase in conductivity will result in a shorter tracer test (see Equations 3.1 and 3.2). With labor at $80/hour and duration ranging from days to weeks, the cost savings for the TCW tracer technique at a high conductivity site can be considerable. For the TCW multi-dipole technique, cost will generally not change with changing hydraulic conductivity. However, high hydraulic conductivity will result in smaller head differences which may produce inaccuracies. Figure 4.5 compares costs of applying the TCW tracer technique at high and low conductivity sites for two aquifer thicknesses.

Figure 4.5 Influence of hydraulic conductivity on TCW-T costs for (a) 7.6m and (b) 21.3 m thick aquifers

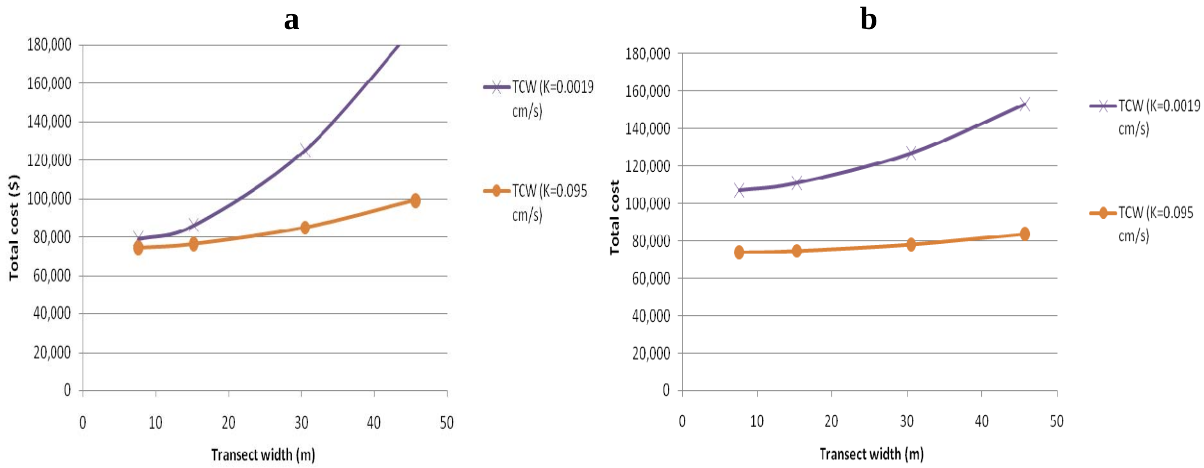


Figure 4.5 indicates that for thicker aquifers, costs are less than for thin aquifers as the transect widths increase. This is primarily due to the higher pumping rates (because of longer screens) in the thicker aquifer, which results in shorter tracer tests. Especially at larger transect widths, the cost of the tracer test dominates, so the effect of the thicker aquifer is significant.

4.3.4 Cost comparison for all methods

Comparisons were done for all methods at the low, medium, and high spatial resolutions. The comparisons are displayed graphically in Figures 4.6 and 4.7 for low spatial resolution. Costs for the TCW tracer technique and the MIPT, which are dependent on hydraulic conductivity, are displayed at both the high and low values for K. This creates a range of possible costs associated with implementing the methods.

Figure 4.6 Costs for all measurement methods versus transect width at low spatial resolution (7.6m thick aquifer)

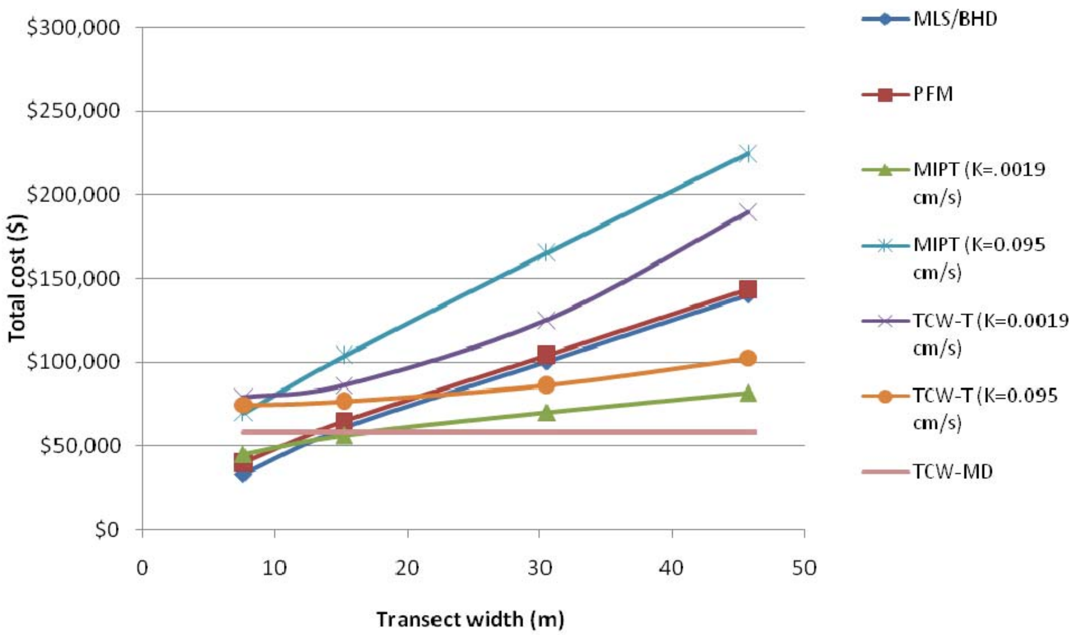
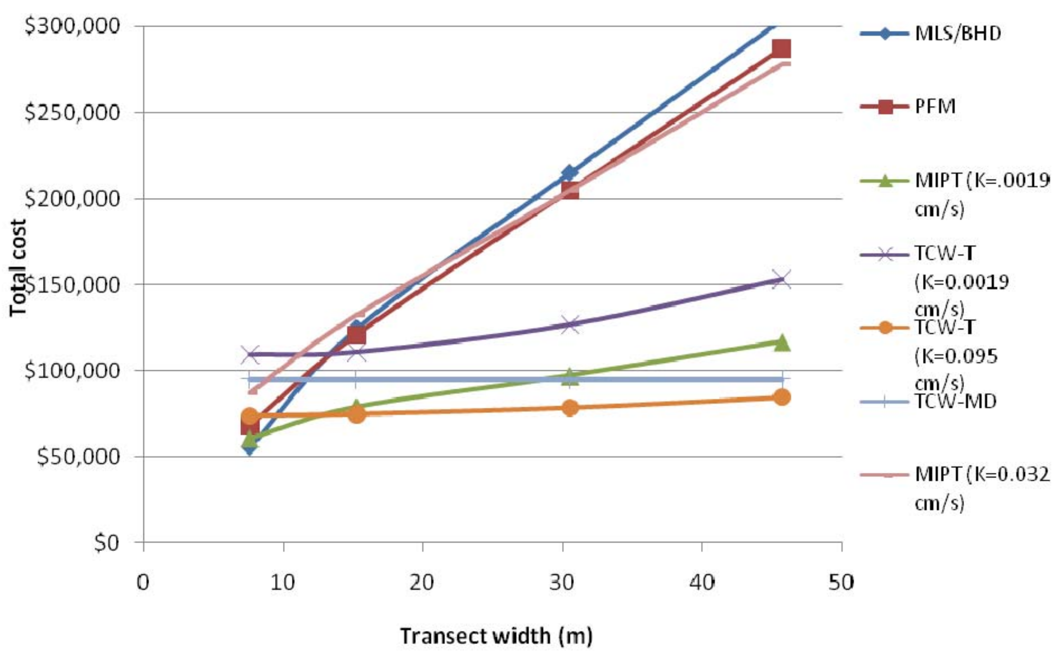


Figure 4.7 Costs for all measurement methods versus transect width at low spatial resolution (21.3m thick aquifer)



At low resolution, the point methods are generally less expensive than the other methods, except for the MIPT at low hydraulic conductivities. The TCW tracer technique at high hydraulic conductivity and TCW multi-dipole technique become relatively more cost efficient when applied in the thicker aquifer (see Figure 4.7).

Figures 4.8 and 4.9 compare costs for all methods at a medium resolution for aquifer thicknesses of 7.6 and 21.3 meters, respectively.

Figure 4.8 Costs for all measurement methods versus transect width at medium spatial resolution (7.6m thick aquifer)

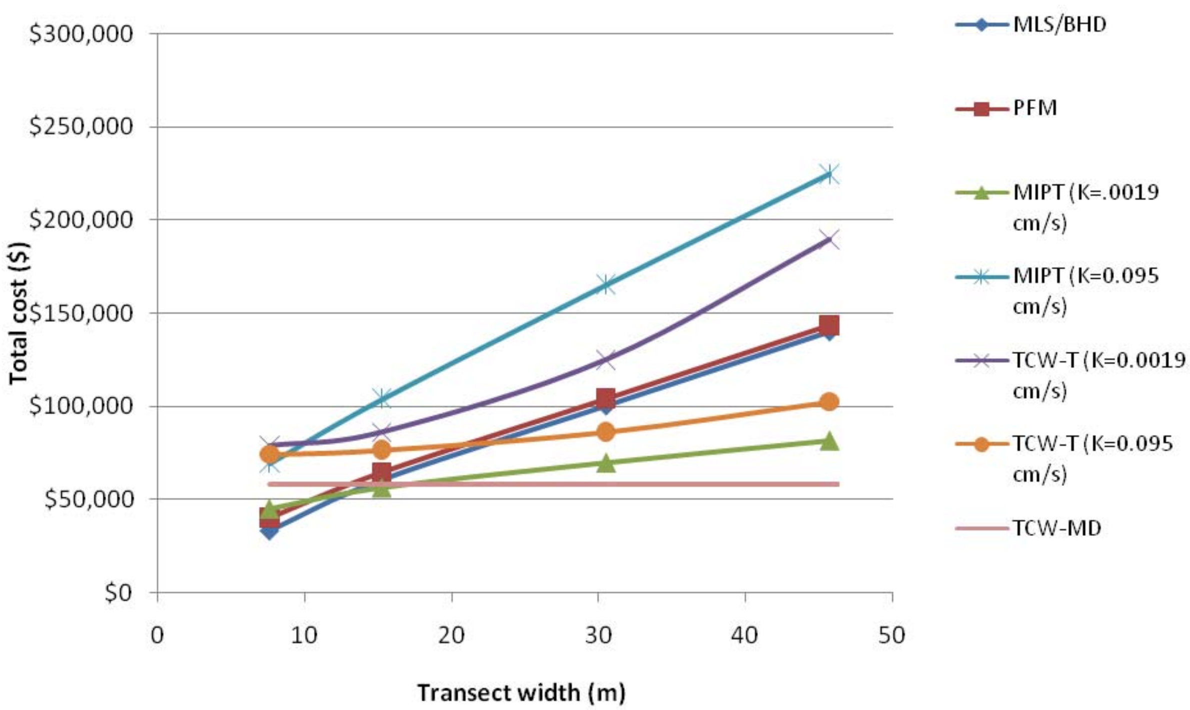
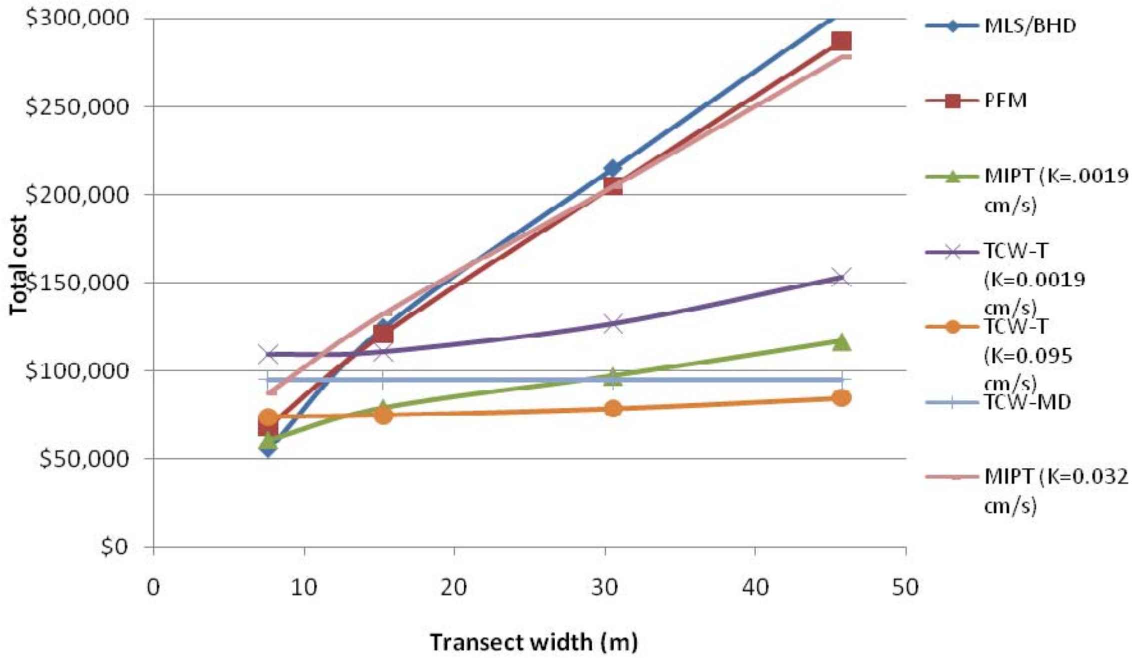


Figure 4.9 Costs for all measurement methods versus transect width at medium spatial resolution (21.3m thick aquifer)



As expected, the graphs in Figures 4.8 and 4.9 reflect a considerable cost increase in the point methods, and to a lesser extent the MIPT method, when compared to costs calculated for low resolution measurements (see Section 4.3.2). However, the MIPT will only provide limited horizontal resolution whereas the point methods will provide both vertical and horizontal resolution. Figures 4.10 and 4.11 compare costs at the high spatial resolution.

Figure 4.10 Costs for all measurement methods versus transect width at high spatial resolution (7.6m thick aquifer)

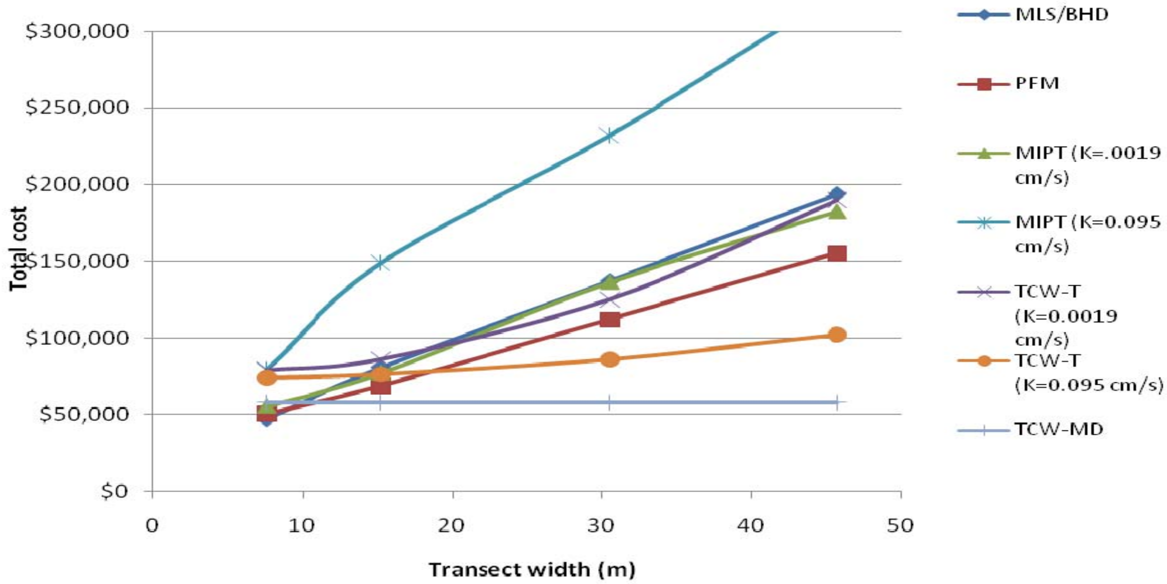
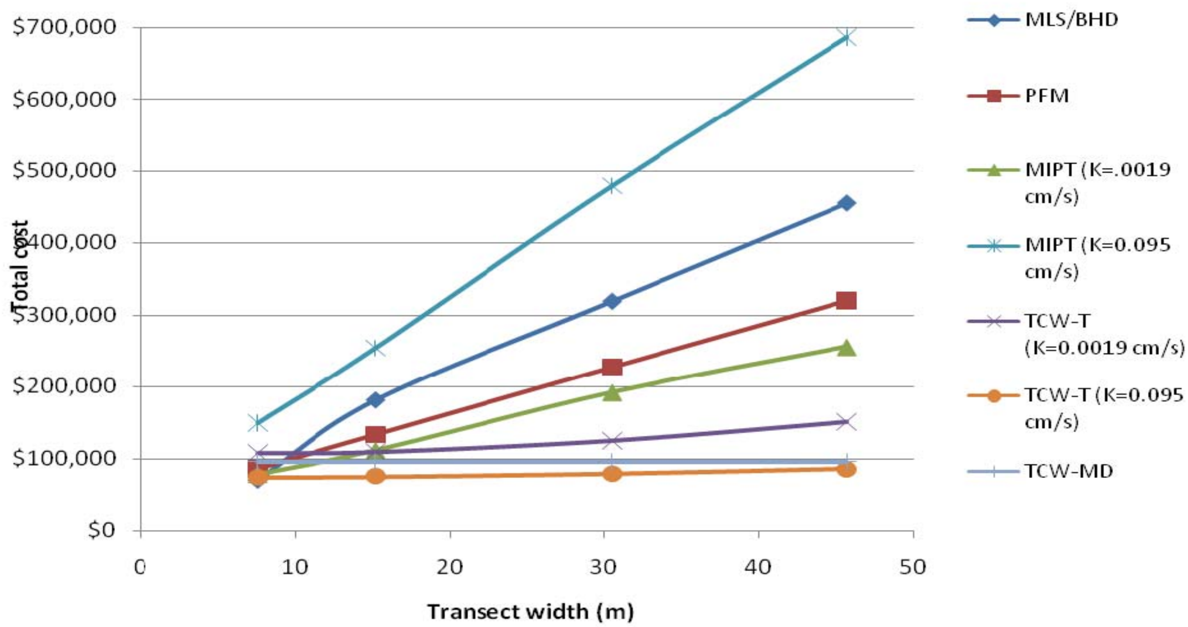


Figure 4.11 Costs for all measurement methods versus transect width at high spatial resolution (21.3m thick aquifer)



As would be expected, costs at high spatial resolution increase for the point methods due to the increase in the number of wells and the number of sampling points. The cost savings of using the PFM method as opposed to the TM/BDT become much greater at this sampling resolution. The MIPT also has increased costs with the addition of more wells. In Figures 4.6 through 4.11, it is apparent that the slopes of the cost versus transect width curves for the different methods are quite different. Looking at these figures, it is unclear how point and pumping method costs might compare, as hydraulic conductivity varies. In particular, since conductivity greatly affects pumping method costs (Figures 4.4 and 4.5), the question should be asked, is there a conducitvity value at which the point method and pumping method costs are approximately equal? To answer this, we developed Figure 4.12, which shows the hydraulic conductivity value at which the TCW-T and MIPT will incur simular costs at the PFM and TM.

Figure 4.12 Graph of methods applied to 7.6m thick aquifer with different hydraulic conductivities

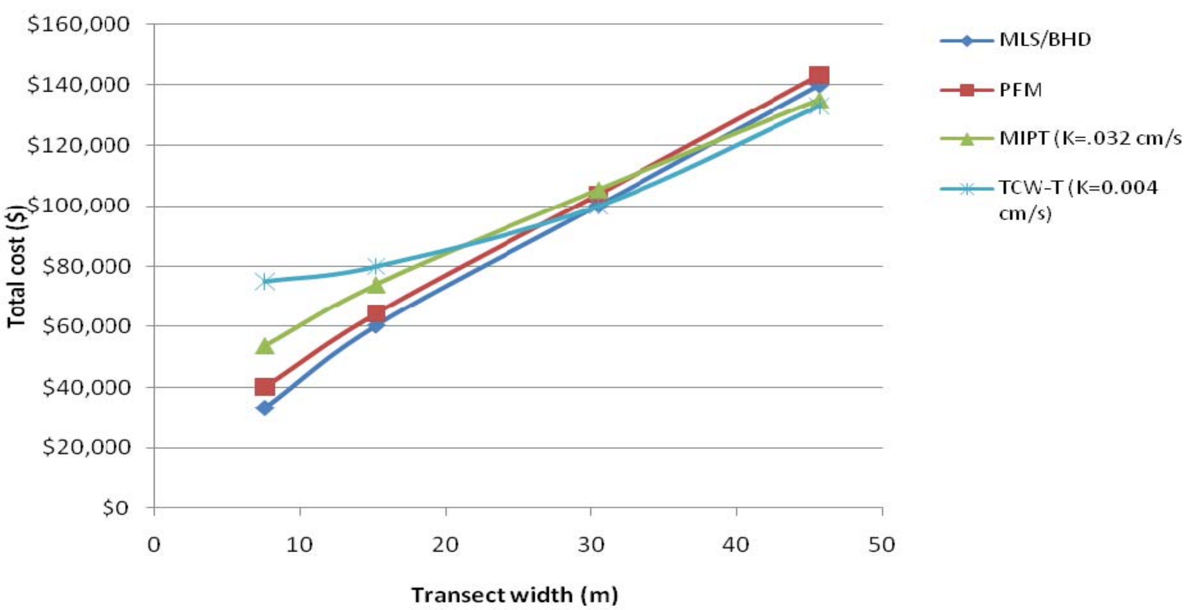
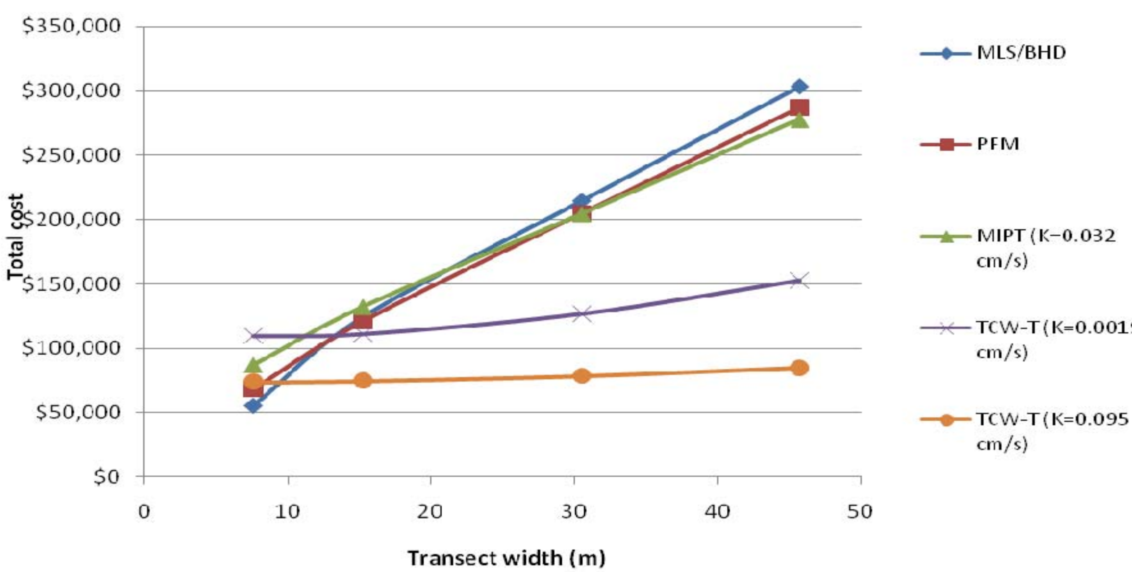


Figure 4.12 shows that the MIPT method applied at medium resolution in an aquifer with a hydraulic conductivity of 0.032 cm/s has costs that are similar to the point methods. Thus, since costs of the MIPT method decrease as conductivity decreases (Figure 4.4), we can

conclude that at conductivites lower than 0.032 cm/s, the MIPT method appears more economical than the point methods (at medium resolution). Likewise it can be surmised that the TCW tracer method will be more economical than the medium spatial resolution point methods when hydraulic conductivity is greater than 0.004 cm/s.

For the thicker aquifers, Figure 4.13 shows the hydraulic conductivities where the MIPT and the TCW tracer technique become more economical then the point methods at medium resolution.

Figure 4.13 Graph of methods applied to 21.3 m thick aquifer with different hydraulic conductivities



From Figure 4.13, it can be seen that at medium spatial resolution in a thick aquifer, the MIPT becomes more economical than the point methods when the hydraulic conductivity is less than 0.032 cm/s. The TCW-T technique will be most economical for determining contaminant mass flux at the full range of hydraulic conductivities. This is true for both the MIPT and the TCW methods for the two aquifer thicknesses used in the template sites.

Having reviewed the factors that influence cost, we now develop a decision tree that can be used by site managers to help them incorporate all factors relevant to the question of which flux measurement technology to use at a site, with given management objectives and site conditions.

4.4 Decision Tree

The decision tree is built with the assumption that basic information on site conditions and management objectives is available (see figure 4.14). Initial investigation of the site should provide the stakeholder with estimates of hydraulic conductivity, aquifer thickness and contaminant plume width, hydraulic gradient, and mean flow direction. From these estimates, a series of decision points will guide the stakeholder to the appropriate and most cost effective approach to measuring contaminant mass flux.

The decision tree, in figure 4.14, is limited to the range of the variables used in the 16 template sites (e.g., hydraulic conductivities between 0.0019 and 0.095cm/s) and the assumptions made in Chapter 3 (e.g., the cost for treating extracted groundwater is $1/gal).

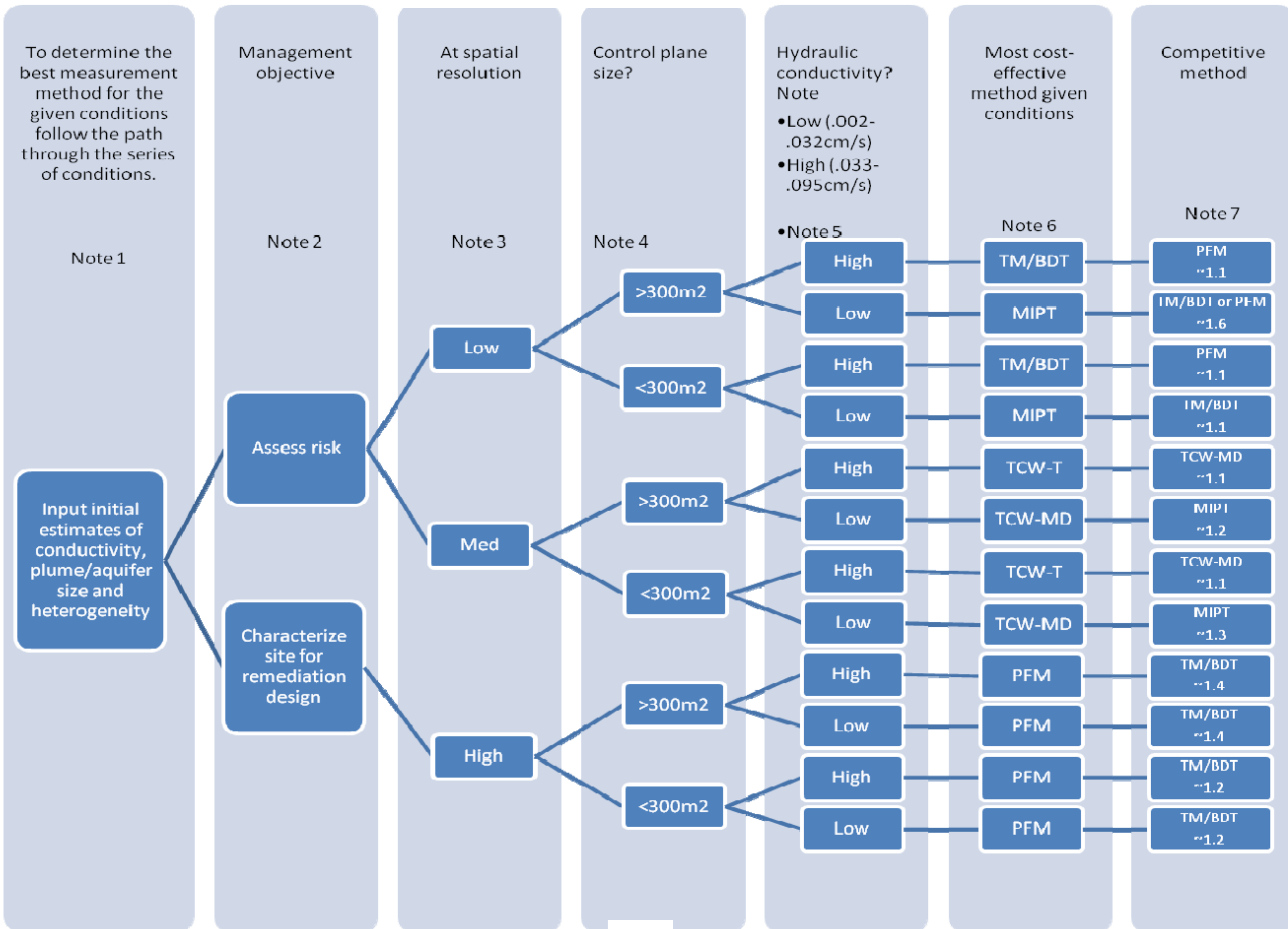
Referring to section 1.2, the objective of the flux measurement may be to either: (1) assess risk in order to develop cleanup priorities or evaluate remediation effectiveness or (2) characterize the site to design a remediation technology. The management objective will drive the required spatial resolution. For assessing risk or evaluating remediation methods, high spatial resolution will typically not be required; however, high resolution may be necessary for site characterization to support a remedial design.

In addition to the objective of the measurement, a second factor that affects the required measurement resolution is the heterogeneity of the site and contaminant distribution. Another factor that affects the decision is the scope of the measurement, as quantified by the control plane area. The control plane area is calculated as the product of the plume width and the aquifer saturated thickness. The control planes at the template sites range in area from $60m^2$ to nearly $1,000m^2$. Based on this range, we have chosen $300m^2$ as a convenient point to delineate large and small control planes.

The last factor used in the decision tree is hydraulic conductivity. The hydraulic conductivities at the template sites range from 0.002 to 0.095 cm/s. Based on this range, we have chosen 0.032 cm/s as a convenient point to delineate low and high conductivities. Note that this also happens to be the conductivity at which the MIPT becomes more economical than the PFM and transect method.

The decision tree outputs the two most cost-effective measurement methods for the given management objectives, spatial resolution, control plane size, and hydraulic conductivity. With these two suggested methods in mind, the technology user must then consider additional factors to ensure the method that is ultimately selected satisfies other requirements. These other factors include regulatory considerations, whether or not the technology is readily available, and complexity of method application and analysis.

Figure 4.14 Decision tree
(See note 8)



4.4.1 Notes for decision tree

This decision tree is a qualitative method of accounting for site conditions and management objectives to help a technology user decide on what flux measurement technology might be most suitable. If site conditions are well known and the required spatial resolution defined, the cost comparison figures (Figures 4.6 through 4.13) might be more useful in selecting the most cost-effective technology.

1. Preliminary site investigations should provide estimates of hydraulic conductivity, aquifer depth, hydraulic gradient, plume size, and heterogeneity.

2. Management objects should provide direction on the type of mass flux investigation to be conducted. Site characterization to support a remedial design, particularly in support of a source zone cleanup, may require considerable spatial detail while a risk assessment in a relatively homogeneous aquifer could be accomplished with low resolution sampling.

3. Spatial resolution is driven by the management objective and level of heterogeneity. Detail on the definition of low, medium, and high spatial resolution can be found in Section 3.3.3.

4. For this decision tree to apply, the size of the control plane should be between 60 and 1000 m^2. 300 m^2 was chosen as the dividing area between large and small control planes. This point is large enough to encompass both aquifer depth templates yet small enough to allow for the illustrate advantages of the point

methods for small areas. This is also the point in which cost differences tend to become more defined under the templates. For control plane areas that are about $300m^2$, both paths (large and small options) should be considered and potential technologies confirmed using the cost comparison figures (Figures 4.6 through 4.13).

5. This decision tree only applies to hydraulic conductivities ranging from 0.002 to 0.095 cm/s. Hydraulic conductivity is separated into high and low ranges, delineated by a value of 0.032 cm/s. This value was selected because it is also the value below which the MIPT method becomes competitive with the point methods. For hydraulic conductivities near 0.032 cm/s, both paths (high and low) should be considered and potential technologies confirmed using the cost comparison figures (Figures 4.6 through 4.13).

6. The end product are the two least expensive methods.

7. The second "competitive method" is provided with a relative cost increase. For example, the annotation TCW-T, ~1.1 next to TM/BDT indicates that the TCW tracer method is estimated to be 1.1 times as expensive as the TM method under the specified conditions.

8. In addition to the criteria in the decision tree, the potential user should review other subjective factors provided in Table 4.2 to ensure the method will meet requirements and avoid possible regulatory or implementation problems.

Table 4.2 Subjective factors to consider when selecting a flux measurement method

Factor to consider		TM/BDT	PFM	MIPT	TCW	
					Multi-dipole	Tracer
Type of measurement		Point	Time-averaging	Volume Averaging	Volume Averaging	Volume Averaging
Simplicity/Implementability*		Good	Moderate to poor	Good	Moderate	Poor
Regulatory Considerations*		Good to Moderate*	Moderate	Moderate	Good	Poor
Availability*		Good	Poor	Good	Poor	Poor
Concerns*	**1**	(TM) Point sampling in time which can't account for temporal hydrologic changes	Very sensitive to divergence and convergence factor (α), which is difficult to quantify	Extracts groundwater and therefore requires the treatment of contaminated water	Small head differences may be difficult to measure	May require a relatively long and expensive tracer test (days-weeks)
	2	(TM) Potential to miss "hot spots" if sampling resolution is insufficient	Can be sensitive to biological activity which could degrade resident tracers	May be difficult to measure small head differences	If contaminate is primarily in the upper or lower region of the aquifer, circulation pattern has potential to spread contaminant	Does not provide the spatial resolution of the point methods which may be required for site characterization
	3	(BDT) Sensitive to divergence and convergence factor (α), which is difficult to quantify	Requires special knowledge to deploy method and interpret results	Does not provide the spatial resolution of the point methods which may be required for site characterization, although emplacement of additional wells will allow for horizontal delineation of contaminant concentrations	Does not provide the spatial resolution of the point methods which may be required for site characterization	Requires the use of tracers which may raise some regulatory concerns
	4	(BDT) Requires the use of tracers which may raise some regulatory concerns	Deployment of method requires days to weeks		Calculations for determining the flux can be rather complex	Calculations for determining the flux can be rather complex
	5		Requires the use of tracers which may raise some regulatory concerns		Has not been field tested	Has not been field tested

* Determined from the considering tables 2.2, 2.3, 2.4, and 2.5

4.4.2 Examples for decision tree use

Example 1:

- Aquifer thickness of 10 m
- Plume width of 40 m
- Hydraulic conductivity of 0.07 cm/s
- Low resolution required to assess risk (homogeneous conditions)

Given these conditions, the decision tree shows the TM with the BDT is the least expensive method followed by the PFM, which is 10% more expensive. This can be seen in figure 4.6 where the cost for implementing the TM is about $50K and the PFM about $55K. From table 4.2, we see there are potential regulatory concerns with the use of the BDT. In addition, at low resolution, the TM could possibly miss "hot spots" if the contaminant distribution and aquifer are not homogeneous. If this is the case, it may be necessary to use a higher sampling resolution or employ a pump method.

Example 2

- Aquifer thickness of 20 m
- Plume width of 40 m
- Hydraulic conductivity of 0.003 cm/s
- Medium resolution required to assess risk (homogeneous aquifer, different source locations)

Using the decision tree for the given conditions, the TCW-MD is the least costly method followed by the MIPT. Referencing table 4.2 there are no issues that would impact a decision to employ either method.

Example 3

- Aquifer thickness of 25 m
- Plume width of 45 m
- Hydraulic conductivity of 0.005 cm/s
- High resolution required (site investigation for designing a remediation)

Using the decision tree, the optimum economic choice is the PFM with the next option being the TM with an estimated cost of 40% over the cost of the PFM. Reviewing table 4.2, assistance and training may be required to design, assemble and deploy the PFM. However the savings associated with using the PFM make the PFM more attractive. When using the PFM, inaccuracies in estimating the convergence/divergence factor should be considered.

V. Conclusions

5.1 Summary

This study examined methods for measuring contaminant mass flux in groundwater. Mass flux measurements may be used to 1) prioritize contaminated groundwater sites for remediation, 2) evaluate the effectiveness of source removal technologies or natural attenuation processes, and 3) define a source term for groundwater contaminant transport modeling, which can be used as a tool to achieve the previous two objectives.

The following mass flux measurement methods, some in use and some emerging, were discussed in detail: (1) the conventional transect method using multi-level sampling (MLS), (2) the borehole dilution test (BDT) which complements the MLS by measuring groundwater flux, (3) the modified integral pump test (MIPT), (4) passive flux meters (PFMs), and (5) tandem circulating wells (TCWs). Discussion included the advantages and limitations of each method, as well as the results of field and laboratory evaluations on method accuracy.

Detailed cost analyses were conducted to quantify the relative costs of applying the different measurement methods. In accordance with recommendations that have been made to facilitate technology transfer, the costs of applying each method to 16 different template sites, which spanned a broad spectrum of hydrogeological conditions, were determined. The costs were then compared to determine which method was most economical under which site condition. Finally, the results of the literature review and the cost analysis provided the necessary information to develop a decision tree to aid potential technology users in determining which method would best meet both management objectives and economic constraints.

5.2 Conclusions

The results of the cost analysis offer insight as to how various factors, which are a function of site conditions and management objectives, affect the costs for implementing the different methods. Each variable affects the cost of methods differently. The results of the analysis, which looks at the impact of these factors on cost, follows.

5.2.1 Spatial resolution

Spatial resolution has the greatest effect on the cost of the two point methods: the TM and PFM methods. As the spatial resolution increases, so do the costs of applying the methods. We found that at high resolution, the PFM will be less costly than the TM/BDT method. The volume averaging MIPT and TCW methods do not offer the same spatial resolution flexibility or capability as the point methods. The MIPT can provide limited horizontal resolution without incurring large cost increases. The cost for adding an additional well is tempered by a reduction in extracted groundwater treatment costs as a result of more efficient pumping to meet the capture width. The TCW only offers increased resolution by installing additional well pairs. This would increase costs by 40 to 90% for only a limited increase in horizontal resolution.

5.2.2 Control plane area

The area of the control plane is driven by the plume width and the aquifer thickness and both measurements have an influence on the costs of implementing the methods. An increase in control plane area increases costs for the point methods, as more sampling points and wells are required to maintain a specified level of sampling resolution. Application of the MIPT will also become more costly as the control plane area increases. This is because more water flows across the control plane, resulting in the need for more contaminated groundwater to be captured and extracted, and therefore treated and disposed of. In this study, the TCW method only used a single pair of wells. Thus, the only additional TCW-T method costs associated with an increase in control plane area are due to increases in tracer test time (thereby incurring additional labor and analytical costs) as a result of having to locate the TCWs farther apart from each other. The TCW multi-dipole method would not incur any additional costs.

5.2.3 Hydraulic conductivity

The hydraulic conductivity has little influence on the cost of the point methods. The only effect that lower conductivity would have on the point methods is that the duration of the PFM and BDT methods would increase. The MIPT and TCW method costs are sensitive to changes in hydraulic conductivity. The MIPT, when treatment of extracted groundwater is required, will sustain higher costs as conductivity increases. Conversely, the TCW tracer technique will incur higher costs as conductivity decreases, due to increases in the tracer test duration. Assuming additional TCWs are not required, the cost of applying the TCW multi-dipole technique is not affected by conductivity though method accuracy may be improved at low conductivities, due to larger head differences to measure at the wells. Hydraulic conductivity can be a decisive element when choosing between methods.

5.2.3 Conclusion on method costs

Field and laboratory test results of the newer methods indicate that their accuracy is as good as, or better than, the accuracy of the TM, the currently accepted method. Comparing test results with known fluxes and to that of the TM, under the same conditions, provide the "apple to apples" comparison suited for stakeholder decision making.

Based on the cost analysis, it can be concluded that for low spatial resolution the point methods will be the least expensive. This is true for control plane sizes smaller than 300 m^2 and where the hydraulic conductivity is greater than 0.016 cm/s. If the hydraulic conductivity is less than 0.016 cm/s, it is likely that the MIPT will be most cost effective for larger control planes. Additionally, for control plane areas approaching or exceeding 1,000 m^2, the TCW-MD technique will provide the most economical means to measure flux.

At a medium sampling resolution, cost comparisons of the methods illustrate how the pumping methods can either exceed or be under the costs associated with the point methods. At control plane areas lower than about 30 m^2, the point methods prove least expensive. It is also apparent that at low hydraulic conductivities, the MIPT will be least expensive for a wide range of control plane sizes (200 to 600m^2). At larger control planes and low hydraulic conductivity, the TCW-MD will be the least expensive.

At high sampling resolutions, the pumping methods are necessarily excluded. Aside from very small control planes (less than 70 m^2), the PFM will prove to be less costly than the TM. This difference increases as the size of the control plane increases.

5.3 Recommendations for future research

1. Define spatial resolution as a function of measurement objective and heterogeneity (of both the aquifer and the contaminant) more rationally. This study based the relationship (in Table 3.4) upon anecdotal reports of past practices. Hopefully, a measure of heterogeneity (perhaps correlation length), in concert with a specified measurement objective, can be used to obtain a quantitative measure of spatial resolution.

2. Develop software to automate the decision tree. Produce a user-friendly interface where initial site investigation parameters and management objectives are input and the most applicable and economic measurement methods are output.

3. In the current study, we try to provide decision makers with credible cost and performance information in order to facilitate transfer and commercialization of an innovative measurement technology. How successful was this approach? Is the information presented in a useable form? What else do potential measurement technology users need to facilitate technology transfer?

References

AFRL, IRP, Aerobic cometabolic *in situ* bioremediation technology guidance manual and screening software user's guide, Earth Tech, INC Alexandria VA., 1998

Annable MD, Hatfield K, Cho J, et al. Field-scale evaluation of the passive flux meter for simultaneous measurement of groundwater and contaminant fluxes. *Environmental Science & Technology, 39*(18), 7194-7201. 2005.

API, Groundwater Remediation Strategies Tool. Regulatory Analysis and Scientific Affairs Department, Publication No. 4730, American Petroleum Institute, Washington DC. 2003.

Arnon, S., Z. Ronen, E. Adar, A. Yakirevich, and R. Nativ. Two-dimensional distribution of microbial activity and flow patterns within naturally fractured chalk. *Journal of Contaminant Hydrology* 79, no. 3–4: 165–186. 2005

Basu, N. B., Rao, P. S. C., Poyer, I. C., Annable, M. D., & Hatfield, K. Flux-based assessment at a manufacturing site contaminated with trichloroethylene. *Journal of Contaminant Hydrology, 86*(1-2), 105-127. 2006.

Bauer, S., Bayer-Raich, M., Holder, T., Kolesar, C., Muller, D., and Ptak, T., Quantification of groundwater contamination in an urban area using integral pump tests. *Journal of Contaminant Hydrology,* 75(3-4), 183-213. 2004.

Bernstein, A., Adar., Yakirevich A., and Nativ R. Dilution Tests in a Low-Permeability Fractured Aquifer: Matrix Diffusion Effect, *Ground Water* 45, no. 2: 235–241. 2007.

Bidaux, P., and Tsang, C.F., Fluid flow patterns around a well bore or an underground drift with complex skin effects: *Water Resources Research*, v. 27, no. 11, p. 2,993–3,008. 1991.

Bockelmann, A., T. Ptak, and G. Teutsch, An analytical quantification of mass fluxes and natural attenuation rate constants at a former gasworks site, *Journal of Contaminant Hydrology, 53*: 429-453, 2001.

Bockelmann, A., D. Zamfirescu, T. Ptak, P. Grathwohl, and G. Teutsch, Quantification of mass fluxes and natural attenuation rates at an industrial site with a limited monitoring network: a case study, *Journal of Contaminant Hydrology, 60*: 97-121, 2003.

Brooks, M.C., Wood, A.L., Annable, M.D., Hatfield, K., Cho, J., Holbert, C., Rao, P.S.C., Enfield, C.G., Lynch, K and Smith, R.E. Changes in Contaminant Mass Discharge from DNAPL Source Mass Depletion: Evaluation at two field sites, *Journal of Contaminant Hydrology*, In Review, Submitted September 2007.

Brooks M.C., A.L. Wood, US Environmental Protection Agency, Ground Water and Ecosystems Restoration Research, Ada, Ok. Personal correspondence. 2008

Campbell, T. J., K. Hatfield, H. Klammler, M. D. Annable, and P.S.C. Rao. Magnitude and directional measures of water and Cr(VI) fluxes by passive flux meter. *Environmental Science and Technology*, 40, 6392-6397. 2006.

Drost, W., Klotz, D., Koch, A., Moser, H., Neumaier, F., Rauert, W., Point dilution methods of investigating groundwater flow by means of isotopes. *Water Resources Research,* 4 (1), 125– 146. 1968.

Einarson, M. D. and D. M. Mackay, Predicting impacts of groundwater contamination, *Environmental Science and Technology, 35*(3):66A-73A, 2001.

ESTCP Project CU-0114, 2006a University of Florida and Purdue University, Field Demonstration and Validation of a New Device for Measuring Water and Solute Fluxes at CFB Borden, November 2006.

ESTCP Project CU-0114, 2006b University of Florida and Purdue University, Field Demonstration and Validation of a New Device for Measuring Water and Solute Fluxes, December 2007.

Fetter C. W., Applied Hydrogeology. Prentice Hall Inc., Simon and Schuster, New Jersey. 1994

Goltz, M. N., J. Huang, M. E. Close, M. Flintoft, and L. Pang, Use of horizontal flow treatment wells to measure hydraulic conductivity without groundwater extraction, submitted *Journal of Contaminant Hydrology,* 2004.

Goltz M. N., J. Huang, M. E. Close, M. J. Flintoft, and L. Pang, Use of Tandem Wells To Measure Hydraulic Conductivity, submitted *Journal of Contaminant Hydrology,* 2007a.

Goltz M. N., S. Kim, H. Yoon and J. Park, Review of Groundwater Contaminant Mass Flux Measurement, *Korean Society of Environmental Engineers*, 2007b.

Guilbeault, M. A., B. L. Parker, and J. A. Cherry, Mass and flux distributions from DNAPL zones in sandy aquifers, *Ground Water*, 43(1): 70-86, 2005.

Groundwater foundation. (2008, March 10). *The Groundwater Foundation*. Retrieved September 15, 2007, from The Groundwater Foundation: http://www.groundwater.org/gi/sourcesofgwcontam.html. 2008

Halevy, E., Moser, H., Zellhofer, O., and Zuber.A. Borehole dilution techniques: a critical review. *Proceedings of the 1966 Symposium of the International Atomic Energy Agency*, 531-564. 1967

Hatfield, K., Annable, M., Cho, J., Rao, P. S. C., & Klammler, H. A direct passive method for measuring water and contaminant fluxes in porous media. *Journal of Contaminant Hydrology, 75*(3), 155-181. 2004.

Huang, J., M. E. Close, L. Pang, and M. N. Goltz, Innovative method to measure flux of dissolved contaminants in groundwater, Fourth International Conference on Remediation of Chlorinated and Recalcitrant Compounds, Monterey California, 24-27 May 2004.

Huang, J., Goltz, M. N., & Roberts, P. V. Comment on 'Analytical solution for solute transport resulting from instantaneous injection in streams with transient storage' by F. de smedt, W. brevis, and P. debels, 2005. *Journal of Hydrology, 330*(3-4), 759-760. 2006

Kabala, Z. J., Dipole flow test; a new single-borehole test for aquifer characterization, *Water Resource Research*, 29(1): 99-107, 1993.

Kao, C.M., Wang, Y.S. Field investigation of natural attenuation and intrinsic biodegradation rates at an underground storage tank site. *Environ. Geol*., 40 (4–5), 622– 631. 2001.

Kim, S. J., Validation of an innovative groundwater contaminant flux measurement method, MS thesis, AFIT/GES/ENV/05-02, Department of system and engineering management, Air Force Institute of Technology, Wright-Patterson AFB OH, 2005.

Klammler, H., Hatfield, K., and Annable, M. D. Concepts for measuring horizontal groundwater flow directions using the passive flux meter. *Advances in Water Resources, 30*(4), 984-997, 2007.

Kubert M., and M. Finkel, Contaminant mass discharge estimation in groundwater based on multi-level point measurements: A numerical evaluation of expected errors. *Journal of Contaminant Hydrology, 2005.*

Lee, J., Rao, P. S. C., Poyer, I. C., Toole, R. M., Annable, M. D., & Hatfield, K. Oxyanion flux characterization using passive flux meters: Development and field testing of surfactant-modified granular activated carbon. *Journal of Contaminant Hydrology, 92*(3-4), 208-229, 2007.

Lile O.B., M Morris, J.S. Ronning. Estimating groundwater flow velocity from changes in contact resistance during a saltwater tracer experiment. *Journal of Applied Geophysics, 38*(2), 105-114, 1997.

Mandalas, G. C., Christ, J. A., & Goltz, M. N. Screening software for an innovative in situ bioremediation technology. *Bioremediation Journal, 2*(1), 7-15, 1998.

McCarty, P. L., M. N. Goltz, G. D. Hopkins, M. E. Dolan, J. P. Allan, B. T. Kawakami, and T. J. Carrothers, Full-scale evaluation of *in situ* cometabolic Degradation of trichloroethylene in groundwater through toluene injection, *Environmental Science and Technology, 32*(1):88-100, 1998.

Meinardus, H. W., Dwarakanath, V., Ewing, J., Hirasaki, G. J., Jackson, R. E., Jin, M., et al. Performance assessment of NAPL remediation in heterogeneous alluvium. *Journal of Contaminant Hydrology, 54*(3-4), 173-193. 2002.

National Research Council (NRC), Innovations in ground water and soil cleanup. Washington DC. 1997.

Nielsen David M., Practical Handbook of Environmental Site Characterization and Ground-Water Monitoring, CRC press, 2006.

Pitrak, M. (Department of Applied Geophysics, Charles University), Mares, S., & Kobr, M. A simple borehole dilution technique in measuring horizontal ground water flow. *Ground Water, 45*(1), 89-92. 2007.

Ptak, T., L. Alberti, S. Bauer, M. Bayer-Raich, S. Ceccon, P. Elsass, T. Holder, C.Kolesar, D. Muller, C. Padovani, G. Rinck, G. Schafer, M. Tanda, G. Teutsch, and A. Zanini, Integrated concept for groundwater remediation, *Integral groundwater investigation*, 2003.

Rao, P. S. C., Annable, M.D., Sillan, R.K., Dai, D., Hatfield, K., Graham, W.D., Wood, A.L., Enfield, C.G., Field-scale evaluation of in situ cosolvent flushing for enhanced aquifer remediation, *Water Resources Research*, 33 (12), 2673-2686. 1997.

RS Means, Environmental remediation estimating methods. Construction publishers and consultants, Kingston MA. 2003

RS Means, Heavy construction cost data, Construction publishers and consultants, Kingston MA. 2006

Wilson J. T., W. A. Mandell, F.L. Paillet, E.R. Bayless, and R.T. Hanson et al. An evaluation of borehole flowmeters used to measure horizontal ground-water flow in limestones of Indiana, Kentucky, and Tennessee, 1999, Water-Resources Investigations Report 01-413. 2001.

SERDP. Annual Technical Report: Impacts of DNAPL source zone treatment: Experimental and modeling assessment of the benefits of partial source removal, December, 2004.

Teutsch, G., Ptak, T., Schwarz, R. and Holder, T. (2000): A new integral procedure for quantification of contaminant mass flow in groundwater. *Ground Water* 4(5), 170-175. 2000

Yoon, H. Validation of methods to measure mass flux of a groundwater contaminant , MS thesis, AFIT/GES/ENV/06-02, Department of system and engineering management, Air Force Institute of Technology, Wright-Patterson AFB OH, 2006.

NGWA. (n.d). *National Groundwater Association*. Retrieved Feb 7, 2008, from Groundwater facts: http://www.ngwa.org/public/gw_use/faqs.aspx, 2008.

USEPA, Ground Water and Ecosystems Restoration Division, Flux-Based Methods for DNAPL Remediation Design and Assessment 2007 Draft

USEPA. (2008, January 10). *United States Environmental Protection Agency*. Retrieved 08 08, 2007, from Under ground storage tanks: http://www.epa.gov/oust/cat/index.htm, 2008.

USGAO. *Leaking Underground Storage Tanks.* Washington, D.C. 20548: Government Accountability Office. 2007.

APPENDIX A

For Aquifer Thickness of 7.6m (4.6m BSL) (Low Resolution)									
								Total cost ($) for:	
Method	Aquifer depth (m)	Plume width (m)	number of wells	sample every (m)	Instal well cost ($)	Capital cost ($)	Cost / episode	K=0.0019 cm/s	K=0.095 cm/s
TM/BDT	7.6	7.6	2	8	$13,153	$7,640	$4,960	$25,753	$25,753
PFM	7.6	7.6	2	8	$13,153	$13,140	$5,973	$32,266	$32,266
MIPT	7.6	7.6	2	n/a	$13,158	$10,023	$10,900	$34,081	$61,717
TCW	7.6	7.6		n/a	$46,268	$25,484	$7,256	79,008	$74,202
TM/BDT	7.6	15.24	3	8	$16,944	$7,640	$9,015	$33,600	$33,600
PFM	7.6	15.24	3	8	$16,944	$13,140	$9,576	$39,660	$39,660
MIPT	7.6	15.24	2	n/a	$13,158	$10,023	$11,464	$34,645	$89,916
TCW	7.6	15.24		n/a	$46,268	$25,484	$14,603	86,355	$76,345
TM/BDT	7.6	30.5	4	8	$20,736	$7,640	$12,020	$40,396	$40,396
PFM	7.6	30.5	4	8	$20,736	$13,140	$13,179	$47,055	$47,055
MIPT	7.6	30.5	3	n/a	$16,952	$12,832	$16,616	$46,400	$144,983
TCW	7.6	30.5		n/a	$46,268	$25,484	$53,542	125,294	$84,990
TM/BDT	7.6	45.7	6	8	$29,889	$7,640	$18,030	$55,559	$55,559
PFM	7.6	45.7	6	8	$29,889	$13,140	$20,386	$63,415	$63,415
MIPT	7.6	45.7	4	n/a	$20,746	$15,641	$21,792	$58,179	$201,276
TCW	7.6	45.7		n/a	$46,268	$25,484	$118,045	189,797	99,312

For Aquifer Thickness of 21.3m (4.6m BSL) (Low Resolution)									
								Total cost ($)	
Method	Aquifer depth (m)	Plume width (m)	number of wells	sample every (m)	Instal well cost ($)	Capital cost ($)	Cost / episode	For K=0.0019	For K=0.095
TM/BDT	21.3	7.6	2	8	$23,254	$7,640	$9,620	$40,514	$40,514
PFM	21.3	7.6	2	8	$23,254	$13,140	$13,140	$50,896	$50,896
MIPT	21.3	7.6	2	n/a	$21,995	$10,248	$12,032	$44,275	$127,400
TCW	21.3	7.6		n/a	$57,319	$25,484	$26,830	$109,633	$73,770
TM/BDT	21.3	15.24	3	8	$31,311	$7,640	$22,005	$60,956	$60,956
PFM	21.3	15.24	3	8	$31,311	$13,140	$22,370	$66,821	$66,821
MIPT	21.3	15.24	2	n/a	$21,995	$10,248	$13,729	$45,971	$212,221
TCW	21.3	15.24		n/a	$57,319	$25,484	$28,369	$111,172	$74,617
TM/BDT	21.3	30.5	4	8	$40,938	$7,640	$29,340	$77,917	$77,917
PFM	21.3	30.5	4	8	$40,938	$13,140	$30,238	$84,315	$84,315
MIPT	21.3	30.5	3	n/a	$31,327	$13,170	$20,656	$65,152	$361,679
TCW	21.3	30.5		n/a	$57,319	$25,484	$44,162	$126,966	$78,034
TM/BDT	21.3	45.7	6	8	$58,622	$7,640	$44,010	$110,271	$110,271
PFM	21.3	45.7	6	8	$58,622	$13,140	$45,974	$117,735	$117,735
MIPT	21.3	45.7	4	n/a	$40,959	$16,091	$27,656	$84,706	$515,126
TCW	21.3	45.7		n/a	$57,319	$25,484	$70,325	$153,128	$83,694

For Aquifer Thickness of 7.6m (4.6m BSL) (med resolution)									
								Total cost ($)	
Method	Aquifer depth (m)	Plume width (m)	number of wells	sample every (m)	Instal well cost ($)	Capital cost ($)	Cost / episode ($)	K=0.0019 cm/s	K=0.095 cm/s
TM/BDT	7.6	7.6	3	4	$ 16,944	$ 7,640	$ 8,746	$ 33,331	$ 33,331
PFM	7.6	7.6	3	4	$ 16,944	$ 13,140	$ 10,138	$ 40,223	$ 40,223
MIPT	7.6	7.6	3	n/a	$ 16,952	$ 12,832	$ 15,105	$ 44,889	$ 69,454
TCW	7.6	7.6		n/a	$ 46,268	$ 25,484	$ 7,256	$ 79,008	$ 74,202
TM/BDT	7.6	15.24	6	5	$ 29,889	$ 7,640	$ 23,093	$ 60,622	$ 60,622
PFM	7.6	15.24	6	5	$ 29,889	$ 13,140	$ 21,511	$ 64,540	$ 64,540
MIPT	7.6	15.24	4	n/a	$ 20,746	$ 15,641	$ 19,843	$ 56,230	$ 103,825
TCW	7.6	15.24		n/a	$ 46,268	$ 25,484	$ 14,603	$ 86,355	$ 76,345
TM/BDT	7.6	30.5	11	5	$ 50,416	$ 7,640	$ 42,337	$ 100,393	$ 100,393
PFM	7.6	30.5	11	5	$ 50,416	$ 13,140	$ 40,465	$ 104,021	$ 104,021
MIPT	7.6	30.5	5	n/a	$ 26,110	$ 18,450	$ 25,089	$ 69,649	$ 165,151
TCW	7.6	30.5		n/a	$ 46,268	$ 25,484	$ 53,542	$ 125,294	$ 84,990
TM/BDT	7.6	45.7	16	5	$ 70,944	$ 7,640	$ 61,581	$ 140,165	$ 140,165
PFM	7.6	45.7	16	5	$ 70,944	$ 13,140	$ 59,419	$ 143,503	$ 143,503
MIPT	7.6	45.7	6	n/a	$ 29,904	$ 21,259	$ 30,328	$ 81,491	$ 224,588
TCW	7.6	45.7		n/a	$ 46,268	$ 25,484	$ 118,045	$ 189,797	$ 99,312

For Aquifer Thickness of 21.3m (4.6m BSL) (med resolution)									
								Total cost ($)	
Method	Aquifer depth (m)	Plume width (m)	number of wells	sample every (m)	Instal well cost ($)	Capital cost ($)	Cost / episode	K=0.0019 cm/s	K=0.095 cm/s
TM/BDT	21.3	7.6	3	4	$31,311	$7,640	$16,692	$55,643	55,643
PFM	21.3	7.6	3	4	$31,311	$13,140	$23,945	$68,396	68,396
MIPT	21.3	7.6	3	n/a	$31,327	$12,832	$16,112	$60,608	134,497
TCW	21.3	7.6		n/a	$57,319	$25,484	$26,830	$109,633	73,770
TM/BDT	21.3	15.24	6	5	$58,622	$7,640	$58,185	$124,446	124,446
PFM	21.3	15.24	6	5	$58,622	$13,140	$49,124	$120,885	120,885
MIPT	21.3	15.24	4	n/a	$40,959	$15,641	$21,794	$78,844	222,003
TCW	21.3	15.24		n/a	$57,319	$25,484	$28,369	$111,172	74,617
TM/BDT	21.3	30.5	11	5	$100,476	$7,640	$106,672	$214,788	214,788
PFM	21.3	30.5	11	5	$100,476	$13,140	$91,088	$204,705	204,705
MIPT	21.3	30.5	5	n/a	$49,021	$18,450	$29,002	$97,036	384,297
TCW	21.3	30.5		n/a	$57,319	$25,484	$44,162	$126,966	78,034
TM/BDT	21.3	45.7	16	5	$140,761	$7,640	$155,159	$303,560	303,560
PFM	21.3	45.7	16	5	$140,761	$13,140	$133,053	$286,954	286,954
MIPT	21.3	45.7	6	n/a	$58,654	$21,259	$36,192	$116,780	547,200
TCW	21.3	45.7		n/a	$57,319	$25,484	$70,325	$153,128	83,694

For Aquifer Thickness of 7.6m (4.6m BSL) (High Resolution)									
								Total cost ($)	
Method	Aquifer depth (m)	Plume width (m)	number of wells	sample every (m)	Install well cost ($)	Capital cost ($)	Cost / episode	K=0.0019 cm/s	K=0.095 cm/s
TM/BDT	7.6	7.6	4	2	$20,736	$7,640	$21,290	$47,471	$47,471
PFM	7.6	7.6	4	2	$20,736	$13,140	$16,929	$50,805	$50,805
MIPT	7.6	7.6	4	n/a	$20,746	$15,641	$19,358	$55,745	$79,542
TCW	7.6	7.6		n/a	$46,268	$25,484	$7,256	$79,008	$74,202
TM/BDT	7.6	15.24	6	2	$29,889	$7,640	$43,343	$80,872	$80,872
PFM	7.6	15.24	6	2	$29,889	$13,140	$69,040	$69,040	$69,040
MIPT	7.6	15.24	6	n/a	$29,904	$21,259	$28,379	$77,383	$148,963
TCW	7.6	15.24		n/a	$46,268	$25,484	$14,603	$86,355	$76,345
TM/BDT	7.6	30.5	11	2	$50,416	$7,640	$79,462	$137,518	$137,518
PFM	7.6	30.5	11	2	$50,416	$13,140	$112,271	$112,271	$112,271
MIPT	7.6	30.5	10	n/a	$50,444	$35,304	$50,697	$136,445	$231,947
TCW	7.6	30.5		n/a	$46,268	$25,484	$53,542	$125,294	$84,990
TM/BDT	7.6	45.7	16	2	$70,944	$7,640	$115,581	$194,165	$194,165
PFM	7.6	45.7	16	2	$70,944	$13,140	$71,419	$155,503	$155,503
MIPT	7.6	45.7	15	n/a	$67,190	$46,540	$68,740	$182,470	$325,567
TCW	7.6	45.7		n/a	$46,268	$25,484	$118,045	$189,797	$99,312

For Aquifer Thickness of 21.3m (4.6m BSL) (High Resolution)									
								Total cost ($)	
Method	Aquifer depth (m)	Plume width (m)	number of wells	sample every (m)	Install well cost ($)	Capital cost ($)	Cost / episode	K=0.0019 cm/s	K=0.095 cm/s
TM/BDT	21.3	7.6	4	2	$40,938	$7,640	$21,290	$69,867	$69,867
PFM	21.3	7.6	4	2	$40,938	$13,140	$31,738	$85,815	$85,815
MIPT	21.3	7.6	4	n/a	$40,959	$16,091	$20,333	$77,383	$148,963
TCW	21.3	7.6		n/a	$57,319	$22,950	$26,830	$107,099	$73,770
TM/BDT	21.3	15.24	6	2	$58,622	$7,640	$114,885	$181,146	$181,146
PFM	21.3	15.24	6	2	$58,622	$13,140	$61,724	$133,485	$133,485
MIPT	21.3	15.24	6	n/a	$58,654	$21,934	$30,330	$110,917	$254,077
TCW	21.3	15.24		n/a	$57,319	$22,950	$28,369	$108,638	$74,617
TM/BDT	21.3	30.5	11	2	$100,476	$7,640	$210,622	$318,738	$318,738
PFM	21.3	30.5	11	2	$100,476	$13,140	$114,188	$227,805	$227,805
MIPT	21.3	30.5	10	n/a	$100,535	$36,542	$54,610	$191,687	$478,947
TCW	21.3	30.5		n/a	$57,319	$22,950	$44,162	$124,431	$78,034
TM/BDT	21.3	45.7	16	2	$140,761	$7,640	$306,359	$454,760	$454,760
PFM	21.3	45.7	16	2	$140,761	$13,140	$166,653	$320,554	$320,554
MIPT	21.3	45.7	15	n/a	$132,784	$48,228	$74,604	$255,615	$686,035
TCW	21.3	45.7		n/a	$57,319	$22,950	$70,325	$150,594	$83,694

REPORT DOCUMENTATION PAGE	*Form Approved* *OMB No. 074-0188*

1. REPORT DATE *(DD-MM-YYYY)*	**2. REPORT TYPE**	**3. DATES COVERED** *(From – To)*
27-03-2008	Master's Thesis	August 2007 – March 2008

4. TITLE AND SUBTITLE An Evaluation and Implementation Guide for Current Groundwater Mass Flux Measurement Practices	**5a. CONTRACT NUMBER** **5b. GRANT NUMBER** **5c. PROGRAM ELEMENT NUMBER**
6. AUTHOR(S) Wheeldon, Jack G., Major, USAF	**5d. PROJECT NUMBER** **5e. TASK NUMBER** **5f. WORK UNIT NUMBER**
7. PERFORMING ORGANIZATION NAMES(S) AND ADDRESS(S) Air Force Institute of Technology Graduate School of Engineering and Management (AFIT/EN) 2950 Hobson Way, Building 640 WPAFB OH 45433-8865	**8. PERFORMING ORGANIZATION REPORT NUMBER** AFIT/GEM/ENV/08-M23
9. SPONSORING/MONITORING AGENCY NAME(S) AND ADDRESS(ES) U.S. Environmental Protection Agency Ground Water and Ecosystem Restoration Division (GWERD) National Risk Management Research Laboratory 919 Kerr Research Drive, P.O. Box 1198 Ada, OK 74821	**10. SPONSOR/MONITOR'S ACRONYM(S)** U.S. EPA, NRMRL, GWERD **11. SPONSOR/MONITOR'S REPORT NUMBER(S)**

12. DISTRIBUTION/AVAILABILITY STATEMENT
APPROVED FOR PUBLIC RELEASE; DISTRIBUTION UNLIMITED.

13. SUPPLEMENTARY NOTES

14. ABSTRACT
Contaminant mass flux is an important parameter needed for decision making at sites with contaminated groundwater. New and potentially better methods for measuring mass flux are emerging. This study looks at the conventional transect method (TM), and the newer passive flux meter (PFM), modified integral pump test (MIPT), and tandem circulating well (TCW) methods. In order to facilitate transfer and application of these innovative technologies, it is essential that potential technology users have access to credible information that addresses technology capabilities, limitations, and costs. This study provides such information on each of the methods by reviewing implementation practices and comparing the costs of applying the methods at 16 standardized "template" sites. The results of the analysis are consolidated into a decision tree that can be used to determine which measurement method would be most effective, from cost and performance standpoints, in meeting management objectives at a given site.

The study found that, in general: (1) the point methods (i.e. the TM and PFM) were less expensive to use to characterize smaller areas of contamination while the pumping methods (the MIPT and TCW) would be more economical for larger areas, (2) the pumping methods are not capable of high resolution sampling, which may be required to characterize heterogeneous systems or to design remediations, and (3) when high resolution is required, the PFM is more economical then the TM. Finally, the study demonstrated that, arguably, test results of the newer methods indicate that their accuracy is as good as, or better than, the accuracy of the TM, the currently accepted method.

15. SUBJECT TERMS
Groundwater mass flux evaluation, Implementation guide and decision analysis

16. SECURITY CLASSIFICATION OF:			**17. LIMITATION OF ABSTRACT**	**18. NUMBER OF PAGES**	**19a. NAME OF RESPONSIBLE PERSON** Goltz, Mark N., USAF
a. REPORT	**b. ABSTRACT**	**c. THIS PAGE**			**19b. TELEPHONE NUMBER** *(Include area code)*
U	U	U	UU	121	(937) 255-3636, ext 4638 (Mark.Goltz@afit.edu)
					Standard Form 298 (Rev. 8-98) Prescribed by ANSI Std. Z39-18