# AN INTEGRATIVE APPROACH FOR MUNICIPAL SOLID WASTE (MSW) LANDFILL SELECTION:

# A CASE STUDY OF Oʻ AHU, HAWAIʻI

A DISSERTATION SUBMITTED TO THE GRADUATE DIVISION OF THE UNIVERSITY OF HAWAIʻI IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF

DOCTOR OF PHILOSOPHY

IN

NATURAL RESOURCES AND ENVIRONMENTAL MANAGEMENT

DECEMBER 2011

BY

Su Kwan Jung

Dissertation Committee:

John F. Yanagida, Chairperson

Greg Bruland

Michael Robotham

Tomoak Miura

James Moncur

# ACKNOWLEDGEMENTS

As my academic advisor and dissertation chairman, Professor John F. Yanagida provided the major driving force and incentive for this study. He has been always generous in providing his time, resources, and intellectual and philosophical advice for my academic experience. I therefore wish to express my sincerest and deepest gratitude to him.

I am also equally indebted to the other dissertation committee members (Drs. Tomoaki Miura, Greg Bruland, Michael Robotham, and James Moncur). Dr. Miura helped me with the GIS analysis, Dr. Bruland assisted in gaining scientific knowledge related to landfills, and Dr. Robotham provided an idea related to Scenario 2 in the GIS-analysis. Even though Dr. Moncur has retired, he helped me to complete my thesis by providing economic advice.

I would like to thank the anonymous officers in the Department of Environmental Services, City and County of Honolulu. They provided public information related to landfill sites and much needed technical information.

I also would like to thank my family including my wife, son, mother, and brother. This study could not be completed without their help. My mother and brother assisted me financially and spiritually. My wife and newborn baby shared my hardship.

Thank you (Mahalo).

# ABSTRACT

Landfill site selection is becoming increasingly difficult because of growing environmental and public health concerns and increasing scarcity of land for landfill construction. The City and County of Honolulu recently encountered difficulties in selecting a new landfill site. Developing a consistent and reasonable methodology is critically important.

This study develops an integrative methodology that links a geographic information system (GIS)-based analysis with an economic framework. The basic site selection framework minimizes social costs given constraints (exclusionary criteria based on rules and restrictions) using a two-part process: (i) a GIS-based screening of potential sites that satisfy constraints and (ii) benefits transfer methods (e.g., meta-analysis and mean transfer value approaches) that rank the selected sites according to social cost minimization. Meta-analysis models are evaluated in terms of sensitivity, validity, and reliability criteria. Sensitivity analysis examines aggregate values in response to changes in a selected variable (e.g., income, distance from target sites, discount rates, and lengths of landfill life). Validity requires statistical tests (e.g., t and sign rank tests) to examine differences between transfer values and original values. Reliability checks their similarity by utilizing an absolute percentage difference measure.

The GIS-based analysis was conducted on: (i) the City's 45 potential sites (Scenario 1) and (ii) the entire island of Oahu (Scenario 2). Together, both scenarios found 7 sites that satisfied the exclusionary criteria. Benefits transfer (BT) results indicate: (i) social costs are high, and (ii) social costs vary by target sites due to market conditions (e.g., income levels, housing values, population densities, and the number of households), different methods used (e.g., a contingent valuation method and a hedonic price method), different models used (e.g., meta-analysis (MA) models 1 and 2), and landfill life lengths. In terms of the sensitivity, validity, and reliability criteria, meta-analysis models are preferred to the mean transfer value approach. This study provides a potential method that evaluates and illustrates the process of site selection which can be applied to other site selection process (e.g., hazardous waste sites and incinerators). Further research is needed to analyze public preferences incorporating various individuals, groups, and minority populations.

# TABLE OF CONTENTS

# LIST OF TABLES

# LIST OF FIGURES

# LIST OF ABBREVIATIONS

BT: benefits transfer

CVM: contingent valuation method

GIS: geographic information system

GLS: generalized least squares

H-POWER: honolulu power of waste energy recovery

HPM: hedonic price method

ISWM: integrated solid waste management

MA: meta-analysis

MA-BT: meta-analysis for benefits transfer

MSW: municipal solid waste

MTV: mean transfer value

MWTP: marginal willingness to pay

OLS: ordinary least squares

TE: transfer error

WGSL: waimanalo gulch sanitary landfill

WTA: willingness to accept

WTP: willingness to pay

# LIST OF APPENDICES

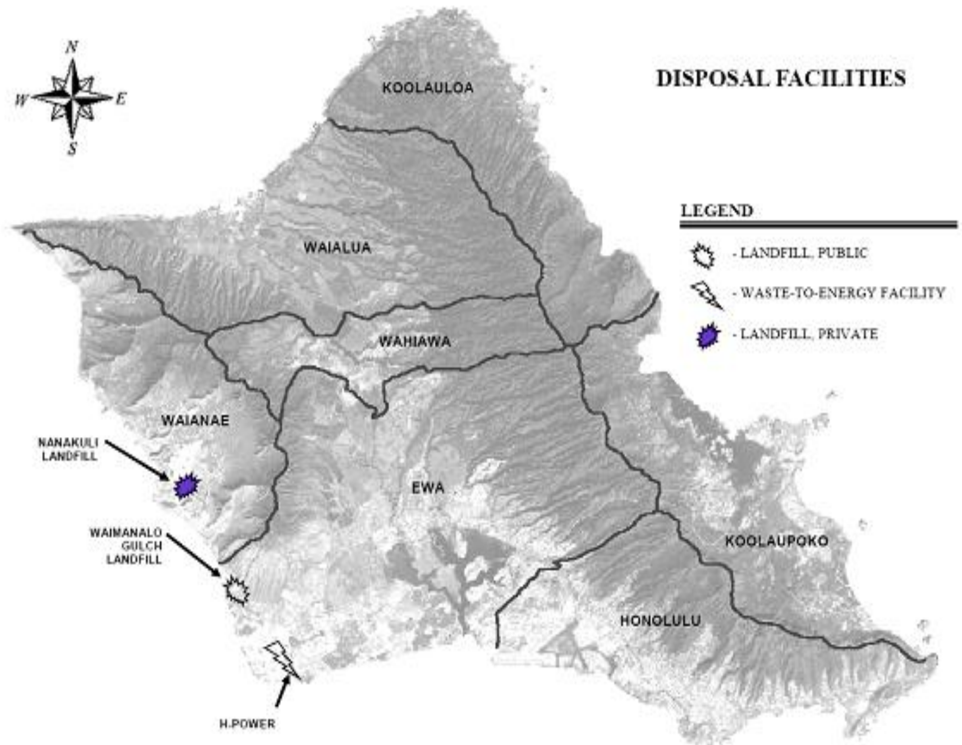
# CHAPTER1. INTRODUCTION

## 1.1. Waste disposal in the City and County of Honolulu

Currently, the City and County of Honolulu employs an integrated solid waste management (ISWM) approach including source reduction, recycling, the Honolulu Power of Waste Energy Recovery Program (H-POWER) plant, and landfill disposal. In terms of disposal facilities, there exist the public Waimanalo Gulch Sanitary Landfill (WGSL) for municipal solid waste (MSW) disposal, the private Nanakuli landfill for construction and demolition (C&D) waste and petroleum-contaminated soils disposal, and the H-POWER plant that converts combustible solid waste into energy (See Figure 1.1). The City tries to reduce dependency on the current landfill through source reduction, recycling, and the H-POWER plant.

In Fiscal Year (FY) 2006, the island of Oahu, Hawaii generated approximately 1.79 million short tons (1.63 million metric tons) of waste from residential, commercial, and industrial sources (see Table 1.1). Recycling diverted about 0.63 million short tons (0.57 million metric tons).[1] The H-POWER plant converted approximately 0.6 million short tons (0.55 million metric tons) of combustible municipal solid waste (MSW) into green electricity that accounted for 7% of Oahu's electricity. The existing Waimanalo landfill disposed of about 0.34 million short tons (0.31 million metric tons) (19%) including municipal solid waste (MSW), recycling residue, and ash and residue from the energy recovery (H-POWER) plant. The private Nanakuli landfill accepts 0.2 million short tons (0.18 million metric tons) of construction and demolition (C&D) waste and petroleum contaminated soils.

[1] Recycling increased when compared to 74,000 short tons or 67,131 metric tons in 1988.




Source: City & County of Honolulu, Department of Environmental Services (2004)
Figure 1.1. Current disposal facilities on Oahu

Table 1.1. Waste Generated in Fiscal Year (FY) 2006

| Management | Amount of Waste (Short Tons) | Percentage (%) |
|---|---|---|
| Recycled/Reused | 628,373 | 35 |
| H-POWER | 602,520 | 34 |
| Public Waimanalo Gulch Sanitary Landfill | 337,667 | 19 |
| Subtotal | 1,569,560 | 86 |
| Private Nanakuli Landfill (Estimation)* | 200,000 | 11 |
| Unpermitted Sites (Estimation) | 25,000 | 1 |
| Total | 1,793,560 | 100 |

Source: City and County of Honolulu (2006, 2008)
*The City keeps records of the amount of municipal solid waste (MSW). Thus, the amount of waste treated by the private Nanakuli site and unpermitted sites is the City's estimation. About 152,801 short tons (138,618 metric tons) of waste delivered to the Landfill were redirected from H-POWER due to periodic closures for facility maintenance (or capacity limitations).

## 1.2. Current operating Waimanalo Gulch Sanitary Landfill (WGSL)

The existing landfill site is located in Waimanalo Gulch, Kahe Valley on the island of Oahu, Hawaii (see Figure 1.1). The City & County of Honolulu owns the existing Waimanalo Gulch Sanitary Landfill (WGSL) facility under jurisdiction of the Department of Environmental Services (ENV), and the Hawaii Waste Management operates the landfill for the ENV.

The climate of the WGSL area is arid and has low precipitation (15 inches or 38.1 cm per year). There exist few residences, schools, and one business (Ko Olina Resort at about 1500 feet or 450.7 meter distance). Because the site is located in the State Agriculture District and the City Zoning General Agricultural District (AG2), its extension requires the State's Special Use Permit by the State Land Use Commission (LUC) and the Conditional Use Permit by the Department of Planning and Permitting (DPP) (City & County of Honolulu 2006).

Table 1.2. Existing and Proposed Use of Waimanalo Gulch Sanitary Landfill (WGSL)

| Description | Acreage | Subtotal |
|---|---|---|
| Used Landfill Area, Scheduled for Closure in 2008 | 60.5 | |
| Administrative and Operational Support | 20 | |
| Roadway and Drainage Area Improvements | 6 | 86.5 |
| 2003 Expansion Area | 21 | 107.5 |
| 2008 Expansion Area | 92.5 | |
| Total Approximate Area of Site | 200 | 200 |

Source: City & County of Honolulu (2006)

The WGSL facility opened in 1989 with about 200 acres or 0.81 $\mathrm{km}^2$. Approximately 107.5 acres or 0.44 $km^2$ of the site are composed of used landfill area (60.5 acres or 0.24 $\mathrm{km}^2$), operation and maintenance (20 acres or 80,937$\mathrm{m}^2$), the internal roadway (6 acres or 24,281$m^2$). The expansion area in 2003 (21 acres or 84,983$\mathrm{m}^2$) was used by 2008. The remaining 92.5 acres or 0.37 $\mathrm{km}^2$ could be used for the next expansion (see Table 1.2).

## 1.3. Problem statement

The existing Waimanalo Gulch Sanitary Landfill (WGSL) site has been the major disposal facility on the island of Oahu that treated about 19% of the island's waste generation in 2006. Despite an attempt to reduce dependency on the current landfill (e.g., source reduction, recycling, and waste energy recovery), increasing production of waste and decreasing capacity of the existing landfill site necessitated finding either more landfill space or another landfill. The City had recently encountered site selection problems (e.g., problems of the site selection process and interested parties). After the complex procedure, the City decided to expand the WGSL site and obtained a special use permit for its extension till 2012. Recently, the State Land Use Commission (LUC) order (2009) and the City's Integrated Waste management Plan requested the City to start the process of site selection. The City (under new Mayor Peter Carlisle's term) is finding a new landfill site.

Landfill site selection is becoming increasingly difficult because of growing environmental and public health concerns, decreased amounts of public funding, and increasing scarcity of land for landfill construction (Kao & Lin 1996). Strong conflicts (e.g., environmental justice, participation, and legal challenges) delay the process (Kreith and George 2002). Even the best case schedule for establishing a landfill takes at least 3.6 years, and legal challenges and obtaining all permits takes even more time (see Table 1.3).

Table 1.3. Average, Worst Case Schedules for Establishing Landfill

| Task Name | Duration |
|---|---|
| Best Case | 1285 days (3.6 years) |
| Average Case | 1648 days (4.5 years) |
| Worst Case | 1854 days (5.1 years) |

Source: State of Hawaii, Auditor (2004)
* Excludes legal challenges and the time required to obtain all permits.

The City had encountered landfill site selection problems (e.g., problems of the site selection process and interested parties). The City's approach (under former Mayor Jeremy Harris's term) incorporates a two-part process: (i) a preliminary screening process based on exclusionary criteria and (ii) a further assessment (the Committee's evaluation based on its criteria and weights).[2] The

[2] Former Mayor Jeremy Harris convened a 15-member committee for site selection (the Mayor's Advisory Committee in 2003), which composed of citizens including community representatives

process of reducing the number of potential sites is shown in Table 1.4. Their preliminary screening procedure selected 8 sites from 45 based on criteria including U.S. EPA restrictions and restrictions on land use, groundwater, and minimum landfill capacity (at least a 10 year).[3] U.S. EPA's exclusionary criteria eliminated 5 sites from 45 to 40. Restrictions on land use reduced 6 sites from 40 to 34. Restrictions on groundwater (e.g., Board of Water Supply staff review and evaluation) decreased 18 sites from 34 to 16. Restriction on landfill capacity decreased eight sites from 16 to 8.

Table 1.4. Attrition of Sites during the Evaluation Process

| Phase of Evaluation | Numbers of Sites | |
|---|---|---|
| | Before applying criteria | After applying criteria |
| Preliminary Site Selection Criteria | | |
| • US EPS's Six Restrictions | 45 | 40 (-5)* |
| • Restrictions on land use | 40 | 34 (-6) |
| • Restriction on groundwater | 34 | 16 (-18) |
| • Restrictions on Landfill Capacity | 16 | 8 (-8) |
| Committee Evaluation Process | | |
| • Committee Criteria | 8 | 8 (-0) |
| • Committee Consensus | 8 | 5 (-3) |
| • Committee Vote** | 5 | 4 (-1) |

Source: Report of Mayor's Advisory Committee on Landfill Selection (2003)
* ( ) is the number of attrition of sites.
** Four Committee members resigned against the vote.

The Committee ranked the 8 sites based on its criteria and weights and reduced the 8 sites to 5 sites by consensus.[4] The remaining 5 sites were Ameron Quarry, Maili, Makaiwa, Nanakuli B, and the current WGSL site (see Figure 1.2). Although the existing WGSL site was the top ranked site, the Committee vote failed to include it and submitted a ranked listing based on the other 4 sites to the City Council. The City Council adopted a Council Resolution (2004) that the City should consider the WGSL for a new landfill. However, the

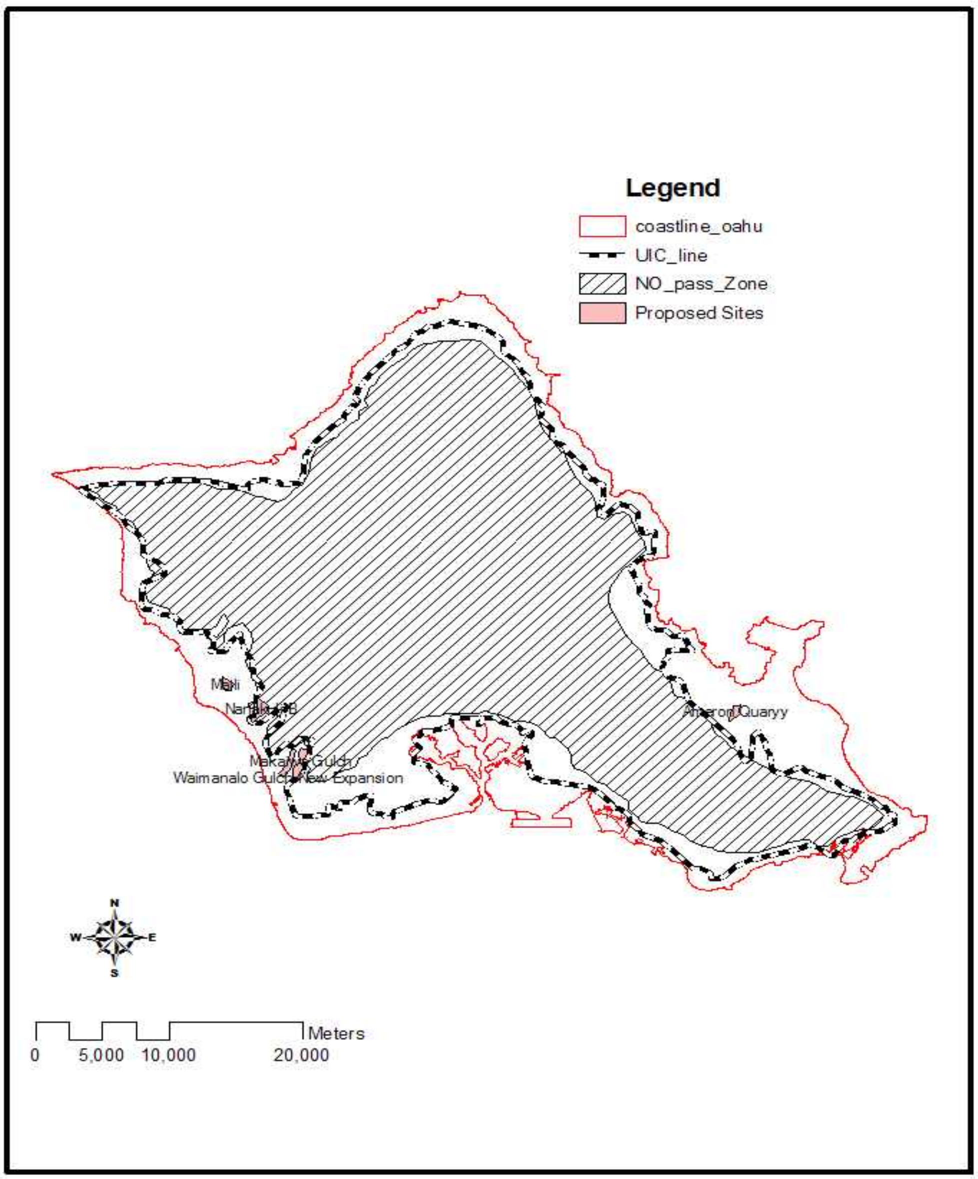
---

from various geographic areas of the City, the business community, and the Department of Health (DOH) (a majority of the Committee members were from Leeward communities).

[3] The Committee decided to limit its consideration to sites with at least 10 years of landfill capacity.

[4] The Committee developed the 31 criteria and weights for criteria from 1 (least important) to 3 (most important) were assigned.

Council changed its stance by passing the Council Bill 37 (2005) instructing the City to close the WGSL in 2008. The former Mayor Mufi Hannemann vetoed the bill because the City complied with the site selection process, and the current WGSL had at least a 20 year of life remaining. The City Council chose the existing site, and the City obtained a special use permit for its extension till 2012.




<UTM Zone 4N, NAD83>

Figure 1.2. Proposed sites by the Committee and current WGSL

Recently, the State Land Use Commission (LUC) order (2009) and the City's Integrated Waste management Plan requested the City to start the site selection process. Mayor Peter Carlisle appointed a 9 member of the Committee for landfill site selection. The Committee developed its criteria and weights and will determine a site with the highest score.[5] Together with the process, the City appealed the deadline scheduled in 2012 and is pursuing the extension of the existing WGSL site with a 15 year of life remaining.

Developing a consistent and reasonable methodology is of critical importance. Without a viable selection or evaluation process, the City will encounter difficulties in selecting a future site.

Environmental justice or equity and participation are important in the process of landfill site selection. Lack of participation can incur strong opposition to a proposed landfill and delay the process or increase costs. When various interest groups involved in the process of landfill site selection participate, the project can be successful (Pearce et al. 2006). Environmental and social equity is becoming an important consideration in evaluating resource use. All relevant policy appraisal processes (e.g., environmental impact statements) require addressing this issue. Despite their importance, this study will not address these issues.

Environmental equity is partially managed by benefits transfer methods (e.g., meta-analysis and mean transfer value approaches) that consider various factors (e.g., income levels, population densities, the number of households, and housing values). In terms of social costs (society's burdens), the least impacted site fulfilling exclusionary criteria (e.g., airports, wetlands, floodplains, land use, and groundwater) is selected. This study assumes that compensation for impacted households and aid to reduce transaction costs for low-income

[5] Eight members differed from those of the former Mayor Jeremy Harris's Committee. The existing WGSL site will not be considered as a new landfill site. The new Committee developed 20 criteria (e.g., landfill capacity, proximity to schools, health care facilities and parks, and impacts on residential tourists and commercial facilities) and weights for criteria with a score of 1 (least important) to 20 (most important). It will determine a site with the highest score.

households (e.g., providing legal service and information) can improve inequity. Managing participation is not desirable for this preliminary study, which has several reasons (e.g., need for rapid decision-making and lack of knowledge of complex issues on the part of the participants).

## 1.4. Objectives

The primary objective of this study is to develop a general methodology for municipal solid waste (MSW) site selection by linking a geographic information system (GIS)-based analysis with an economic framework (e.g., social cost minimization given exclusionary criteria). The GIS-based method acts as a preliminary screening procedure which eliminates unsuitable areas from consideration as possible landfill sites and analyzes geographical characteristics (e.g., land use and soil properties). The economic analysis can provide the basis for measuring the impacts of proposed landfill sites and rank these target sites according to social cost minimization, which has advantages (e.g., replicability, transparency, and ease of comparison between target sites or other projects).

The second objective is to apply this developed methodology to the island of Oahu, Hawaii as a case study. The island of Oahu, Hawaii is ideal as a case study because it has site selection concerns and problems of the site selection process. However, data are available to assist the site selection process. Results are compared with a recent City and County of Honolulu study. This example can assist policy makers and planners in evaluating and illustrating the effectiveness of this approach for the process of landfill site selection.

## 1.5. Integrative methodology

This research employs an integrative methodology that links a GIS-based analysis with an economic analysis. The basic site selection framework minimizes social costs given constraints (restrictive or exclusionary criteria) based on rules and restrictions mandated by public agencies including airports, wetlands, floodplains, groundwater, land use, and landfill capacity with a two-part process: (i) a GIS-based method that selects potential sites to satisfy constraints and (ii) an economic framework that ranks the remaining selected sites according to social cost minimization. The first stage utilizes a GIS-based method to select sites fulfilling constraints and to analyze geographical characteristics (e.g., soil properties) (see Baban and Flannagan 1998; Kontos et al. 2003; Daneshvar et al. 2005; Sener et al. 2006). By processing a large amount of data in a short time, the GIS analysis can reduce the effort required for information collection and processing.

The second stage utilizes benefits transfer (BT) methods (e.g., meta-analysis and mean transfer value approaches) in order to measure social costs for the selected target sites and ranks these sites according to social cost minimization. If planners fail to consider social costs, they will likely grossly underestimate the costs of the landfill and possibly locate the landfill in a higher overall cost location. The measurement of external or social costs can help planners to locate a new landfill.

This study utilizes a meta-analysis for benefits transfer (MA-BT) approach and follows the standard for valid and reliable benefits transfer: (i) welfare measure consistency (e.g., hedonic price methods), (ii) commodity consistency (e.g., distance from landfills), and (iii) theory consistency (e.g., positive distant effects) (Bergstrom and Taylor 2006; Smith et al. 2002). By focusing on hedonic price method (HPM) studies for landfill sites, the current research develops meta-analysis (MA) functions which have relationships between marginal willingness to pay (MWTP) for distance and explanatory variables such as core economic variables (e.g., income levels and population densities) and study design variables (e.g., the number of observations, standard errors of each selected study, and functional form). The MA functions are utilized to measure social costs for the selected sites and to rank these sites. For a comparison purpose, the mean transfer value (MTV) approach is also utilized.[6] The estimated MA models are evaluated in terms of sensitivity, validity, and reliability criteria.

Conducting a primary hedonic price method (HPM) and a contingent valuation (CVM) (e.g., multiple proposed sites) is not desirable for this project designed to develop and test a method for preliminary analysis, which has several reasons (e.g., inaccessible data, short time frame, and little money). Conduction HPM is difficult to find reliable findings from proposed sites without the existing landfill, and implementing CVM can measure external costs for each target site but requires substantial time and money. If researchers and planners have enough time and money, they can conduct primary CVM for each target site. If not, this preliminary analysis is much better than no framework.

## 1.6. Contributions of the study

In terms of contributions to the frontier of knowledge, this research stretches over the following fields: GIS, natural resources and environmental management, and resource and environmental economics, and related subject areas.

This study contributes to the frontier of knowledge by developing an integrative approach-linking a GIS analysis with a meta-analysis for benefits transfer (MA-BT) approach. Although GIS has been widely used for landfill site selection (Siddiqui et al. 1996; Baban and Flannagan 1998; Kontos et al. 2003, 2005; Daneshvar et al. 2005, Sener et al. 2006; Hasan et al. 2009), there is a tendency to disregard economic analysis. Few studies (Swallow et al. 1992; Opaluch et al. 1993) have linked a GIS analysis with an economic framework for landfill site selection. They employed a GIS for data collection and utilized a direct survey method to evaluate public preferences for landfill site selection. However, the direct survey method requires substantial time and a large

[6] Generally, meta-analysis transfer functions perform better than mean transfer value approaches because of fitting characteristics (e.g., income levels and population densities) of target sites (Rosenberger and Loomis 2003).

monetary budget. Few studies (Brisson and Pearce 1995; Walton et al. 2006; Ready 2005) have utilized meta-analysis (MA) in order to examine factors that affect different impacts of landfills on housing values. However, they did not employ meta-analysis for landfill site selection. To the author's knowledge, this is the first study that links a GIS analysis with a meta-analysis for benefits transfer (MA-BT) approach to analyze landfill site selection.

This research is also the first formal study on landfill site selection to the island of Oahu which: (i) compared the study results with those of a recent City and County of Honolulu study, (ii) evaluated the process of landfill site selection, and (iii) checked results in terms of sensitivity, reliability, and validity criteria. As will be shown later, this integrative approach helps to evaluate and illustrate the process of site selection.

## 1.7. Outline of following chapters

Chapter 2 provides: (i) definitions of terms and basic concepts involving waste management and the landfill site; (ii) anticipated impacts of landfills on the environment and nearby community (e.g., physical, chemical, and biological degradation); (iii) a discussion of the proper operation and effective management necessary to mitigate these adverse impacts; (iv) the landfill site selection process with the use of a GIS including a process of site selection and the exclusionary criteria for landfill site selection; (v) environmental justice and participation; and (vi) economic valuation methods (e.g., hedonic price methods, contingent valuation methods, and benefits transfer methods) to measure external effects from landfills.

Chapter 3 provides the proposed integrative methodology that links a GIS-based analysis with an economic framework: (i) a GIS-based screening of possible landfill sites that satisfy constraints and (ii) an economic framework (e.g., benefits transfer methods) that ranks the remaining selected sites according to social cost minimization. This chapter discusses: (i) the procedure and data used of the GIS-based preliminary screening; (ii) the process and data utilized for benefits transfer methods (e.g., meta-analysis and mean transfer value approaches); and (iii) the procedure to check validity and reliability of benefits transfer (BT) methods.

Chapter 4 presents results of this study. As will be shown later, the integrative approach was implemented for the island of Oahu regarding landfill site selection. The GIS-based analysis was applied to: (i) the City's 45 potential sites (Scenario 1) and (ii) the entire island of Oahu (Scenario 2). Together, both scenarios found a total of 7 sites that satisfied the given criteria. Benefits transfer (BT) methods (e.g., meta-analysis and mean transfer value approaches) were utilized to measure social costs and to rank the selected 7 target sites. The results clearly demonstrate that social costs are high and vary by target sites due to different market conditions, different methods used, different models used, and differences in discount rates (lengths of landfill life in years). In terms of sensitivity, reliability, and validity criteria, meta-analysis models are preferred to a mean transfer value approach.

Chapter 5 provides: (i) a review of this study, (ii) policy implications, and (iii) conclusions and discussion.

# CHAPTER 2. LITERATURE REVIEW

## 2.1. Waste management and landfill

Before discussion about anticipated impacts of landfills, proper operation and effective management, and landfill site selection, this section defines terms and describes basic concepts involving: (i) municipal solid waste (MSW) and sanitary landfill, (ii) the integrated waste management system, and (iii) landfill design.

### 2.1.1. Definition of terms

Solid waste is frequently categorized into municipal solid waste (MSW) and other solid waste. Municipal solid waste (MSW) is referred to as every day waste generated from households and small businesses (e.g., small shops, schools, and institutions).[7] Municipalities usually provide a disposal service for MSW at subsidized rates. Other solid waste is referred to as putrescent matter or large quantities of waste produced by businesses including construction & demolition (C&D) debris and industrial waste. Businesses dispose of it themselves or hire a private disposal service to remove it (Porter 2002).

A landfill is a physical facility used for solid waste disposal into the land. Formerly, the term sanitary landfill indicated a facility in which the waste was enclosed by the use of daily cover. Today, the sanitary landfill is the engineered facility for MSW disposal that is designed and operated in order to minimize public health concerns from environmental pollution (Tchobanoglous et al. 1993). Current research focuses on the sanitary landfill for MSW disposal.

---

[7] Despite its variations due to local and regional factors, such as climate and level of commercial activity, residential waste is 55 to 65 percent of total MSW generation. Commercial waste constitutes between 35 and 45 percent of MSW (U.S. EPA 2007).

Definitions of some terms useful for discussing waste disposal in a sanitary landfill are as follows (Tchobanoglous et al. 1993 p. 362-363):

> *"The term cell is used to describe the volume of material that is deposited in a landfill during one operating period (usually one day) (see Figure 2.1). The cell contains solid wastes and daily cover materials with 6 to 12 inches (15.2 cm to 30.5 cm) of native soil or alternative materials such as compost in order to enclose the working faces of the landfill at the end of each operating period. Daily cover controls the windblown litter and water infiltration into the landfill during operation and prevents pests (e.g., rats, flies, mice) and other disease vectors from approaching the landfill. Cover is applied to the entire landfill surface after all waste disposal operations are completed. The final cover consists of multiple layers with 6 to 10 feet (182.88 cm to 304.08 cm) of soil and/or geomembrane materials."*

Emissions of landfill gas and leachate can result in various environmental impacts (e.g., global warming, air pollution, groundwater pollution, and disamenities) Kreith et al. (2002 p. 670-672) defined these landfill gas and leachate below:

> *"Landfill gas is the mixture of gases that are generated within a landfill. The bulk of landfill gas consists of methane and carbon dioxide, the principal products of the anaerobic process of the MSW in the landfill. Other components of landfill gas contain nitrogen, oxygen, and ammonia. Leachate refers to the liquid that occurs when any water contacts with waste. Leachate results from the percolation of precipitation, uncontrolled runoff into the landfill, groundwater intrusion, and water initially contained in the waste. Leachate contains various chemical constituents derived from the dissolution of the materials in the landfill and from the products of chemical and biochemical reactions occurring within the landfill."*

### 2.1.2. Integrated solid waste management (ISWM)

Solid waste management is the discipline to control generation, storage, collection, transfer and transport, processing, and solid waste disposal in a manner that follows the best principles of public health, economics, engineering, conservation, aesthetics, environmental considerations, and response to public attitudes (Kreith et al. 2002). Integrated solid waste management (ISWM) is to select and apply suitable technologies and management programs in order to achieve specific waste management objectives and goals such as source reduction, recycling, waste transformation, and landfill disposal (Tchobanoglous et al. 1993). Tchobanoglous et al. (1993 p. 16) defined these elements as follows:

> *"Source reduction involves decreasing the amount of waste and toxicity at the source of waste. Reduction is the most effective way to reduce the quantity of waste, the cost associated with its handling, and its environmental impacts. Waste reduction may occur through the design, manufacture, and packaging of products with minimum toxic content and minimum volume of material. Recycling involves: (i) the separation and collection of waste materials; (ii) the preparation of these materials for reuse, reprocessing, and remanufacture; and (iii) the reuse, reprocessing, and remanufacture of these materials. Recycling assists in reducing the amount of waste for landfill disposal. Waste transformation involves the physical, chemical, or biological change of waste. The change is: (i) to improve the efficiency of solid waste management operation and systems; (ii) to recover reusable and recyclable materials; and (iii) to recover energy in the form of heat and combustible biogas.[8] The use of landfills is the most common method for waste disposal. Landfills handle (i) the solid waste that cannot be recycled and are no further use; (ii) the residual after solid waste has been separated at a material recovery facility; and (iii) the residual from waste to energy recovery."*

The integrated solid waste management (ISWM) approach evolved as a response to the regulations or laws. For example, the Hawaii Revised Statutes (HRS), Chapter 342G requires that each county manage an ISWM plan once every five years. The Oahu County Refuse Division of the Department of Environmental Services prepared a 25 year plan with the main objective to maximize the recovery of solid waste and extend the life of the existing landfill in an environmentally safe manner (City & County of Honolulu 2008a). Landfills are inevitable in its plan because alternative methods (e.g., source reduction, recycling, and waste to energy recovery) produce untreated waste that are disposed of in landfills.

[8] For example, the H-POWER plant for the City & County of Honolulu has been trying to reduce the volume of waste for landfill disposal by transforming combustible solid waste into energy.

### 2.1.3. Landfill design

There are two major concepts of landfill design: (i) natural attenuation landfills and (ii) modern sanitary landfills. While the natural attenuation landfill concept does not have liners, the modern sanitary landfill contains liners to prevent escaping leachate from contaminating groundwater (McBean et al. 1995). The Federal Resource Conservation and Recovery Act (RCRA) Subtitle D of 1976 requires that sanitary landfills constructed after 1979 include liners (Kreith et al. 2002).

The natural attenuation (NA) landfill design concept allows leachate to percolate through the landfill base to the neighboring environment with the expectation that the leachate becomes purified by the natural geology (e.g., soil) and hydrogeology (e.g., groundwater aquifer). Historically, natural attenuation (NA) landfills were common due to their relatively lower cost of construction, operation, and management and lower environmental concerns (Bagchi 1994). However, this method had a high possibility of contaminating neighboring environment, especially groundwater. Recently, the regulatory trend is toward containing and removing leachate before it contaminates the environment (McBean et al. 1995).

Sanitary landfill is scientifically designed and operated in order to prevent environment and humans from the harmful landfill gas and leachate including liner systems, cap (final cover) systems, gas management systems, leachate management systems (Tchobanoglous et al. 1993). Figure 2.1 shows the typical sanitary landfill.

Landfill liners prevent leachate from escaping the surrounding environment especially, groundwater. The liner is sloped to direct leachate to the leachate collection system through which leachate is collected and moved to a treatment system. Subtitle D of RCRA requires that new sanitary landfills include composite liners (a combination of compacted clay and geomembrane materials) separated by a drainage layer (Tchobanoglous et al. 1993).

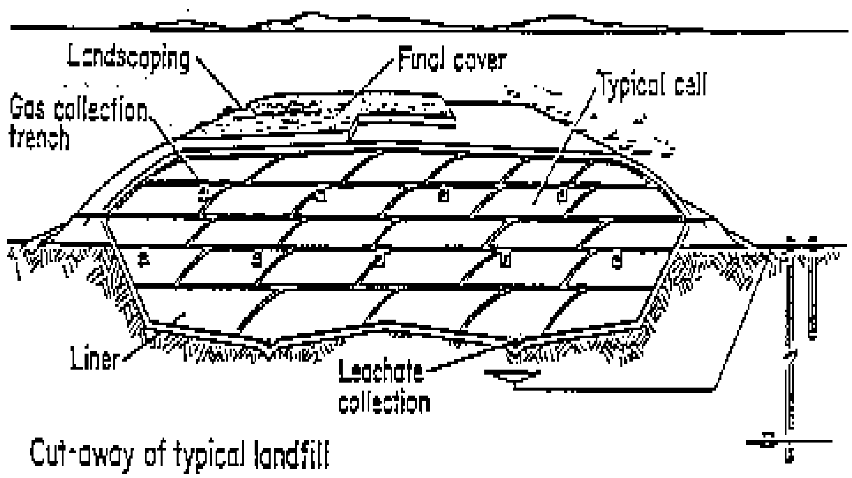



Figure 2.1. Typical sanitary landfill
Source: Tchobanoglous et al. (1993)

The primary purpose of the final cover design is to minimize leachate generation arising from percolating precipitation, to convert the infiltration into surface runoff, and to contain landfill gas for proper management. Additional objectives are to prevent approaching pests (e.g., birds, rats, mice, and flying insects) and the blowing of litter or dust onto nearby neighbors. As a system of multiple components, the final cover contains two primary layers: (i) the surface layer; and (ii) the hydraulic barrier layer. As a barrier layer, low-permeable soil materials (e.g., clay) or/and non-soil materials (e.g., geomembranes) are utilized for minimizing water infiltration through the cover. Topsoil provides suitable soil for vegetation growth and reduces water infiltration and wind erosion. As topsoil, medium-textured soil (e.g., loam soil) has the best overall characteristics for seed germination and plant root system development (McBean et al. 1995).

Gas management systems prevent gas migration away from the landfill boundaries or gas emissions through the landfill surface. The systems include gas collection wells, gas conveyance piping, and gas recovery treatments. Gas collection wells extract landfill gas, and conveyance piping moves the gas to a processing area for proper destruction (e.g., gas flares) or beneficial reuse (e.g., electric generation or compressed natural gas) (McBean et al. 1995).

Leachate management systems control leachate contaminating the nearby environment (e.g., soil and groundwater) including leachate collection pipes, leachate conveyance piping, storage tanks, and onsite treatments. Leachate collection pipes in the liner capture leachate, and conveyance piping moves the leachate to a central storage and treatment area. The treatment system conveys the leachate to a nearby stream or a local sanitary sewer system for further treatment or beneficial reuse on site (McBean et al. 1995).

Even highly engineered landfills incur environmental pollution. The next section will discuss impacts from landfills on nearby residents and environment.

## 2. 2. Impacts of landfills on environment

Landfills generate landfill gas and leachate that result from waste degradation. These landfill gas and leachate incur various environmental impacts (e.g., physical, chemical, and biological degradation). This section will discuss related issues regarding (i) landfill gas and leachate and (ii) anticipated impacts of landfills (e.g., physical, chemical, and biological degradation).

### 2.2.1. Landfill gas and leachate

#### 2.2.1.1. Biodegradation in landfill

Landfill gas and leachate are closely related to biodegradation in landfill. Understanding the nature of waste degradation can help to control environmental degradation (Westlake 1995). Wastes in a landfill are composed of organic materials (e.g., paper and food wastes) and inorganic materials (e.g., glass, cans, and metals). Despite the variability in its composition, organic materials constitute a high proportion of total wastes (around 80%). Aerobic and anaerobic processes break down most organic matter into simpler compounds (El-Fadel et al. 1997).

When waste is disposed of in a landfill, oxygen is present. During aerobic decomposition, biodegradable organic matter reacts quickly with oxygen to form carbon dioxide, water, and other by-products. Aerobic decomposition is characteristically rapid (e.g., days or months) due to the limited availability of oxygen in the landfill. Oxygen depletion within the landfill leads the anaerobic decomposition phase, which dominates the waste decomposition process and contributes significant gas formation (EL-Fadel et al. 1997).

Major waste degradation steps in the anaerobic decomposition phase including hydrolysis, acidogenesis, acetogenesis, and methanogenesis are shown in Figure 2.2. Generally, organic matter in anaerobic ecosystems breaks down the complex to the simple starting with the hydrolysis of complex matter (e.g., proteins, carbohydrates, and lipids) to simpler polymers (e.g., amino acids, sugars, and high molecular fatty acids). Amino acids and sugars are either converted into intermediate by-products (e.g., propionic, butyric, and other volatile acids) or directly fermented to acetic acid. High molecular fatty acids are oxidized to intermediate by-products or directly fermented to acetic acid. Methane and carbon dioxide are generated primarily through acetate cleavage. Methane is produced thorough carbon dioxide reduction with hydrogen (El-Fadel et al. 1997).

The key acetate cleavage in the methanogenic stage is $CH_3COOH \rightarrow CH_4 + CO_2$. The reaction of the use of hydrogen is $4H_2 + CO_2 \rightarrow CH_4 + 2H_2O$. These two reactions in the methanogenic stage produce energy that can act as a source of potential renewable energy (McBean et al. 1995).

The process of waste decomposition affects the composition and/or characteristic of landfill gas and leachate. Thus, an understanding of the process will help to identify the associated problems and manage them.

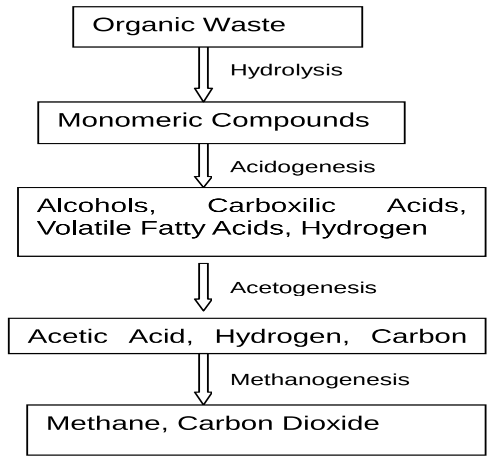



Figure 2.2. Major waste degradation steps during the anaerobic decomposition phase Source: El-Fadel et al. (1997)

#### 2.2.1.2. Landfill gas

The composition of landfill gas varies according to waste characteristics, moisture content, and age of landfill, but under a stabilized methanogenic condition, by volume, landfill gas typically contains 45-60% methane, 40-60% carbon dioxide, and small components of nitrogen, oxygen, ammonia, sulfides, hydrogen, and nonmethane organic compounds (NMOCs) (ATSDR 2001). Methane and carbon dioxide are major components in landfill gas (over 90% of the total landfill gas). Methane is colorless, odorless, and flammable with a density lighter than air. Carbon dioxide is colorless, odorless, but non-flammable with a density denser than air. Sulfides and ammonia incur unpleasant odors even at very low concentrations less than 1% (City & County of Honolulu 2008) (see Table 2.1).

Nonmethane organic compounds (NMOCs) are composed of mostly volatile organic compounds (VOCs) including hazardous air pollutants such as vinyl chloride, ethyl benzene, toluene, and benzene. Significant concentrations of VOCs in landfill gas occurred in older landfills that accepted industrial and commercial waste with VOCs. After banning co-disposal of the industrial and commercial waste in MSW landfills, the concentration of VOCs tends to decrease (Kreith et al. 2002).

Table 2.1. Typical Landfill Gas Components and Characteristics

| Components | Percent (Volume) | Characteristics |
|---|---|---|
| Methane | 45–60 | Methane is colorless, odorless, and flammable. Landfills are the single largest source of U.S. man-made methane emissions. It is lighter than air. |
| Carbon Dioxide | 40–60 | Carbon dioxide is colorless, odorless, non-flammable, and slightly acidic. It is denser than air. |
| Nitrogen | 2–5 | Nitrogen is odorless, tasteless, and colorless. |
| Oxygen | 0.1–1 | Oxygen is odorless, tasteless, and colorless. |
| Ammonia | 0.1–1 | Ammonia is a colorless gas with a pungent odor. |
| NMOCs | 0.01–0.6 | NMOCs (Non-Methane Organic Compounds) are organic compounds (i.e., compounds that contain carbon). NMOCs may occur naturally or be formed by synthetic chemical processes. |
| Sulfides | 0–1 | Sulfides (e.g., hydrogen sulfide, dimethyl sulfide, and mercaptans) can cause unpleasant odors even at very low concentrations. |
| Hydrogen | 0–0.2 | Hydrogen is an odorless, colorless gas. |
| Carbon monoxide | 0–0.2 | Carbon monoxide is an odorless, colorless gas. |

Source: ATSDR (2001), Tchobanoglous et al. (1993), and City & County of Honolulu (2008)*
*The WGSL site flares (including all wells) composed of approximately 44% methane, 40% carbon dioxide, and small amounts of carbon monoxide, nitrogen, oxygen, hydrogen, and sulfides.

The migration of landfill gas in a landfill is a significant consideration due to the associated problems (e.g., health hazards, vegetation stress, and odors) (McBean et al. 1995). During landfill operation prior to final capping, landfill gas will move upward because of the natural tendency of landfill gas such as methane with a lighter density than air. Landfill gas vents through the cover to the atmosphere if daily cover is permeable. After final capping, densely compacted waste and final cover inhibit vertical movement of landfill gas. The restricted vertical movement enables the gas to start horizontal movement within the landfill. Landfill gas such as carbon dioxide with a higher density than air accumulates in the bottom of landfill. Thus, landfill gas at the bottom of the landfill is collected and treated for proper destruction or beneficial reuse (Westlake 1995).

The migration of landfill gas is affected by climate (precipitation and wind) and landfill cover types. If landfill gas vents through the landfill cover, wind carries gas naturally into the air and dilutes the gas with fresh air. Wind speed and direction determine the concentration of gas in the air, which can vary greatly from day to day, even hour by hour. For example, in the early morning, mild winds provide the low dispersion of the gas to other areas. Precipitation enhances lateral migration since it causes surface materials within landfill caps to swell and close surface cracks (Westlake 1995). Good compaction and an impermeable final cover (e.g., natural soil like silt and clay or geosynthetic materials) inhibit upward movement of landfill gas and enhance horizontal migration within the landfill (ATSDR 2001).

Computer models predicted the approximate migration patterns from existing landfills despite the difficulty in predicting the distance of the travelling landfill gas. Various factors (e.g., soil conditions, waste compositions, and designs) may affect its underground migration. A study conducted by the New York State Department of Health found that of the 38 landfills, gas migrated underground up to 1000 feet (or 304.8 meters) at 1 landfill, 500 feet (or 152.4 meters) at 4 landfills, and 250 feet (or 152.4 meters) from the landfill boundary at 33 landfills (ATSDR 2001).

#### 2.2.1.3. Leachate

Leachate occurrence is one of the most significant threats to the surrounding environment, especially, groundwater. Water movement in a landfill affects leachate occurrence that results from: (i) the initial waste moisture content (primary leachate); and (ii) the water added to the landfill from precipitation, uncontrolled runoff, and groundwater intrusion (secondary leachate) (See Figure 2.3). Generally, in the wet climates, secondary leachate is dominant in the landfill. Leachate reaches the bottom of the landfill and moves through the base of the landfill or into the subsurface (McBean et al. 1995).

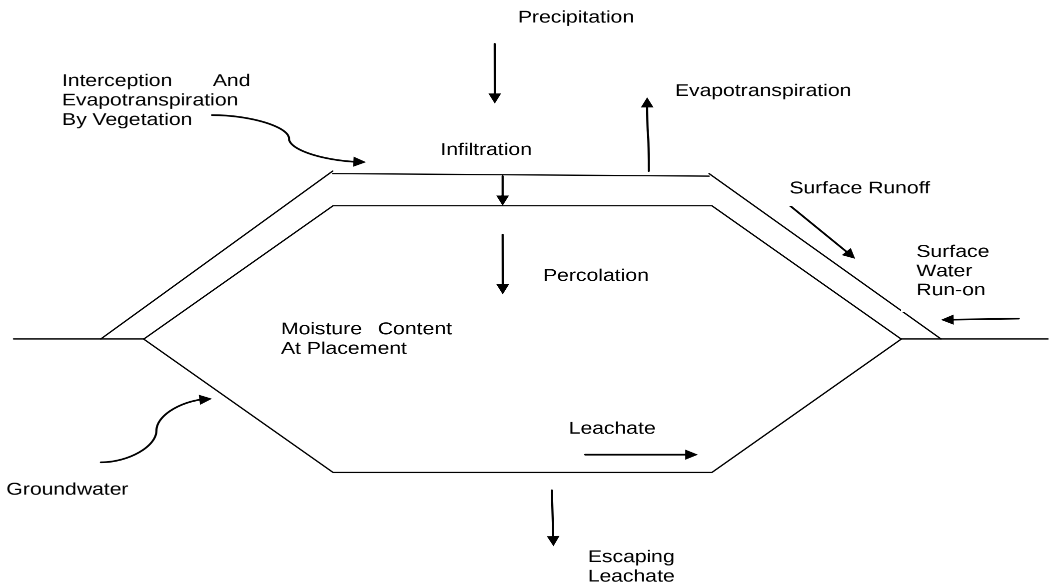



Figure 2.3. Water movement at a landfill
Source: McBean et al. (1995)

Various chemical reactions within a landfill occur as follows: (i) dissolution of materials and biological conversion products in the liquid percolation through the waste; (ii) evaporation and vaporization of chemical compounds and water into the evolving landfill gas; (iii) sorption[9] of volatile organic compounds (VOCs) into the waste material; (iv) the decomposition of organic compounds; and (v) the solubility of metal salts. The dissolution of biological compounds and other compounds (especially, organic compounds) is of special importance because escaping leachate may transport harmful materials (e.g., VOCs and heavy metals) out of the landfill and contaminate nearby environment (Tchobanoglous et al. 1993).

The quality of leachate varies with time and across landfills (or cells) because of different water infiltration rates, types of MSW, and age of landfills (McBean et al. 1995). Table 2.2 shows typical chemical concentrations in leachate over time.

---

[9] Sorption refers to the action of both absorption and adsorption taking place simultaneously. Absorption is the incorporation of a substance in one state into another of a different state (e.g., liquids being absorbed by a solid or gases being absorbed by a liquid). Adsorption is the physical adherence or bonding of ions and molecules onto the surface of another molecule.

Table 2.2. Typical Chemical Concentrations in Landfill Leachate over Time

| Parameter | Young Leachate Concentration (mg/L) (the first few years) | Old Leachate Concentration (mg/L) (after 4 to 5 years) |
|---|---|---|
| Chemical oxygen demand (COD) | 20,000-40,000 | 500-3,000 |
| Biological oxygen demand measured over 5 days ($BOD_5$ ) | 10,000-20,000 | 50-100 |
| Total organic carbon (TOC) | 9,000-15,000 | 100-1,000 |
| Volatile fatty acids (as acetic) | 9,000-25,000 | 50-100 |

Source: McBean et al. (1995)

During the initial phases in the landfill, leachate contains high concentrations of organic carbon (EC 2000). TOC, BOD, and $\mathrm{BOD}_5$ measure the amount of carbonaceous materials. Organic carbon in the leachate tends to decrease over time. In the first few years, young leachate tends to be acidic due to the high concentrations of volatile fatty acids (the pH stays in the range of 6 to 7). COD, TOC, and $\mathrm{BOD}_5$ are high in the young leachate. With time (after 4 to 5 years), the concentrations of volatile fatty acids, COD, TOC, and $\mathrm{BOD}_5$ significantly decrease (the pH increase to the range of 7 to 8) (McBean et al. 1995).

### 2.2.2. Contribution of landfills to physical, chemical, and biological degradation

#### 2.2.2.1. Physical degradation

Soil erosion

The soil condition is one of the best indicators of physical degradation onsite because soil influences vegetation, runoff, infiltration, and land management. Erosion by water (precipitation and runoff) is affected by some factors: (i) erosivity (the potential of precipitation to cause erosion such as intensive precipitation in a short period); (ii) erodibility (soil property); (iii) land cover (natural vegetation that protects the soil); and (iv) landfill management (landfill use) (Barrow 199).

During operation of the landfill, waste is disposed of and compacted in a landfill before the use of daily cover encloses the working space (Tchobanoglous et. al. 1993). The properly compacted waste can obtain a field density of 475 kg/$m^3$(800 lb/$yd^3$). High in-place density of waste obtained by good compaction and the use of a daily cover can reduce (i) voids in the refuse that water infiltrates, (ii) windblown litter, and (iii) the possibility that waste will wash away during heavy precipitation (McBean et al. 1995).

Once landfill operation completes, final cover encloses the top of the landfill. The use of impermeable soil (e.g., silt and clay) or synthetic cover material can reduce water infiltration but enhances runoff. A primary concern is the slope of the site that promotes runoff without causing severe erosion of the final cover (3-5% is ideal) (McBean et al. 1995).

After the final cover, revegetation may prevent soil erosion and increases the structural stability of the landfill cover by creating a leaf layer above the soil that reduces the kinetic energy of precipitation and decreases the wind and water runoff velocity (McBean et al. 1995). An important consideration is to select the appropriate vegetative species which are indigenous to the area, hardy, and drought resistant. However, during the operation, vegetation is removed in order to prevent offsite movement of pollutants (City & County of Honolulu 2008b).

### Disamenities

Disamenities are localized nuisances (e.g., litter, dust, and noise) from landfills, which concentrate on nearby residents (for the economic measurement, see Section 2.5 of this study). Problems associated with disamenities usually occur during landfill operation. Nearby residents showed significant concerns about windblown litter, dust, and smell (Garrod and Willis 1998).

- Litter: Uncontrolled litter incurs unacceptable appearance and provides a food source for vermin and flies. Dry and windy weather worsens the litter problem (McBean et al. 1995).

- Dust: Traffic on roadways and operation of equipment create dust that contributes to allergies, nuisances, and accidents onsite and in nearby communities. Strong winds worsen these problems (McBean et al. 1995).
- Odors: The release of uncontrolled landfill gases causes odors. The presence of small concentrations of sulfide (0-1% by volume) and ammonia (0.1-1% by volume) can contribute to odors. Sulfides produce a strong, rotten-egg smell that humans can detect even at very low concentrations. Ammonia produces a pungent odor. Humidity worsens the problems associated with odors (Westlake 1995).
- Noise: Noise occurs from onsite activities such as operating equipment and vehicle movements. Noise may increase stress and cause hearing impairment to on-site personnel (Westlake 1995).
- Pests (e.g., carrion eating birds, rats, and mosquitoes): A significant proportion of organic matter in waste (about 80%) increases pests that may spread diseases. For example, rats carry the plague disease, and mosquitoes spread malaria. Gulls can be hazards to aircraft at landfill sites situated near airports (Westlake 1995).

Offsite effects

High blown wind carries dust and litter to offsite communities and ecosystems, especially downwind or in the direction of the prevailing wind (Barrow 1992). Landfill construction can alter geology e.g., increased slopes and redirection of drainage, which may increase off-site flows and sediments. Uncontrolled runoff transports sediments to sensitive receptors e.g., water bodies and wetlands (US EPA 1993). A release of leachate may incur risks to human health and nearby environment e.g., groundwater, low-lying land, streams, and rivers (Barrow 1992). Of great concern in landfill management is the threat to groundwater. Over 25 % of the Superfund sites listed on the National Priority List in the United States were MSW landfills (El-Fadel et al. 1997).

#### 2.2.2.2. Chemical degradation

Explosion

Migrating gas finds its way into buildings and underground facilities erected on or near a landfill site, which can form gas pockets that contribute to creating potential explosive hazards. Methane within the 5-15% range by volume is flammable when mixed with oxygen. With little danger of explosion in the anaerobic decomposition phase, methane mixtures mixed with air are liable to fire or explosion (Tchobanoglous et al. 1993). Methane gas control levels are set based on the 5-10% flammable range by volume (Westlake 1995).

Impacts on vegetation

The lack of oxygen damages plants. Uncontrolled landfill gas migrates upward and escapes into the atmosphere by venting through the landfill cover. This process displaces oxygen and enables plant roots to an increase in exposure to high concentrations of methane and carbon dioxide. The oxygen deficiency causes plant death and asphyxia (El-Fadel et al. 1997).

Methane oxidation near the surface can cause oxygen deficiency. Although methane may not directly affect plant growth, methane oxidation creates carbon dioxide, water, and heat. Heat released during methane oxidation increases soil temperature that can create plant asphyxia (Stocking et al. 2001). Particularly high concentrations of landfill gas and carbon dioxide generated from methane oxidation (30 to 45%) can be harmful to plant growth (El-Fadel et al. 1997).

Offsite effects

Global climate change: Landfills may contribute to global climate change. The greenhouse effect is the warming of the Earth’s atmosphere by the accumulation of greenhouse gases (e.g., methane, carbon dioxide, nitrogen oxide, and sulfur oxide) which absorb reflected solar radiation. Global warming can result in negative long-term consequences (e.g., extreme climate change and sea level rise) (Westlake 1995). Landfill gases including carbon dioxide and methane contribute to global warming. Methane especially has received recent attention as a contributor to global warming because of molecular structure. Methane ( $CH_4$) has a relative effect 20-25 times greater than carbon dioxide ( $CO_2$) because it is more effective to trap infrared radiation in the atmosphere (El-Fadel et al. 1997). Methane contributes about 18% of total global warming effects (Church and Shepherd 1989). Methane contributes about 500 million short tons (453.5 metric tons) per year of which sanitary landfills emit about 40 million short tons (36.28 million metric tons) to 75 million short tons (68.038 million metric tons) (Sheppard et al. 1982; Bingemer and Crutzen 1987; Pearce 1989). Unless recovery control systems are implemented, landfills can become a significant contributor to methane due to the continued trends in population increase and urbanization (El-Fadel et al. 1997).

Air pollution: Industrial wastes can contribute to the presence of various chemicals in landfill gases with adverse effects. Especially the emission of volatile organic compounds (VOCs) at a very low percentage (less than one percent of landfill gases) can increase cancer risks in local communities and contribute to ambient ozone formation (El-Fadel et al. 1997). Banning industrial waste disposal in a MSW landfill can reduce the health risks.

Groundwater/surface water pollution: As well as leachate, landfill gas may contribute to groundwater pollution. Carbon dioxide has a high possibility of polluting sensitive receptors (e.g., groundwater and surface water) by increasing acidity, corrosivity, and hardness of the water. Trace toxic gases (especially, VOCs) may contaminate groundwater or surface water when they travel in the unsaturated zone and touch with water (EI-Fadel et al. 1997).

#### 2.2.2.3. Biological degradation

Impacts on fauna

Introduced species affect ecology on or near sites. Landfill materials (e.g., putrescible materials) attract rodents (e.g., rats), predators (cats and dogs), and raptors (e.g., hawks, eagles, and owls). The introduced rats, cats, and dogs compete with native rodents and predators. Predatory birds (e.g., hawks, eagles, falcons, and owls) predate native birds. The introduced species often eat native plants and spread pathogens on native fauna and flora (Mueck and Nye 2006).

Impacts on flora

Landfills may change biodiversity of flora species. During landfill operation, native plants on sites are removed. Introduced species (ruderal species that are specific in waste places) dominate plant communities at or near the sites. The recent investigation at the WGSL site showed that only 6 species of a total 50 species were indigenous species (plants native to Hawaii but found elsewhere as natives). After final capping, selection of indigenous species (hardy and drought resistant) for revegetation helps restoration (City & County of Honolulu 2009).

Offsite effects

Uncontrolled runoff or/and leachate may degrade offsite ecosystems especially, groundwater and surface water downstream of the landfill (El-Fadel et al. 1997). If nutrients transported by runoff enter in streams and other watercourses, the surface water is converted to a europhic state in which fish and plants cannot survive due to the lack of dissolved oxygen. The escaping leachate degrades the food web through which common toxic compounds in leachate (e.g., heavy metals and ammonia) are bioaccumulated (Westlake 1995).

Windblown litter or contaminants may degrade the remnant native vegetation and fauna offsite. Some potential issues are as follows (Mueck and Nye 2006):

- Direct deleterious impacts of leakage of soil and groundwater pollutants on amphibians (e.g., frogs and toads) and, via soil invertebrates, on the food chains of other native fauna.
- Entanglement of birds and fauna in litter items (e.g., plastics, fishing line)
- Effects of dust and other wind-blown pollutants on nearby vegetation.

## 2.3. Proper operation and effective management

### Proper operation

Proper operation of a landfill can reduce disamenities (e.g., litter, dust, insects, noise, odor, and visual impairments). Wind-blown litter can be reduced by the use of daily covers, movable screens, and regular cleanup or pickup. The use of daily covers reduces windblown litter, visual impairments, and intrusion of carrion-eating birds, insects, and rats. Small movable screen wire can be placed in downwind locations to catch windblown papers. Regular manual pickup at the end of a working day removes litter (Bagchi 1994).

Dust from landfills often occurs on the dry roads where heavy traffic passes through. Watering of the dry roads (e.g., the use of wheel washers) can reduce dust during dry and windy conditions, but it may increase leachate. Thus, the extent of watering should be monitored (Bagchi 1994).

Odor can be reduced by special handling of odorous material (e.g., food wastes and sewage sludge), the use of daily covers, and the use of odor-neutralizing solvents. For example, the City & County of Honolulu refuse vehicle can identify odorous material. The identified odorous load was discharged into the specific trench or pit. The bulldozer immediately covered the odorous material with excavated non-odorous solid waste. Daily cover soil was placed and compacted above the waste. The regular use of odor-neutralizing solvents and deployment systems reduced the odors (City & County of Honolulu 2008).

Noise can be controlled by: (i) use trees to attenuate noise; (ii) build a buffer zone or barrier between the sources and neighboring residents; (iii) use a suitable separation distance between the active site face and residents; (iv) maintain trucks and landfill equipment; and (v) regulate operation hours compatible with adjacent land uses (McBean 1995).

The introduction of feral animals (e.g., fox, cats, and dogs) and raptors (e.g., hawks, eagles, and owls) can be prevented by good operational practices. The most effective management strategy is to minimize the size of the active tipping face and ensure that the landfill material is fully covered by an adequate depth of compacted clean fill at closure each day. The establishment and maintenance of a predator-exclusion perimeter fencing of the site will further reduce the potential for the introduction of feral animals. Minimization of any area of open water on the site can reduce the number of raptors (Mueck and Nye 2006).

While these disamenities mainly occur during the site operation, degradation related to landfill gas and leachate may be present throughout or beyond the lifetime of the landfill site. Thus, the control of landfill gas and leachate is of significance in the landfill design, operation, and management.

### Effective management of landfill gas

Effective landfill gas management can minimize the associated problems such as explosion, plant death, global warming, and groundwater pollution. Landfill gas generation can be prevented through controlling waste inputs (limiting organic waste) and through controlling the process of waste decomposition (minimizing moisture content to limit gas production). However, these preventive methods cannot eliminate landfill gas generation.

In most cases, landfill gas management systems control escaping landfill gas away from the landfill or through the landfill cover. Landfill gas is collected and treated for gas flares or beneficial reuse (U.S. EPA 1993). Flaring of landfill gas is thermal destruction. Modern flaring facilities meet rigorous operating requirements because of the concerns over air pollution (for typical requirements of air quality, see Kreith et al. 2002). Landfill gas can be converted to electricity. Energy from landfill gas (especially, methane) has been identified as a potential renewable energy. It was estimated that annual energy generation from landfills in the United States represented 1% of the total energy needs or 5% of the natural gas utilization in the United States (El-Fadel et al. 1997).

Monitoring landfill gas requires observation of whether landfill gas management systems are operating properly. Routine monitoring for methane and NMOCs are required in Subtitle D of RCRA (Westlake 1995).

Effective management of leachate

Once leachate enters groundwater, it is difficult and expensive to restore groundwater quality. The best way of controlling pollution by leachate is to prevent escaping leachate from reaching the groundwater. Leachate management systems are commonly used to control leachate migration that can contribute to onsite and offsite degradation. Typically, a clay or synthetic liner is used to minimize leachate formation that results from water infiltration or groundwater intrusion. Landfill cover is designed to reduce water infiltration but enhance runoff. Runoff is collected through drainage channels constructed at the surrounding edge of the landfill and discharged to detention ponds. The structure can prevent stormwater from flowing onto the landfill from surrounding areas and runoff or sedimentation impacts to neighboring environment (US EPA 1993).

When leachate reaches the bottom of the landfill, it is collected and eliminated onsite or transported offsite for treatment (the local sanitary sewer systems). Treatment must meet the drinking water quality standard, which are set to prevent harm to public health. Groundwater monitoring wells ensure whether the liner and leachate management systems are operating properly (Kreith et al. 2002).

Even well designed, operated, and managed landfills may incur environmental contamination. Proper site selection can reduce these environmental concerns. For example, a proposed landfill should not be located in the environmentally and ecologically sensitive areas (e.g., floodplains, wetlands, groundwater, and critical habitats) and unstable areas (e.g., seismic impact zones and fault areas).

## 2.4. Landfill site selection

Site selection is the process of finding sites that satisfy specified conditions or restrictive criteria (Heywood et al. 2002). Despite the complicated process, the process of site selection is divided into a preliminary screening phase to eliminate unsuitable areas as landfill sites according to the restrictive criteria and a further assessment phase to evaluate environmental impact statements, communality involvement, and further technical analysis for hydrogeology characteristics that include geologic formations, groundwater depth, drainage, and flow directions (Daneshvar et al. 2005). A further assessment phase is beyond the scope of this research. The preliminary screening process determines or selects suitable sites fulfilling the restrictive or exclusionary criteria.

### 2.4.1. Terms for geographic information system (GIS) analysis

Before discussing the related issues regarding landfill site selection, it is important to clarify terms used in the GIS-based analysis. First, the term geographic information system (GIS) is a computer system including software, hardware, spatial data, management and analysis procedures, and people (Heywood et al. 2002). Its primary roles involve collecting, sorting, retrieving, managing, and displaying spatially referenced data (Davis 2001). Once a GIS database management system is developed, it can provide efficient means for analyzing site characteristics (Siddiqui et al. 1995).

Objectives are the goals that a project or research will achieve. For example, one objective for this study is to select sites satisfying constraints (e.g., airports, wetlands, floodplains, land use, groundwater, and landfill capacity). Criteria are rules that meet the objective (Kao et al. 1996). Constraints are restrictive or exclusionary criteria mandated by public agencies that eliminate unsuitable sites from further consideration (Kontos et al. 2003). To satisfy the objective, the GIS analysis must comply with the restrictive criteria.

Restrictive criteria act as a preliminary screening process where the use of a GIS can process large amounts of geographic data in a short time and exclude unsuitable landfill candidate sites based on a set of restrictive criteria (Sener et al. 2006). Thus, the GIS-based preliminary screening reduces the effort required for information collection and further processing (Kao et al. 1996).

A layered approach is the most common method to structure GIS digital data incorporating restrictive criteria. Each GIS layer represents different types of geographic features (e.g., soil, groundwater, surface water, and wetland), and the layers can be integrated together (Heywood et al. 2002). The layer method will organize the complexity of the site selection process into a simpler representation.

### 2.4.2. Landfill site selection process

The process of landfill site selection involves selecting sites fulfilling constraints or restrictive criteria. This research will follow the process of utilizing a preliminary screening process adapted from Daneshvar et al. (2003) and Barban and Flannagan (1998). Four steps for landfill site selection are illustrated in Figure 2.4.

The first step is to identify objectives and constraints. The objective of the landfill site selection is to find sites satisfying constraints (i.e., eliminating unsuitable areas from further consideration for economic analysis). The process of landfill site selection should comply with restrictive criteria mandated by federal, state, and county agencies. Before GIS analysis, constraints or restrictive criteria for site selection should be developed.

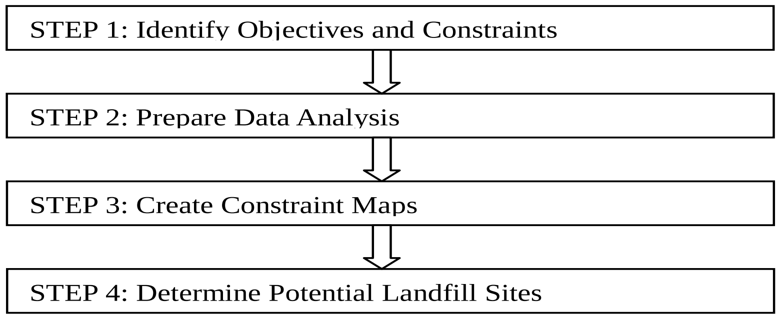



Figure 2.4. Landfill site selection process

Lober et al. (1995) addressed the Not in My Back Yard (NIMBY) phenomenon including social criteria to express opposition to any facility location near one's area. They transformed distance from an individual's residence to the waste facility into a map of opposition attitudes towards the facility. Despite the importance of social criteria, limited data makes the application of social criteria as part of the preliminary screening process difficult. For this study, economic analysis manages social criteria (e.g., income levels, population densities, housing values, and the number of households). A GIS can help data collection by identifying census tract data within a 3-mile distance from each target site.

Economic factors (e.g., direct costs and external costs) are important for site selection (Sener et al. 2006). However, the GIS analysis for this proposed research will not consider these cost factors. Rather, the economic approach incorporates these cost factors. For example, the GIS analysis will select sites fulfilling criteria, and then economic analysis will estimate social costs and rank the selected sites according to the objective of social cost minimization.

The second step is to prepare for data analysis. Once objectives and constraints for site selection are established, corresponding GIS data should be prepared. The availability of data is important for the GIS analysis. Public agencies provide spatial data (e.g., land use, topography, flooding zones, and wetlands) that represent geographic features incorporating restrictive criteria.

Some data can have different formats (e.g., paper maps). Georeferencing and digitizing[10] make the data compatible in a GIS. In order to view multiple layers in a GIS correctly, all layers must be in the same projection and datum (Danesheval et al. 2003). For example, the North American Datum 83 (NAD83) and Universe Transverse Mercator (UTM) in Zone 4 are commonly used for the Island of Oahu.

In the third step, a constraint mapping method is commonly used for identifying whether a proposed landfill site complies with the restrictive criteria mandated by public agencies (Barban and Flannagan 1998; Danesheval et al. 2005). Each constraint map (e.g., binary maps for airports, wetlands, floodplains, groundwater, and land use) can be integrated on each other or in any combination. This constraint mapping procedure helps to identify whether a potential site meets restrictive criteria (Maantay and Ziegler 2006).

Caution should be used for placing equal or different weights on criteria. Some studies (Siddiqui et al. 1996; Kontos et al. 2003; and Sener et al. 2006) on landfill site selection place weights on different restrictive criteria according to their relative importance. However, determining weights are quite controversial and occasionally subjective because different people may view the problem differently (Heywood et al. 2002). Different ordering of criteria can also lead to different results (Luthbom and Lagerkvist 2003). Equal weights for criteria are recommended for preliminary screening because they do not affect results despite different orderings in applying criteria (Daneshvar et al. 2003).

The final step is to find sites fulfilling constraints (exclusionary criteria). Developing a database makes it easier to manage and analyze data and act as a digital bank for the next use (Danesheval et al. 2003).

---

[10] Because scanned maps do not contain location information, geo-referencing assigns coordinates on the image of known locations or other referenced data layer. Digitizing is necessary to add spatial features such as points, lines, or polygons to existing data.

### 2.4.3. Restrictive or exclusionary criteria for landfill site selection

The Federal Resource Conservation and Recovery Act (RCRA) of 1976 established general standards for landfill selection. In the late 1980s, the U.S. Environmental Protection Agency (EPA) published new landfill regulations, known as Subtitle D which includes restrictions for landfill site selection (e.g., restrictions on airports, floodplains, wetlands, seismic impact zones, faults, and geologically unstable areas) (O'Leary and Walsh 2002b). The State Department of Health (DOH), Hawaii delegated and administrated the U.S. EPA's restrictions for landfill selection. The City & County of Honolulu added restrictions on groundwater, land use, and capacity of landfill to the U.S. EPA's restrictions (City & County of Honolulu 2002, 2003). A proposed landfill needs to meet these restrictive criteria.

#### 2.4.3.1. U.S. EPA restrictive criteria

- Airport Restriction. A proposed landfill should be 10,000 feet (3,030 meter) away from the end of any airport runway (used by turbojet aircraft) or 5,000 feet (1,520 meter) away from any airport runway (used only by piston driven aircraft) in order to avoid posing a bird hazard.
- Floodplains. Landfill sites should be out of the 100-year floodplains.
- Wetlands: Landfill sites should not be into wetlands.
- Fault Areas. Developers cannot develop landfill sites within 200 feet (61 meter) of fault areas.
- Seismic Impact Zones. If a landfill site is located in a seismic impact zone, operators must demonstrate that its structures (e.g., liners, leachate collection systems, and water control systems) can resist the effects of ground motion from earthquakes.
- Unstable Areas. Operators must demonstrate that the structure will not be compromised during geologically destabilizing events including: (i) debris

flows resulting from heavy rainfall, (ii) fast formation of sinkholes (caused by excessive ground-water withdrawal), (iii) rock falls, and (iv) the liquefaction of soil after a long period of repeated wetting and drying.

2.4.3.2. Restrictions on groundwater

The State Underground Injection Control (UIC) program and the City Board of Water Supply (BWS) Groundwater Protection Zone (No Pass Zone) exist in order to protect groundwater resources.

- The UIC line. The State of Health (DOH) established the Underground Injection Control (UIC) program in 1984 that intended to protect potable groundwater and surface water from pollution.
- The No Pass Zone (Groundwater Protection Zone). Prior to 1987, the City Board of Water Supply (BWS) identified groundwater recharge area as the No Pass Zone in order to protect groundwater contamination. Since 1987, the State DOH has administrated the No Pass Program.

2.4.3.3. Restriction on land use

- Critical Habitats for threatened and endangered (T&E) species. Operators should not allow landfill sites in the ecological sensitive areas such as critical habitats for endangered and threatened species.
- Developed Areas. The City & County of Honolulu determined that it would not build landfill sites within areas with buildings or residential housing. In addition, urban or build-up land is not suitable for landfill sites because a landfill has negative effects on housing values and human health concerns.

2.4.3.4. Minimum life span (at least 10 Years)

Because of the complicated process and the length of time to permit a new landfill site, potential landfill sites should have at least a 10 year capacity based on the site volume (in cubic yards) and the annual projected disposal volume.

## 2.5. Environmental justice or equity and participation

Environmental justice or equity and participation are important in the process of landfill site selection. Environmental equity is becoming important because all relevant policy appraisal processes (e.g., environmental impact statements) require addressing this issue. When various interest groups involved in the process of landfill site selection participate, the project is more likely to be successful (Pearce et al. 2006).

### 2.5.1. Environmental justice or equity

The notion of environmental justice or equity means that harm from pollutants and facilities that manage industrial, commercial and household waste should be fairly managed for minorities and low-income households (Ringquist 2005). Environmental equity is becoming important because all relevant policy appraisal processes (e.g., environmental impact statements) require addressing this issue (Kreith and George 2002).[11]

The environmental justice literature usually examined a relationship between environmental quality (e.g., concentration of pollution and distance from hazardous waste sites and municipal solid landfills) and demographics (e.g., low-income households and racial minorities) (Banzhaf 2008). Empirical findings about the existence of injustice were mixed (for review, refer to Bowen 2002). The reasons can be explained by differences in: (i) potential environmental risks

[11] Intentional discrimination would violate the equal protection clause of the Constitution and possibly Title VI of the Civil Rights Act, which prevents agencies from receiving federal funding.

that result from different types of facilities (e.g., noxious facilities and landfill sites); (ii) impacted geographic areas (e.g., states, cities, postal ZIP codes, and census tracts); (iii) comparison groups (e.g., race and class); and (iv) study quality (e.g., controlling for explanatory variables) (Ringquist 2005). Through a meta-analysis of 49 environmental equity studies, Ringquist (2005) found ubiquitous evidence of environmental inequities.

In terms of economic analysis, the ultimate social goal is to improve unequal distribution of wealth for impacted residents (e.g., nearby residents) and/or disadvantaged groups (e.g., low-income households) (Banzhaf 2008). While benefits provided with a new landfill site are shared by all citizens, harm from a landfill (e.g., reduction in housing values) concentrates on nearby residents i.e., unequal distribution of wealth. In terms of the Kaldor-Hicks criterion, if winners from a new landfill site compensate losers and there is a still net gain, a new landfill site may improve the social welfare (Roberts et al. 1991; Kunreuther and Kleindorfer 1986; Mitchell and Carson 1986). The concern remains because low-income households usually have fewer choices and less bargaining power. Aid for low-income households (e.g., providing legal service and clear information) can improve inequity. The following economic models (e.g., a hedonic model and a Coasian theorem) can provide insights into public policy regarding environmental equity.

#### 2.5.1.1. Market mechanism

A hedonic model, later discussed in detail, provides important insights into equity issues. Housing markets provide an opportunity for individuals and groups to enhance their welfare, given their limited resources. Unequal distribution related to landfill site selection may be attributed to housing market mechanisms and household's location decisions (Banzhaf 2008). For example, a landfill site is first selected more or less at random based on criteria, and then its presence decreases surrounding house prices. High-income households will escape disamenities from the landfill and to which low-income households then flock to acquire cheap housing despite the disamenities from landfills (Porter 2002). Empirical findings also supported this i.e., willingness to pay (WTP) has a positive relationship with income levels (Reichert et al. 1992; Nelson et al. 1998; Roberts et al. 1991l Smith and Desvousges 1986b). Focusing on improving welfare of impacted households and/or low-income households with low political power and knowledge can manage inequity.

2.5.1.2. Coase theorem and compensation

The Coase theorem provides a framework to understand the design of compensation. With zero transaction costs[12] and fully defined property rights, if contracts between polluters and impacted parties could be negotiated and enforced, trades would be efficient regardless of the initial allocation of property rights. If affected residents have property rights to have an undisturbed neighborhood, providers (e.g., waste sites) should pay compensation to the affected residents, landfill location is determined, ceteris paribus, where its environmental damage would be the least with the lowest compensation. If polluters have property rights to generate disamenities, affected residents should pay to avoid it (Hamilton 1995).

Some studies provided decentralized market approaches: (i) auction mechanisms (Kunreuther and Kleindorfer 1986) and (ii) formal referenda where communities enforce a property right to approve a new landfill (Mitchell and Carson 1986). With well-defined property rights, compensation packages may contribute to improve inequality. Although monetary payments are theoretically preferred to payment in-kinds, Kunreuther and Easterling (1996) found that

[12] A transaction cost means a cost incurred in making an economic exchange. There are different kinds of transaction costs: (i) search costs of locating information about opportunities for exchange; (ii) negotiation costs in terms of the exchange; and (iii) enforcement costs of enforcing the contract (Eggertsson 1990).

monetary payments are viewed as less desirable than in-kind compensation (e.g., state money for schools and road improvements), free garbage collection, reimbursement for medical costs, and property value guarantees. The concern remains when low-income households have fewer choices and less bargaining power. In this case, planners need to focus on reducing transaction costs for low-income households (e.g., providing legal service and information related to transaction) (Banzhaf 2008).

This preliminary study did not manage the equity issue because this study assumed that: (i) unequal distribution is attributed to housing markets and household's decision, and (ii) environmental injustice or inequity can be reduced through compensation for impacted households and aid to reduce transaction costs for low-income households (e.g., providing legal advice and information).[13] Managing this equity issue is not desirable for this preliminary study, which has several reasons (e.g., lack of political power, need for rapid decision-making, and lack of time and monetary budget).

### 2.5.2. Participation

Participation is important in the process of landfill site selection because lack of participation can incur strong opposition to a proposed landfill and delay the process or increase costs. However, when various interest groups involved in the process of landfill site selection participate, a project is more likely to be successful (Pearce et al. 2006). For environmental and social justice, planners should incorporate minority populations (e.g., race, color, and gender) and lower-income communities in the site selection process and provide clear information related to the process of site selection (see O'Hare et al. 1983; Kreith and George 2002).

---

[13] By releasing public reports or news coverage regarding the justification for landfill sites, planners can decrease risk perception (Gayer and Viscusi 2002).

Because individuals or groups have different interests, values, cultures, technical knowledge, and risk perceptions, levels of participation in the decision are different (Ortolano 1997):

- Proximity: Nearby residents are concerned about public health and decreases in housing values.
- Economic impact: Individuals or groups such as land developers have a strong interest in the effects of the facility on economic development.
- Users: Prospective users of a facility can feel their health risks.
- Social and environmental issues: Citizens may be concerned about the facility's effect on social and environmental justice or equity and risks to people and the environment.
- Values: Individuals or groups have different values.
- Legal mandates: Governmental agencies at the local and state levels play the most significant roles in landfill selection; however, federal agencies may become participants depending upon the issues involved.

The size and composition of the involved interested parties can change through stages of the process of landfill selection (Walsh and O'Leary 2002). For example, the Committee on landfill site selection in City and County of Honolulu (2003) was initially composed of 15 members including various individuals (e.g., technical consultants, planners, businesspersons, planners, and community leaders). However, four members resigned against the vote to exclude the existing landfill site. A recent Committee (under Mayor Peter Carlisle) was composed of 9 members of which 8 members differed from those of the prior Committee.

Planners and decision makers need to consider that public (various citizens) involvement in the process of site selection for a successful project. US EPA (1995) provided the following guides in order to facilitate it.

- Understanding multiplicity of stakeholders: Identifying the different parties (or publics) with different interests, values, economic impacts, and legal mandates in the community is the first step.

- Communicating with the public: Public involvement should be a dialogue (two-way communication) in which clear objective and information should be provided, and the public's concerns, opinions, and ideas should be considered.
- Conflicts management: Because conflicts from various interested parties are inevitable and unrecoverable, planners need to manage conflicts.
- Build credibility: Communicating accurate technical information is important in order to reduce mistrust.
- Addressing possible impacts: Common concerns about landfills including process issues, health risks, environmental issues, and local impacts should be managed.

Because this study focuses on preliminary analysis that utilizes an integrative methodology, managing participation is not desirable for this preliminary study, which has several reasons (e.g., need for rapid decision-making and lack of knowledge of complex issues on the part of the participants).

## 2.6. Economic valuation

Most economic valuation studies on the general topic of landfill sites have concentrated on examining the impacts of landfill sites on neighboring communities, surrounding environment, and future generations. Disregarding these externalities may incur planner's erroneous determination that landfill disposal is more cost effective than alternate methods e.g., source reduction, recycling, and waste to energy recovery. Estimating external or social costs could help planners to allocate solid waste resources more efficiently.

Externality is an activity where one economic agent affects utility or production functions of other economic agents (Freeman 2003). Disamenity is a general term of nuisances (e.g., dust, noise, and odors). While the externality is a theoretical term based on broader environmental notions, the disamenity includes local specific nuisances (Nelson et al. 1992a; Walton et al. 1996). Because impacts of landfills are mostly site specific, the term disamenity is often utilized as one of negative externalities.[14] For this study, disamenity is utilized as one type of externalities.

Externality or disamenity costs can be estimated by (i) revealed preference methods (e.g., a hedonic price method (HPM)), (ii) stated preference methods (e.g., a contingent valuation method (CVM)), and (iii) benefits transfer (BT) methods (e.g., mean transfer value (MTV) and meta-analysis (MA) approaches) (Brisson and Pearce 1995). Before discussing BT methods, the theoretical basis for HPM and the CVM and the related empirical findings will be discussed.

### 2.6.1. Hedonic model

The Rosen (1974)'s pioneering seminal article developed a formal theory of hedonic prices in the context of competitive markets. A price of a differentiated good (e.g., a house) reveals information about consumer's preference for certain characteristics (e.g., the number of rooms, lot size, and age). A marginal implicit price (MIP) of any characteristic is determined by interactions of marginal bids and marginal offers of buyers and sellers for its characteristic. The supply-side of market is assumed to be fixed in the short run (see Freeman 2003).

Following Rosen (1974), a consumer has a utility function U (x, $z_1$, $z_2$, …,$z_n$), where z=($z_1$, $z_2$, …,$z_n$) represents n characteristics for a differentiated good, and x is a composite good or numeraire representing money or income left over after purchasing the differentiated good. If a consumer buys only one unit of the differentiated good with the price P (z) and the numeraire with the unit price, the budget constraint is y = x + P (z), where y is income. The consumer chooses the levels of each characteristic ($z_1$, $z_2$, …, $z_n$) and a composite good (x) by

[14] Site specific effects are referred to as localized externalities which affect only small areas relative to the size of the urban market (Palmquist 2005)

maximizing a strictly concave utility function on the compact (closed and bounded) budget set:

$$\text{Maximize}_{x,\, z_i}\, U\,(x, z_1, z_2, \dots, z_n) \qquad \text{s.t } y = x + P\,(z) \qquad (2\text{-}1)$$

An optimal condition of the consumer's choice is

$$P_{z_i} = U_{z_i}/U_x, \;\; i = 1, \dots, n. \qquad (2\text{-}2)$$

where $P_{z_i} = \frac{\partial P}{\partial z_i}$ (the marginal implicit price of $z_i$), $U_{z_i} = \frac{\partial U}{\partial z_i}$ (the marginal utility of $z_i$), and $U_x = \frac{\partial U}{\partial x}$ (the marginal utility of x). The optimal condition means that the marginal implicit price of the ith characteristic ($P_{z_i}$) is equal to the marginal rate of substitution between its characteristic and a composite good ($U_{z_i}/U_x$).

This utility maximization problem can be explained by description of the optimal bid. A bid function is defined as $\theta = \theta\,(z_1, z_2, \dots, z_n; u, y)$ as a solution of the indirect function $U\,(x, z_1, z_2, \dots, z_n) = u$, where y (income) and u (any specified level of utility) are given, $z = (z_1, z_2, \dots, z_n)$ represents n characteristics for a differentiated good, and x is a composite good denoted as $x = y - \theta(z; u, y)$. The bid function represents the maximum amount that the individual is willing to pay to obtain the differentiated good with various characteristics.[15] Differentiating the indirect function with respect to $z_i$ leads to the following results:

$$U_x(-\theta_{z_i}) + U_{z_i} = 0 \rightarrow \theta_{z_i} = (U_{z_i}/U_x) \qquad (2\text{-}3)$$

The marginal willingness to pay for the ith characteristic($\theta_{z_i} = \frac{\partial \theta}{\partial z_i}$) is equal to the ratio of marginal utilities between any $z_i$ and x ($U_{z_i}/U_x$). Thus, in the market equilibrium, the partial derivative of the hedonic function P (z) with respect to any characteristic provides an estimate of the marginal implicit price (MIP) or marginal willingness to pay (MWTP) for changes in the attribute (i.e., $P_{z_i} = \theta_{z_i} = U_{z_i}/U_x$).

The hedonic theory by Rosen (1974) has been widely applied to measure impacts from landfills on property values by using distance from landfills (for a

[15] The differences in the individual's utility function and income can affect the bid function.

review of the theoretical basis for valuing environmental quality, see Freeman 1979). When housing characteristics (e.g., the number of rooms, lot size, and age) and distance from landfills are regressed on the sales price of a house, the estimated parameters or coefficients reveal the marginal implicit price (MIP) or marginal willingness to pay (MWTP) for each characteristic and distance to the landfill (Nelson 2004; Kiel and Williams 2007).

An example of the hedonic price method based on Nelson et al. (1992a, b, 1998) is as follows:

$$P(z) = \beta_0 + \beta_1 z_1 + \beta_2 z_2, \ldots \beta_n z_n + \beta_d d \qquad (2\text{-}4)$$

where the dependent variable P (z) is housing values, and $z=(z_1, z_2, \ldots, z_n)$ represents housing characteristics (e.g., the number of rooms, lot size, and age), the variable d is distance from the landfill, and the variable $\beta = (\beta_0, \beta_1, \ldots \beta_n, \beta_d)$ is a vector of the estimated coefficients, which reveals information about marginal values of characteristics. The coefficient of distance from the landfill ($\beta_d$) reveals the marginal implicit price (MIP) or marginal willingness to pay (MWTP) for the distance from the landfill.[16] The sign of $\beta_d$ is expected to be positive. Because a landfill is considered as nuisances (e.g., noise, odors, and debris), average housing values rise further away from the landfill (Nelson et al. 1992a, 1998). Brisson and Pearce (1995) found that average housing values near the boundary of landfill (1/4-1/2 miles or 0.4-0.8 km away) were 20-30% lower than housing values three to four miles or 6.4 km away from the landfill site.

### 2.6.2. Empirical HPM studies that measure external effects

As stated in the previous section, distance from the landfill site is often utilized to examine the effects of landfills on property values. HPM studies seem to emphasize several points: (i) positive distance effects (negative external

16 A non marginal change can be utilized by comparing housing values with a landfill to those in the absence of a landfill (Ready 2005).

effects)[17] and (ii) various distance effects according to the characteristics of the facilities (e.g., size of landfills), local specific features (e.g., income levels), and the age of landfills.

Distance effects from landfills on housing values (MWTP for distance)

Some studies (Gamble et al. 1992; Bouvier et al. 1990; Reichert et al. 1992) found statistically insignificant but unexpected negative distance effects. Planners or developers gave these studies as examples of evidence that landfills do not have negative impacts on housing values. However, the results may be attributed to small samples or site-specific characteristics (e.g., income level and population density). No study conclusively demonstrated that landfills do not have negative distance effects on property values.

Studies (Reichert et al. 1992; Nelson et al. 1992a, b and 1998; Brisson and Pearce 1995; Lim and Missios 2007) found a positive relationship between housing values and distance from landfills. A study by Nelson et al. (1992a) was a good example that showed statistically consistent positive distance effects. They developed an empirical model to estimate the effects of distance from the Anoka Regional Landfill in Ramsey, Minnesota on the value of 708 home sales observed during the 1980s. The area had a reasonably homogeneous housing market. They obtained data (e.g., housing values and characteristics such as the number of rooms, lot size, and age) from local realtors. Within two miles, average housing values increased by $4896 per mile (1.6 km) (MIP or MWTP for distance was 6.2%). Distance effects were negligible beyond two miles (3.2 km) away from the landfill. They concluded that a new landfill should be a few miles away from the residential area in order to reduce the strong opposition from neighboring residents.

---

[17] These negative external effects result in strong opposition by affected residents and environmentalists (Smith and Desvousges 1986b).

The extent of distance effects is debatable, which may vary according to the characteristics of the facilities (e.g., size of landfills), local specific features (e.g., income levels), and the life cycle and age of landfills.

Property values (income levels)

The distance effect (i.e., MWTP for distance) can differ according to different housing market conditions (e.g., different income levels or housing values). Reichert et al. (1992) and Nelson et al. (1998) proved that MWTP could vary on heterogeneous house price strata or income levels.[18] High-income households with greater substitution for landfill tend to be more sensitive to landfill location than low-income households do (Reichert et al. 1992; Nelson et al. 1998).

Reichert et al. (1992) examined the impact of five municipal landfills in Cleveland, Ohio on residential housing values within one mile (1.6 km) of each landfill (1985 to 1989). A pooled cross-section model (all five landfills) found statistically insignificant but unexpected negative signs of distance from landfills.[19] As a mean of defining a more homogeneous sample, they also estimated their model for the larger and more expensive areas nearby the Westlake landfill and the smaller and less expensive areas nearby the Jennings Road landfill. They found that the expensive housing areas had 5.5%-7.3% premiums depending on distance form the landfill while less expensive housing areas had 3%-4% premium depending on distance. They concluded that landfills had larger effects on housing values for the more expensive areas than for the less expensive areas.

[18] Through their literature review, Brinson and Pearce (1995) supported the fact that higher income levels are related to greater distance effects than lower income households are. For CVM research to validate this finding, see Smith and Desvousges (1986b) and Roberts et al. (1991).
[19] In the existence of heterogeneous multiple markets (e.g., different income levels or property values), estimation could lead statistically inconsistent results. Thus, market segmentation may result (Reichert et al. 1992).

Nelson et al. (1998) showed that different housing price levels or strata (income levels) can be a key variable of different distance effects. They examined the distance effects of the Flying Cloud MSW landfill (Eden Prairie, MN) on three different strata of housing values: homes over $150,000 (HS3), homes between $100,000 and $150,000 (HS2), and homes under $100,000 (HS1).[20] They found that price premium per mile (1.6 km) (MWTP) was 8.43 % in HS3 (higher valued strata), 4.32 % per mile (1.6 km) in HS2 (middle valued strata), and 2.64 % per mile (1.6 km) in HS1 (lower valued strata). They argued that because higher valued properties (or higher income households) were more sensitive to location than lower priced properties (or lower income households), landfills should be separated from the high populated urban areas.

The size of landfills

Lim and Missios (2007) and Ready (2005) examined whether the size of the landfill sites affected distance effects (MWTP for distance). They assumed that a larger landfill had higher MWTP than a smaller landfill because larger landfills processed more waste and required higher traffic levels from deliveries implying greater possibility of visual intrusion, odor, and litter. While Lim and Missios (2007) proved this assumption, Ready (2005) failed to validate this assumption.

Lim and Missios (2007) examined two landfill sites with different sizes (the larger Keele Landfill, which is 929 acres or 3.76 $km^2$, and the smaller Britannia Landfill, which is 206 acres or 0.83 $km^2$) in Toronto, Canada. They found that the larger Keele landfill had greater MWTP for distance (US$6260 premium per mile and on a percentage basis 3.7% per mile or 1.6 km) than the smaller Britannia landfill (US$3637 premium per mile or 1.6 km and on a percentage basis 2.2% per mile or 1.6 km). They found that larger landfills had greater distance effects than smaller landfills. They argued that although smaller landfills had lower social

[20] They classified sales of single family homes (from 1977 to 1988) into three different valued strata in 1980 constant dollars based on the national consumer price index.

costs than larger landfills, larger landfills had replaced smaller landfills because of the increasing difficulty in choosing a new landfill site and the advantage from economies of scale in reduced construction and operational costs.

Ready (2005) studied three landfills in southeastern Pennsylvania: Rolling Hills Landfill (120 acres or 0.49 $km^2$), Pioneer Crossing Landfill (92.5 acres or 0.37$km^2$), and Western Berks Landfill (65 acres or 0.26 $km^2$). He identified the spatial limit of each landfill's possible impact, and then estimated the distance effect from each landfill. For Pioneer Crossing Landfill, the MIP per mile or 1.6 km was 10.9% within 2 miles or 3.2 km. For Rolling Hills Landfill, the MIP per mile was 7.2% within 3 miles or 4.8 km. However, smaller Western Berks Landfill did not have statistically significant distance effects. They argued that comparison of Pioneer versus Rolling Hills is complicated. Rolling Hills had smaller MIP per mile (7.2%) than Pioneer Crossing (10.9%), but the impact of Rolling Hills extended over longer distances than Pioneer Crossing. Ready's study did not validate the assumption that larger landfill sites had larger MWTP for distance than smaller landfill sites.

Life cycle and age of landfills

Studies by Kohlhase (1991) and Kiel and McClain (1995a, b) examined the distance effects over time. They assumed that future changes in disamenity levels (i.e., the impact of landfills on housing values) are capitalized into current housing values. The change in the expectations about the impacts of landfills may incur different distance effects over time (e.g., the stage of the life cycle of landfill sites such as rumor to operation) (Kohlhase 1991; Kiel and McClain 1995a, b).[21]

---

[21]The change of impacts can be especially important to interpret information or policy actions such as the subsequent remediation or clean-up procedure. Information (e.g., public announcements for clean-up and newspaper publicity involving justification of landfill sites) provided to nearby residents can change the individual's risk perception of neighboring residents

Kiel and McClain (1995a, b) stressed how the impact of landfills on housing values can change. They examined distance effects of a municipal solid waste-to-energy incinerator in North Andover, Massachusetts at different phases from rumor to operation stages. They found no statistically significant results before construction (pre-rumor 1974-78; rumor 1979-80). Once incinerator construction was under way (1981-84), average housing prices increased $2283 per mile or 1.6 km (3.6%) further away from the incinerator. During the operation of the facility (1985-98), housing prices increased $8100 per mile or 1.6 km (7.2%) away from the incinerator. Housing prices increased slightly $6607 per mile (3.6%) farther away from the facility after 4 years of operation (1988-92). Even though an incinerator was used, they showed that distance effects could change over time.

Kohlhase (1991) analyzed whether the announcement of Superfund status affected consumer expectations in housing market. She examined the impact of 10 hazardous waste sites on housing values in Harris County, Houston at three periods (1976, 1980, and 1985). In 1976, most sites were operating without publicity about their safety. In 1980, the Superfund registration was passed, and the EPA's role in environmental clean-up was being widely publicized. In 1985, all 10 sites in Harris County were announced as being on the National Priority List (NPL) or Superfund status. She included distance to the nearest site and squared distance to determine non-linear distance effects. She did not find statistically significant results in 1976 and 1980. Only in 1985, after the announcement of Superfund status, she found that the marginal price of distance was about $2360 per mile or 1.6 km. She concluded that this may have been a response to the announcement

HPM studies seem to emphasize several points. Average housing prices increase further away from the landfill site (a positive distance effect). However,

---

over time. By releasing reports or news coverage explaining the justification for landfill sites, planners can decrease risk perception (see Gayer and Viscusi 2002).

the extent of distance effects varies according to local specific features (e.g., income levels), the characteristics of the facilities (e.g., size of landfills), and the landfill life period. While HPM utilizes housing markets in order to measure use values (e.g., the effects of landfills on property values), it does not measure nonuse value (Roberts et al. 1991; Pearce et al. 2006). Also, the application of HPM is difficult for multiple target sites without the existing landfill. The following contingent valuation method (CVM) can be an alternative when HPM is not applicable.

### 2.6.3. Theoretical basis for contingent valuation method

A contingent valuation method (CVM) is a survey-based technique to examine willingness to pay (WTP) or willingness to accept (WTA) for the change in environmental quality. Refer to Roberts et al. (1991) that measured WTP to avoid a proposed landfill and/or respondent's characteristics (e.g., age and income) and a level of environmental quality (e.g., distance from the landfill). The basic theory related to CVM is reviewed before discussing empirical findings on landfills (for relevant economic theory of CVM, see Mitchell and Carson 1989; Carson and Hanemann 2005).

A consumer utility maximization problem is to choose $x_j$ by maximizing the utility function subject to the compact budget set with the following equation (Hanemann 1991):

$\mathrm{Maximize}_{x_j}\ u(x, q)$ subject to $y = \sum_{j=1}^{n} p_j\, x_j$ (2-5)

where u (x, q) is individual's utility function (an increasing and quasi-concave utility function), x is a vector of private goods quantities for j = 1, 2, … , n, and q is a level of environmental quality (for simplicity, q is considered as a scalar fixed exogenously). The variable y is income, and p is a price vector of private goods. This yields a set of ordinary demand functions, $x_j = x_j$ (p, q, y) for j = 1 to n.

Inserting the ordinary demand functions into the utility function provides an indirect utility function, v (p, q, y) ≡ v [$x_j$ (p, q, y), q].[22]

The associated dual problem to the utility maximization in (2-5) minimizes total consumer expenditures needed to maintain a given level of utility (Hanemann 1991):

$$\text{Minimize}_{x_j} \sum_{j=1}^{n} p_j x_j \quad \text{subject to } u = u\,(x,\ q) \qquad \text{(2-6)}$$

where $p_j$ is a price vector of private good j, $x_j$ is a vector of quantities of private good j, and $\sum_{j=1}^{n} p_j x_j$ is the total consumer expenditures given utility. This solution yields a set of Hicksian demand functions, $h_j = h_j$(p, q, u) for j = 1, 2, …, n, and expenditure functions, e(p, q, u) ≡ $\sum_{j=1}^{n} p_j h_j$ (p, q, u).

Individual's welfare measures for changes in environmental quality are represented by compensating surplus (CS) and equivalent surplus (ES). If environmental quality changes from $q^0$ to $q^1$, CS and ES are defined as follows (Hanemann 1991; Carson and Hanemann 2005):

$$CS = e\,(p, q^0, u^0) - e\,(p, q^1, u^0) = y - e\,(p, q^1, u^0) \qquad \text{(2-7a)}$$

$$ES = e\,(p, q^0, u^1) - e\,(p, q^1, u^1) = e\,(p, q^0, u^1) - y \qquad \text{(2-7b)}$$

where p is a vector of prices of private goods, $q^0$ and $u^0$ represent initial levels of environmental quality and utility, and $q^1$ and $u^1$ represent subsequent levels of environmental quality and utility (refer to Appendix B of this study).

CS and ES measures are related to willingness to pay (WTP) and/or willingness to accept (WTA). Selecting a relevant welfare measure (CS or ES) depends on policy circumstances. As suggested by Mitchell and Carson (1998), CS is suitable for most policy contexts where values are measured relative to the current state. If a policy question is to locate a new landfill, CS represents the nearby resident's minimum WTA compensation for the loss (Kunreuther and Kleindorfer 1986; Mitchell and Carson 1986). However, if a policy restricts a

---

[22] The indirect utility function v (p, q, y) satisfies the standard properties with respect to p and y: (i) homogeneous of degree zero in p and y; (ii) increasing in y; and (ii) non-increasing in p; and (iv) quasi-convex in p (Carson and Hanemann 2005).

landfill in residential area, CS is the resident's WTP to avoid the landfill (for the use of WTP to measure external costs from a landfill, see Roberts et al. 1991).[23] Although WTP is an appropriate welfare measure for estimating external costs from a landfill, WTA may be theoretically relevant to measure the nearby resident's compensation or to design mitigation and incentive strategies that obtain resident's approval for the proposed landfill (for economic analysis to utilize WTA see Kunreuther and Kleindorfer 1986; Mitchell and Carson 1986; Swallow et al. 1992).

The magnitude of the divergence between willingness to pay (WTP) and willingness to accept (WTA) is debatable. For the quantity (or environmental quality) change, the difference between WTP and WTA is small if an income effect is small (Randall and Stoll 1980). However, WTA can be substantially higher than WTP in case of public goods which have low substitutes with other goods (Hanemann 1999). Pearce et al. (2006) argued that since a new landfill has relatively higher substitutes (e.g., source reduction, recycling, incinerators, and new landfills) than other public goods, the differences between WTP and WTA are relatively small. Empirical studies on landfill sites (Roberts et al. 1991; Smith & Desvousges 1986b) had a tendency to utilize WTP or consumer surplus instead of WTA since obtaining WTA may require substantial political or regulatory reforms, and the difference between WTP and WTA is theoretically small. The slight differences between WTP and consumer surplus or WTA (see Table 2.3) can be due to: (i) different models used and data used (ii) different time lengths and discount rates; and (iii) different risk perceptions (e.g., a hazardous waste site versus a municipal solid waste site).

### 2.6.4. Empirical CVM Studies

[23] The welfare measures are closely related to property rights. If nearby residents have property rights to have an undisturbed neighborhood, CS represents nearby resident's WTA compensation. If the landfill is public owned or regulated, CS (resident's WTP to avoid it) is a correct measure (Freeman 2003).

Few studies (Roberts et al. 1991; Smith & Desvousges 1986b; Opaluch et al. 1993) conducted CVM on landfill sites (see Table 2.3). Roberts et al. (1991) was the most relevant CVM study to examine external costs from a municipal solid waste (MSW) landfill. They surveyed a sample of 150 households from residents (Knox County, Tennessee) within a 4-mile or 6.4 km radius of a proposed landfill site. Under the hypothetical market where residents could pay annual taxes or garbage collection fees into a fund,[24] respondents were asked to indicate WTP to restrict the landfill nearby his or her residence. Responders also answered other questions about their income, age, sex, and perception of health risks. With this information, they examined the relationship between WTP estimates and these characteristics and distance from the landfill. The annual mean WTP per household was $227 in 1988 dollars. They found that (i) WTP decreased further away from the landfill site; (ii) WTP had a positive relationship with household income; and (iii) WTP increased with higher risk of water contamination and concerns on human health. These findings are consistent with those of the HPM studies.[25]

Smith and Desvousges (1986b) developed a household's demand model for distance from a hazardous waste site. They surveyed 609 households in suburban Boston. Responders were asked to choose between two homes with identical characteristics except for their distance from the waste site. One of four marginal values per mile ($250, $600, $1000, and $1300) was assigned to each responder. Respondents were also asked to answer their characteristics (e.g., income, housing values, education, and risk perceptions). They investigated the factors that affected the demand for distance. Their findings are consistent with Roberts et al. (1991): (i) a positive relationship between the demand for distance and income and housing values; (ii) a positive relationship with the demand for distance and risk perceptions that represent the individual's rating of degree of harm with the 1 (= not harmful) to 10 (= very harmful) range; and (iii) a negative

[24] The fund can be utilized to restrict the landfill in residential area or enhance alternate methods e.g., source reduction, recycling, and incinerators.
[25] These findings can serve as a validity check for other HPM studies.

relationship between the demand for distance and marginal prices of distance. They also found that the annual consumer surplus for each mile or 1.6 km of distance from the hazardous site was $330 to $495. They argued that planners would need to provide suitable incentives for having a landfill site built in their neighborhood because a landfill could be a great burden for nearby residents.

Table 2.3. Summary of Contingent Valuation Studies in the United States

| Study | Method | Valuation* |
|---|---|---|
| Landfill in Knox County, Tennessee, USA by Roberts et al. (1991) | CV asking 150 respondents their WTP for a landfill to be located elsewhere | $260 average WTP per household per year to move landfill 4 miles or 6.4 km from current location |
| A hazardous waste disposal site in Boston, USA by Smith and Desvousges (1986b) | CV survey asking 609 households to chooses between two homes that have identical physical characteristics except for their distance to the landfill | $420-$630 average consumer surplus per year per mile or 1.6 km from a landfill |
| Landfill site in Rhode Island, USA by Opaluch et al. (1993) | CV based on the comparisons between two possible sites in order to indicate the factors affecting public decision making | Groundwater and surface water quality were found to be the most important factors |

Source: Brisson and Pearce (1995)
* Valuation (WTP) is in 1992 dollars.

Unlike Roberts et al. (1991) and Smith and Desvousges (1986b) that asked monetary values (e.g., WTP and consumer surplus), Opaluch et al. (1993) employed a contingent choice survey based on paired comparisons in order to evaluate public preferences of Rhode Islanders for a new landfill site. Responders were asked to choose between the two hypothetical landfill sites with different characteristics such as groundwater quality, presence of wildlife habitat, distance to access roads, and the number of homes and presence of schools within a 4 squared mile or 10.4 $k^m$ area of the landfill. A score with the 1 (lower preferences) to 5 (higher preferences) range for one preferred site was assigned to each responder. By investigating the relationship between scores and characteristics, they found that groundwater quality was the most important factor to reduce the probability that a new landfill site would be chosen. Estimated coefficients (or numerical weights) for site characteristics (e.g., marsh area, farm land, groundwater quality, wildlife habitat, and ponds), location characteristics (e.g., park land, farm land, and schools), and annual developing cost per household were used for scoring and ranking the proposed sites in Rhode Island.

While most studies have applied HPM on landfills, few studies (Roberts et al. 1991; Smith & Desvousges 1986b; Opaluch et al. 1993) conducted CVM on landfill sites (see Table 2.3). Poorly constructed and implemented surveys can lead to bias (e.g., overstatement of WTA and understatement of WTP), and the results can misguide assessments or the decision-making process. Researchers encountered difficult in obtaining reasonable results from surveys since respondents often did not understand the situation and the survey process (Diamond and Hausman 1994). Despite these difficulties, carefully designed CVM can reduce these biases (for methods to reduce biases, see Mitchell and Carson 1989). CVM has the great flexibility in obtaining information or measuring environmental quality changes that other valuation methods (e.g., HPM) cannot obtain (e.g., WTA and non-use value that HPM cannot measure) and a relationship between WTP or WTA and other characteristics (e.g., income, gender, race, education, and risk perceptions). CVM can also contribute to check reliability on HPM (Brisson and Pearce 1996).

Conducting these primary HPM and CVM is not desirable for this project designed to develop and test a method for preliminary analysis, which has inaccessible data, short time frame, and little money.  For example, conducting HPM is difficult to find reliable findings from proposed sites without the existing landfill, and conducting CVM can measure external costs for each target site (e.g., 7 target sites) but requires substantial time and money. If researchers and planners have enough time and money, they can implement primary CVM for each target site. If not, benefits transfer (BT) discussed in the following section can be much better than no framework.

## 2.7. Benefits transfer

### 2.7.1. Basic concepts of benefits transfer

Benefits transfer (BT) is a research method applying values or functions from an original context to a different or transfer context (Desvousges et al. 1998).[26] The original context is referred to as a study site where original research was conducted, which may be located on Oahu or elsewhere (see Downing and Ozuna 1996 for the case of other sites on Oahu without data). The transfer context is referred to as a policy or target site (e.g., target sites on Oahu) where a new policy is implemented, or value is measured (Rosenberger and Loomis 2003). When researchers have encountered difficulty in conducting primary research (e.g., time and monetary constraints), benefits transfer (BT) can be a useful tool for obtaining information or approximating the impacts of landfills by utilizing information or empirical findings from other primary studies.

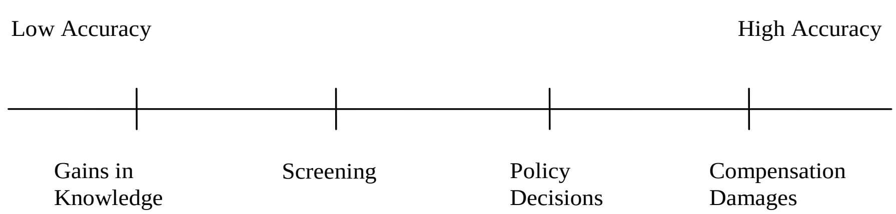
Two approaches to benefits transfer (BT) are generally utilized: (i) a mean transfer value (MTV) approach to utilize mean values for WTP or marginal WTP from primary studies (termed study sites) to target or policy sites and (ii) a transfer function utilizing a function (e.g., a benefit or WTP function or a meta-analysis function) from study sites to target sites (Rosenberger and Loomis 2003). In order to reflect differences between target sites and study sites, mean transfer values for WTP are adjusted with income or population (Eshet et al. 2005; Navrud and Ready 2007). As a more sophisticated method, a transfer function is generally considered to perform better than a mean transfer value because the transfer function can fit some of the characteristics of the target sites (Rosenberger and Loomis 2003).[27]

---

[26] Although a more precise term can be value transfer incorporating both benefits and damages, a term benefits transfer is dominate in the literature (Navrud and Ready 2007).

[27] Rosenberger and Phipps (2006) showed that inclusion of market characteristics (e.g., income levels) improves the validity of meta-analysis for benefits transfer (MA-BT).

Transfer error (TE) measured by the absolute percentage difference between transfer values ($\mathrm{WTP_T}$) and primary values ($\mathrm{WTP_p}$) for target sites is often utilized to assess reliability of benefits transfer (BT) i.e., smaller TE performs better (Rosenberger and Loomis 2003).[28] The reliability for BT is affected by following sources: (i) measurement error that results from the methods used in the primary research (e.g., data qualities, functional forms, and analyst's judgment regarding research design and implementation); (ii) generalization error that arise from differences between the study and policy sites; and (iii) publication selection error (for details of publication selection errors, see Rosenberger and Stanley 2006). Choosing a similar study site as a transfer context reduces generalization error, and inclusion of local demographic variables (e.g., income and population density) significantly improves the reliability of benefits transfer (Rosenberger and Phipps 2007).[29] Considering unpublished papers may reduce publication selection error. However, some degree of inaccuracy from BT still exists.




Source: Brookshire (1992)

Figure 2.5. Required accuracy of benefits transfer

[28] While validity requires statistical tests finding whether transfer values or functions from study sites are statistically same as those estimated at policy sites, reliability requires the relatively smaller absolute percentage (Navrud and Ready 2007).

[29] Rosenberger and Phipps (2001, 2007) showed that using meta-analysis transfer function can reduce these potential biases. Inclusion of market characteristics (e.g., income levels) especially improves the validity of meta-analysis for benefits transfer (MA-BT).

Despite its probable inaccuracy (or transfer error), planners generally accept the use of benefits transfer (BT) as an appropriate method. Figure 2.5 shows the stylized cases that require the level of BT accuracy. A relatively low level of accuracy is acceptable if the objective of BT gains more knowledge about WTP to avoid negative externalities or disamenities from target sites or provide an initial assessment of the value (e.g., ranking or screening sites). If the objective of BT involves actual policy decisions or damage compensation litigations, a higher level of accuracy is required (Brookshire 1992). Together with the acceptable level of accuracy, the use of BT should be considered.

The following sections discuss approaches widely used for BT: a mean value transfer, a WTP transfer function, and a meta-analysis transfer function.

2.7.2. A mean transfer value for WTP

A mean value for WTP from an original context (termed study sites, s) can transfer directly to a transfer context (termed policy or target sites, p) when the original context has similar characteristics as the transfer context.[30] However, this method does not reflect likely divergence between the study and target sites such as the site-specific characteristics (e.g., income levels and population density) and temporal changes. Thus, some adjustments can improve the quality of the transfer. The following formula is widely used for for income adjustments (Pearce et al. 2006):

$$\mathrm{WTP_p} = \mathrm{WTP_s} * (\mathrm{Y_p}/\mathrm{Y_s})^{\mathrm{e}} \qquad (2\text{-}8)$$

where s is the original context (study sites), and p represents target or policy sites. $\mathrm{WTP_p}$ and $\mathrm{WTP_s}$ are mean willing to pay per household at p and s, and $\mathrm{Y_p}$ and $\mathrm{Y_s}$ are per capita income (or median household income) levels in real terms at p and s. The variable e is the income elasticity of WTP for landfills in question. Little empirical evidence is known on how WTP varies with income

[30] This direct mean transfer approach assumes that well-beings of an average household between a study and a target site are same.

(Navrud 2004; Pearce et al. 2006). For BT, a common assumption is a unitary income elasticity (or e =1), which implies that the share of WTP estimates vary proportionally with income (Navrud 2004). Because environmental goods are often considered as luxury goods, e is greater than 1.[31] However, the literature review by Pearce (1980) showed that e was less than one, which implies that the share of WTP falls as income rises. A recent review by Kristrom and Riera (1996) supported that e is less than 1. For example, Pommerehne (1988) and Krupnick et al. (1996) found that e is approximately 0.3. Sensitivity analysis can be performed to check how transferred values could vary with e.

A range of WTP values may provide bounds on the probable WTP at target sites (Rosenberger and Loomis 2003). For example, when multiple study sites exist, a study site with the lowest WTP becomes the lower bound of the transfer, and a study site with the highest WTP becomes the upper bound of the transfer. Alternatively, a confidence level represents a simple statistical range in which the true value (WTP) would fall with some percentage if the mean and standard error of WTP is available (Desvousges et al. 1998). These bounds or confidence levels may provide additional information regarding the precision of study site estimates.

Aggregate WTP can be calculated if WTP is multiplied by the affected population. These social costs or aggregate values should be calculated as present values (e.g., a base year 2008). All values are adjusted to the base year, and a real discount rate adjusts for future inflation (Navrud and Ready 2007).

### 2.7.3. Transfer function

A transfer function method involves applying an estimated demand or WTP function from primary HPM or CVM primary research to measure values for target sites (Navrud and Ready 2007). Value or WTP per household can be calculated at target sites by substituting data or information of target sites into a

[31] Those with higher income express higher WTP for environmental quality than those with less income.

value function. Total values (present values) can be calculated if the individual WTP is multiplied by the number of households. For example, the WTP function reflecting the relationship between WTP and household characteristics and distance from the proposed landfill is provided as follows (Roberts et al. 1991):

$$\mathrm{WTP_s} = \mathrm{f_s}\ (\beta_\mathrm{s}, \mathrm{X_s}) = \beta_{\mathrm{s}_0} + \beta_{\mathrm{s}_1}\mathrm{X}_{\mathrm{s}_1} + \beta_{\mathrm{s}_2}\mathrm{X}_{\mathrm{s}_2} + \ldots\beta_{\mathrm{s}_\mathrm{n}}\mathrm{X}_{\mathrm{s}_\mathrm{n}} \qquad (2\text{-}9)$$

where $\mathrm{WTP_s}$ is household's WTP at s (a study site or original context), $\mathrm{X_s}= (\mathrm{X}_{\mathrm{s}_1}, \mathrm{X}_{\mathrm{s}_2}\ldots\mathrm{X}_{\mathrm{s}_\mathrm{n}})$ is a vector of the characteristics of the affected households at the study (e.g., income, education, home ownership, years of residence, and distance from the proposed landfill site), and the variable $\beta_\mathrm{s}=(\beta_{\mathrm{s}_0}, \beta_{\mathrm{s}_1},\beta_{\mathrm{s}_2}\ldots\beta_{\mathrm{s}_\mathrm{n}})$ is a vector of the constant and estimated coefficients from the primary study.

This approximate transfer function ($\mathrm{f_s}$) can be used for measuring $\mathrm{WTP_p}$ at the target sites:

$$\mathrm{WTP_p} = \mathrm{f_s}\ (\beta_\mathrm{s}, \mathrm{X_p}) = \beta_{\mathrm{s}_0} + \beta_{\mathrm{s}_1}\mathrm{X}_{\mathrm{p}_1} + \beta_{\mathrm{s}_2}\mathrm{X}_{\mathrm{p}_2} + \ldots\beta_{\mathrm{s}_\mathrm{n}}\mathrm{X}_{\mathrm{p}_\mathrm{n}} \qquad (2\text{-}10)$$

where $\mathrm{WTP_p}$ is the estimated WTP at target sites (p = 1, 2, … M), and $\mathrm{X_p}= (\mathrm{X}_{\mathrm{p}_1}, \mathrm{X}_{\mathrm{p}_2}\ldots\mathrm{X}_{\mathrm{p}_\mathrm{n}})$ is a vector of the characteristics of the affected household at the target sites. Average WTP per household at each target site is measured when the coefficients ($\beta_\mathrm{s}$) of the transfer function estimated from the above primary research are multiplied by variables ($\mathrm{X_p}$) at each target site. Social costs (aggregate WTP) at each target site are then estimated if the affected population at the target site is identified.

This transfer function method often fails to find primary CVM research. Few CVM studies were conducted on landfill sites (Roberts et al. 1991; Smith et al. 1986; Opaluch et al. 1993). Even the most relevant function (Roberts et al. 1991) is inapplicable to the target sites because of inaccessible data at target sites (for illustrative examples of transfer function approach, see Loomis et al. 1995; Downing and Ozuna 1996; Kirchhoff et al. 1997). Thus, this study will utilize meta-analysis discussed in the following section.

### 2.7.4. Meta-analysis

#### 2.7.4.1. The basic idea of meta-analysis for benefits transfer (MA-BT)

Meta-analysis (MA) is a statistical method to examine empirical findings from multiple primary studies e.g., marginal willingness to pay (MWTP) for distance from hedonic price method (HPM) studies. As a form of MA, meta-regression analysis examines systematic relationships between reported valuation estimates (MWTP) and explanatory variables that adjust for different MWTP estimates across studies (Stanley and Jarrell 1989).[32] Although meta-regression analysis is a more precise term, the term meta-analysis is dominant in the literature. Thus, this study will utilize the term meta-analysis incorporating meta-regression analysis. The estimated meta-analysis function can be used for measuring social costs in the transfer context (e.g., new or unstudied sites such as target sites on Oahu, Hawaii).

There exists no single clear and universal methodology regarding meta-analysis benefits transfer (MA-BT). However, Bergstrom and Taylor (2006) and Smith et al. (2002) suggested a standard for MA-BT: (i) commodity consistency (e.g., municipal solid waste (MSW) landfills); (ii) welfare change consistency (e.g., MWTP for distance based on HPM research); and (iii) theory consistency (e.g., positive distance effects). The MA-BT model assumes the existence of an underlying meta-analysis valuation function that relates reported MWTP estimates from primary HPM studies to core economic variables (e.g., income levels) and study design variables. Based on economic theory, core economic variables are key factors reflecting differences between target sites. Study design variables can improve the predictive power of the MA-BT model by adjusting for different MWTP estimates that occurs from dissimilar research designs (e.g.,

[32] Meta-regression analysis traditionally has been utilized to statistically examine the variation of findings between the studies.

model types, statistical estimation methods, and sample sizes) (for theory and practice, see Appendix C).

2.7.4.2. Meta- Analysis for Benefits Transfer (MA-BT)

As examined in the previous sections, most valuation studies on the general topic of landfill sites have focused on HPM that examines the effects of landfills on housing values by utilizing distance from landfills. Thus, HPM studies are suitable for meta-analysis (for meta-analysis with HPM studies, see Smith and Huang 1995; Ready 2005; Walton et al. 2006). Few studies (Robert et al. 1991; Smith et al. 1986; Opaluch et al. 1993) utilized CVM because poorly constructed andimplemented surveys lead to bias (e.g., overstating WTA or understating WTP), and the results can misguide either damage assessments of government decision-making (Diamond and Hausman 1994). Despite the difficulties, careful use of CVM can obtain important information (e.g., WTA and non-use value that HPM cannot measure) and a relationship between WTP or WTA and other characteristics (e.g., income, gender, and race). CVM is often utilized to check reliability on HPM (Brisson and Pearce 1996). A mean transfer value for WTP based on primary CVM research can be conducted for a comparison purpose.

The following meta-analysis (MA) function analyzes relationships between $\mathrm{MWTP_i}$ and explanatory variables (Bergh et al. 1997):

$$\mathrm{MWTP_i} = \beta + \sum_{k=1}^{K} \propto_k \mathrm{Z_{ik}} + \mathrm{v_i}, \qquad \mathrm{v_i} \sim \mathrm{N}\,(0, \sigma_{\mathrm{v_i}}) \quad (\mathrm{i} = 1, 2, \ldots \mathrm{L}) \qquad (2\text{-}11)$$

where $\mathrm{MWTP_i}$ is the ith reported estimate (marginal WTP for distance) among the L estimates from HPM studies, and $\mathrm{Z_{ik}}$'s are explanatory variables in an equation of the ith estimate such as core economic variables (e.g., income levels) and study design variables (e.g., the number of observations, functional forms, and model specifications in the primary hedonic regression models). The variable $\beta$ is a vector of true MWTP, and $\propto_k$'s are the meta-analysis coefficients which reflect the different effects of particular study characteristics, and $v_i$ is the disturbance term (for a general idea of MRA, see Stanley and Jarrell 1989).

The estimated meta-analysis function can be applied to approximate average MWTP per household at target sites. By holding the effect of study design variables constant (using mean values in the data), target sites are assumed to have similar study design variables (Smith et al. 2002). Core economic variables for target sites are inserted to reflect the differences between the target sites. Aggregate MWTP can be calculated when average MWTP per household is multiplied by the number of household for target sites (Freeman 2003).

The MA may incur primal econometric problems: (i) the variance of $v_i$ is likely to be heteroskedastic because of primary studies with different data sets, different sample sizes, and different independent variables (Bergh et al. 1997), and (ii) the panel data structure that may occur due to multiple MWTP estimates from single studies can cause an unequal weighing and a bias in the coefficients (for panel stratification in meta-analysis, see Rosenberger and Loomis 2000b).[33] These econometric concerns should be managed by econometric methods (e.g., generalized least squares (GLS), robust standard errors, and data transformation).

2.7.4.3. Existing meta-analysis (MA) studies

Three meta-analysis studies on landfills were completed by Brisson and Pearce (1995), Ready (2005), and Walton et al. (2006). Each study had a slightly different intention. These studies focused on HPM using distance from a landfill site. Brisson and Pearce (1995) did not employ a meta-analysis function for benefits transfer (BT) but provided a basis for benefits transfer (BT) with distance

[33] Rather than transferring one WTP or marginal WTP function from one selected study, MA combines results from multiple studies to provide one common function. In a MA the result from each study is treated as a single observation. When a single study provides multiple results, data structure is similar to panel data (Navrud 2004). This panel data structure can result in a bias in the regression coefficients. Rosenberger and Loomis (2000b) modeled data as panel data and tested for this panel data structure. However, they did not found statistically significant effects of panel data.

from landfill sites as a core variable to measure social costs at target sites. They provided a provisional meta-analysis (MA) function linking a percentage decrease in housing value and distance from the landfill. Their study may suffer from specification bias because of small samples, lack of methodological variables, inclusion of studies using different valuation methods, and inclusion of hazardous waste sites.

Ready (2005) included some methodological variables (e.g., the amount of transported waste per day, sample size of each study, and distance from landfill sites) that determined MWTP. However, Ready's study did not obtain statistically significant results for methodological variables due to the use of a small sample size. He did not explain heteroskedasticity and panel data effects. He also did not utilize meta-analysis (MA) for benefits transfer (BT).

Walton et al. (2006) conducted meta-analysis (MA) on North American hedonic studies. They showed that core economic variables (e.g., income levels) and methodological variables (e.g., sample size, year of each primary study, and functional from) improved the reliability of the meta-analysis for benefits transfer. Through the validity test that compared the results of meta-analysis (MA) with those of the previous primary CVM study (Garrod and Willis 1998), they found that meta-analysis on hedonic price method (HPM) studies slightly underestimates disamenity costs as compared with the original contingent valuation (CVM) study. However, they did not utilize MA for landfill site selection. Also, they did not check sensitivity and validity of results.

Current research extends beyond these studies by following the standard for the MA-BT approach suggested by Bergstrom and Taylor (2006) and Smith et al. (2002): (i) commodity consistency (e.g., MSW landfills); (ii) welfare change consistency (e.g., HPM studies); and (iii) theory consistency (e.g., positive distance effects) (for good examples of the consistent economic concepts and BT consistency see Smith and Huang 1995; Walton et al. 2006). Thus, by focusing on HPM studies for landfill sites, the current study will develop meta-analysis functions that relate MWTP with core economic variables (e.g., income levels and population densities) and study design variables (e.g., the number of observations, standard errors of each selected study, and functional form). The meta-analysis functions can be used in order to measure average MWTP per household at target sites, and Aggregate MWTP can be measured by multiplying average MWTP per household with the number of households at target sites. Meta-analysis models are evaluated in terms of sensitivity, validity, and reliability criteria. A mean transfer value method will be also conducted for a comparison purpose. Meta-analysis models will be evaluated in terms of sensitivity, reliability, and validity criteria.

Chapter 3 provides the proposed integrative methodology that links a GIS-based analysis with an economic framework: (i) a GIS-based screening of possible landfill sites that satisfy constraints and (ii) an economic framework (e.g., benefits transfer methods e.g., meta-analysis and mean transfer value approaches) that ranks the remaining selected sites according to social cost minimization. This chapter discusses: (i) the procedure and data used for the GIS-based preliminary screening; (ii) the process and data utilized for benefits transfer (BT) methods; and (iii) the procedure to check validity and reliability of benefits transfer (BT) methods.

# CHAPTER 3. PROPOSED METHODOLOGY

## 3.1. General methodology: linking a GIS-based analysis with an economic Approach

This study employs an integrative methodology that links a GIS analysis with an economic framework (for a general idea of an integrative approach, see Swallow et al. 1992). While the economic framework yields a theoretical basis, the GIS-based method acts as a practical tool for preliminary screening. The basic site selection framework minimizes social costs given constraints (restrictive or exclusionary criteria based on rules and restrictions mandated by public agencies), which has the following general mathematical formulation (Swallow et al. 1992):

Minimize social cost (J) subject to restrictive or exclusionary criteria (3-1)

where restrictive criteria include U.S. EPA's six landfill selection criteria (e.g., airports, floodplains, wetlands, fault areas, seismic impact zones, and unstable areas) and restrictions on groundwater, land use, and capacity of landfill sites (for at least a ten year) (for restrictive criteria, see Chapter 2 pp 37-39 of current research).

The integrative approach incorporates a two-part process: (i) a GIS-based screening of possible landfill sites that satisfy constraints (e.g., airports, wetlands, floodplains, land use, groundwater, and landfill capacity) and (ii) benefits transfer methods (e.g., meta-analysis and mean transfer value approaches) that rank the remaining selected sites according to social cost minimization (see Figure 3.1).

The first stage utilizes a GIS-based method in order to select sites satisfying the constraints (Daneshvar et al. 2005). The use of GIS assists in analyzing geographic characteristics (e.g., soil properties) and collecting secondary data (e.g., income levels, population densities, the number of households, and housing values) for target sites. By processing a large amount of data in a short time, the GIS analysis can reduce the effort required for information collection and processing. Although the literature on landfill site selection utilized a GIS (Siddiqui et al. 1996; Baban and Flannagan 1998; Kontos et al. 2003, 2005; Daneshvar et al. 2005, Sener et al. 2006; Hasan et al. 2009), there is a tendency to disregard economic analysis. The economic analysis can provide a theoretical basis (e.g., social cost minimization given exclusionary criteria) and measure the impacts of proposed landfill sites, which has advantages (e.g., replicability, transparency, and ease of comparison between target sites or other projects).

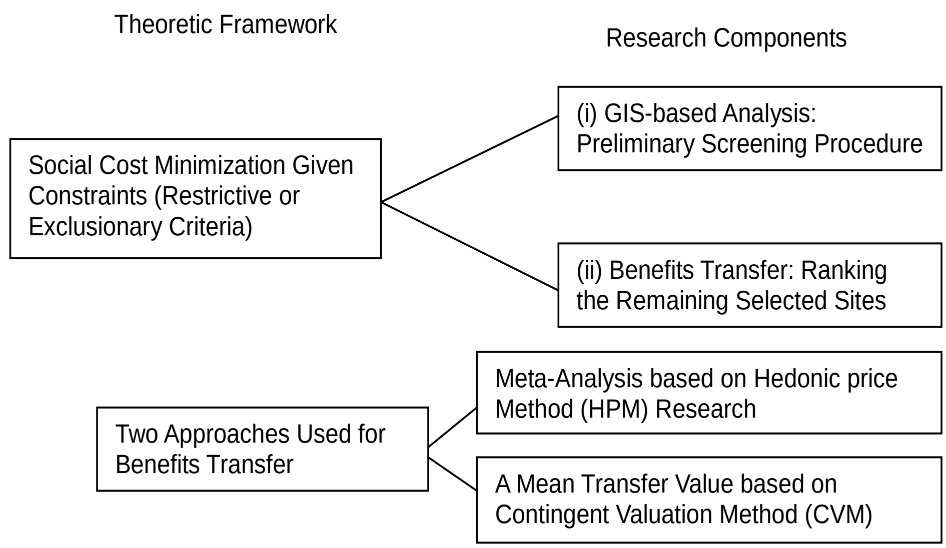



Figure 3.1. Theoretic framework and research components

The second stage utilizes benefits transfer (BT) methods (e.g., meta-analysis (MA) and mean transfer value (MTV) approaches) in order to measure social costs for the remaining selected sites and rank these sites according to social cost minimization. A landfill site has adverse impacts on neighboring communities, the surrounding environment, and future generations (e.g., environmental pollution and decreases in housing values). If planners fail to consider external or social costs, they will likely grossly underestimate the costs of the landfill and possibly locate the landfill in a higher overall cost location.

Careful examination of social costs can help planners to locate a new landfill or allocate solid waste resources more efficiently.

This study utilizes a meta-analysis for benefits transfer (MA-BT) approach suggested by Bergstrom and Taylor (2006) and Smith et al. (2002). As examined in chapter 2, while most valuation studies on landfill sites have focused on HPM utilizing distance from landfills, few studies (Robert et al. 1991; Smith et al. 1986; Opaluch et al. 1993) utilized CVM.[34] Thus, HPM studies are suitable for meta-analysis (for meta-analysis with HPM studies, see Smith and Huang 1995; Ready 2005; Walton et al. 2006). Conducting a primary hedonic price method (HPM) and a contingent valuation method (CVM) is not desirable for this project designed to develop and test a method for preliminary analysis, which has several reasons (e.g., inaccessible data, short time frame, and little money). While implementing HPM is difficult to find reliable findings from proposed sites without the existing landfill, conducting CVM can measure external costs for each target site (e.g., 7 target sites) but requires substantial time and money. If researchers and planners have enough time and money, they can conduct primary CVM for each target site. If not, this preliminary analysis is much better than no framework.

This study follows the standard for valid and reliable benefits transfer suggested by Bergstrom and Taylor (2006): (i) welfare measure consistency (e.g., hedonic price methods), (ii) commodity consistency (e.g., distance from municipal solid waste landfills), and (iii) theory consistency (e.g., positive distant effects). Meta-analysis (MA) functions are developed, which have relationships between marginal willingness to pay (MWTP) for distance (i.e., the impacts of landfills on housing values based on primary HPM studies) and explanatory variables such as core economic variables (e.g., income levels and population

[34] Few CVM studies were conducted on landfills due to some reasons (e.g., poorly constructed and implemented surveys and bias that result from overstating WTA or understating WTP). However, careful use of CVM can obtain important information (e.g., WTA and non-use value that HPM cannot measure) and a relationship between WTP or WTA and other characteristics (e.g., income, gender, and race). CVM is often used to check reliability on HPM (Brisson and Pearce 1996).

densities) and study design variables (e.g., the number of observations, standard errors of each selected study, and functional form). The MA functions are utilized to measure social costs for the selected sites and to rank these sites. For a comparison purpose, a mean transfer value for WTP approach based on CVM research is also conducted. While HPM measures use value (e.g., impacts of landfills on housing values), CVM measures use value and/or non-use value (e.g., WTP to avoid a landfill). Thus, either CVM or HPM can be often employed to check reliability on the other by ranking the selected sites according to the minimization of social costs objective function (Brisson and Pearce 1995).

Although planners and researchers recognize the significance of economic analysis, they lack confidence in applying economic analysis to the site selection process. In order to manage economic efficiency and equity, some studies (Kunreuther and Kleindorfer 1986; Mitchell and Carson 1986) suggested decentralized market approaches including (i) auction mechanisms (Kunreuther and Kleindorfer 1986) and (ii) formal referenda where communities enforce a property right to approve a new landfill (Mitchell and Carson 1986). However, conducting these decentralized market approaches may require substantial political or regulatory reforms and are feasible only when the list of potential sites is short (Swallow et al. 1992). Few studies (Swallow et al. 1992; Opaluch et al. 1993) have linked a GIS analysis with an economic framework for landfill site selection. These studies evaluated public preferences of Rhode Islanders for landfill site selection and employed a GIS for data collection. However, the direct survey method requires substantial time and a large monetary budget.

Few studies (Brisson and Pearce 1995; Ready 2005; Walton et al. 2006) utilized MA to examine factors that affect different impacts of landfills on housing values. They did not employ a MA approach for landfill site selection. To the author's knowledge, this is the first study on landfill site selection that links a GIS analysis with a meta-analysis for benefits transfer (MA-BT) approach.

This integrative approach is implemented for the island of Oahu regarding landfill site selection, which differs from the City & County of Honolulu approach (see Table 3.1). The City's approach incorporates a preliminary screening phase (e.g., preliminary screening and ranking based on the length of landfill life) and a further assessment phase to identify the Committee's preference for specific candidate sites (e.g., Committee's voting and ranking based on its criteria and weights) (for details, see City & County of Honolulu 2003).[35] Rather, this study provides an integrative approach for landfill site selection that links a GIS analysis with a meta-analysis for benefits transfer approach. Meta-analysis models are evaluated in terms of sensitivity, validity, and reliability criteria.

Table 3.1. Site Selection Approaches of Current Research and City & County of Honolulu

| Site selection approach | City & County of Honolulu (Oahu County) | Current research |
|---|---|---|
| Preliminary screen phase | Preliminary screen by Environmental Services (ENV) and consultants<br>Ranking based on a landfill life (at least a 10 year) | Preliminary screen with a GIS analysis<br>Ranking based on benefits transfer according to the social cost minimization |
| Further assessment phase | Committee's voting and ranking based on its criteria and weights | Sensitivity, reliability, and validity checks on MA functions |

Environmental justice or equity and participation are important in the process of landfill site selection. Environmental equity is becoming important because all relevant policy appraisal processes (e.g., environmental impact statements) require addressing this issue. Lack of participation can incur strong opposition to a proposed landfill and delay the process or increase costs. When various interest groups involved in the process of landfill site selection participate, this project is more likely to be successful (Pearce et al. 2006).[36] Despite their importance, this study will not address these issues. Environmental equity is partially managed by BT methods (e.g., meta-analysis and mean transfer value

[35] The Mayor's Advisory Committee on landfill site selection in 2003 was composed of 15 members including community representatives from various geographic areas of the City, the business community, and the Department of Health (DOH).

[36] For environmental equity, planners need to incorporate minority populations (e.g., race, color, and gender) and lower-income communities in the site selection process and provide clear information related to the process of site selection (see O'Hare et al. 1983; Kreith and George 2002).

approaches) that consider various factors (e.g., income levels, population densities, the number of households, and housing values). In terms of social costs (society's burdens), the least impacted site fulfilling exclusionary criteria is selected. This study assumes that compensation for impacted households and aid to reduce transaction costs for low-income households (e.g., providing legal service and information) can improve inequity. Managing participation is not desirable for this preliminary study, which has several reasons (e.g., need for rapid decision-making and lack of knowledge of complex issues on the part of the participants).

## 3.2. GIS analysis for preliminary screening procedure

A GIS-based analysis follows a preliminary screening procedure suggested by Daneshvar et al. (2003): (i) identify objectives and exclusionary criteria, (ii) prepare data analysis, (iii) create constraint maps, and (iv) determine potential sites that satisfy exclusionary criteria (for the process of site selection and restrictive criteria, see Chapter 2 pp. 34-39 of this research). The objective of the GIS analysis is to select sites fulfilling exclusionary criteria before applying the economic analysis. In order to attain this objective, exclusionary criteria are first developed. GIS data compatible with these exclusionary criteria are then prepared, and constraint maps are created to identify whether potential sites satisfy exclusionary criteria.

This study utilizes a constraint mapping method suggested by Baban and Flannagan (1998), Kontos et al. (2003), and Daneshvar et al. (2005). The GIS creates various thematic maps incorporating constraints (restrictive or exclusionary criteria) for landfill selection, which can identify whether a proposed landfill site fulfills restrictive or exclusionary criteria (e.g., airports, wetlands, and floodplains, groundwater, critical habitats, and reserves). Each constraint map can be integrated with other constraint maps in different combination, which helps to eliminate unsuitable sites from consideration as possible landfill sites.[37]

Study Area: As one of the Hawaiian Islands, the Island of Oahu is the most populated island (county) in the State of Hawaii. As a jurisdictional unit, the entire island of Oahu is in the City & County of Honolulu (Oahu County) administrated by a mayor and nine council members. The island of Oahu has a total land area of 599.7 square miles or 1545.45 square kilometers. According to the U.S. Census, population of the City & County of Honolulu is about 0.95 million people (70.1% of the State's population) in 2010. Average temperature ranges from 69.4-80.6 (°F) or from 20.78 to 27 (°C).

As stated earlier in Chapter 1 of this research, the City & County of Honolulu (Oahu County) had site selection concerns and problems of site selection problems. However, data are available to assist in the landfill selection process. The GIS-based analysis is applied to: (i) the City's 45 potential landfill sites (Scenario 1) and (ii) the entire island of Oahu (Scenario 2). Results are then compared between them.

This research uses ArcGIS9.2, an integrated collection of GIS software products for building and deploying a complete GIS. ArcGIS utilizes analytical tools (e.g., querying, buffering, reclassifying, and integrating) to incorporate and manage the restrictive criteria. The following GIS data sources are compatible with the restrictive criteria:

- USGS Hawaii Data Clearinghouse (http://hawaii.wr.usgs.gov)
- Statewide GIS Layers (http://www.hawaii.gov/dbedt/gis/download.htm)

Table 3.2 summarizes digital data used for the GIS analysis. Some data (e.g., the NO Pass Zone) are sourced from scanned image maps provided by the City, which are geo-referenced and digitized. All data are set in the same

---

[37] Determining weights are quite controversial and occasionally subjective because different people may view the problem differently (Heywood et al. 2002). Different ordering of criteria can also lead to different results (Luthbom and Lagerkvist 2003). Equal weights for criteria are recommended for preliminary screening because they do not affect results despite different orderings in applying criteria (Daneshvar et al. 2003).

coordinate system or projection (NAD83, UTM Zone 4).[38] A boundary map (the coastline on Oahu, Hawaii) sets the extent of the study area.

The Public Use Airports database provides a geographic point database of aircraft landing facilities in the City & County of Honolulu including Honolulu International Airport, Kalaeloa Airport (or Barbers Point Naval Air Station), Dillingham Airfield, Wheeler Army Airfield, and Kaneohe Marine Corps Air Station. Buffer areas of 10,000 feet (or 3030 meter) are created around each airport or airfield to protect aircraft from collision with birds.

The Federal Emergency Management Agency (FEMA) Digital Flood Insurance Rate Maps (DFIRMs) of Federal Emergency Management Agency (FEMA) depicts flood risk information that shows the extent to which areas in the City & County of Honolulu are at risk from flooding. Proposed landfill sites with significant areas inundated by 100-year flooding (1% annual chance with flooding) are excluded.

National Wetlands Inventory Maps of the U.S. Fish and Wild Service (FWS) provide current geospatially referenced information on the status, extent, characteristics and functions of wetlands, riparian, and aquatic habitats in priority areas to promote the understanding and conservation of these resources.[39]

Land Use and Land Cover (LULC) data consists of historical land use and land cover classification data based on the interpretation of 1970's and 1980's aerial photography and land use maps and surveys. Developed land is characterized by human modification and for human use, such as where the land-surface is covered by structures and prepared materials. Urban or build-up land (e.g., residential and commercial land) is incompatible with landfill sites.

---

[38] The North American Datum (NAD) 83 and Universe Transverse Mercator (UTM) in Zone 4 are used for the island of Oahu.

[39] Wetlands are transitional lands between terrestrial and aquatic systems in which the water table is usually at (or near) the surface or shallow water covers the land. These wetlands provide the functional value: (i) to support fish and wildlife habitats; (ii) to provide aesthetic and recreational value; (iii) to prevent flooding; and (iv) to sustain aquatic diversity (City & County of Honolulu 2006).

Table 3.2. Summary of Input Layers Used in the GIS Analysis

| Layer Name | Source Map | Description |
|---|---|---|
| Coastline | 1:24,000 scale U.S. Geological Survey (USGS) Digital Line Graphs | Background map |
| Potential Landfill Sites | City & County of Honolulu (2003) | 45 potential sites based on tax map key (TMK). |
| Airports | Federal Aviation Administration (FAA) (2006) | The Public Use Airports database is a geographic point database of aircraft landing facilities in the United States and U.S. Territories. |
| Wetlands | 1:24,000 National Wetlands Inventory Maps | U.S. Department of the Interior, Fish and Wildlife Service (FWS) |
| Floodplains | Scanned and compiled by Federal Emergency Management Agency (FEMA) from 1:24,000 | Digital Flood Insurance Rate Maps (DFIRMs) |
| Groundwater (UIC) | City & County of Honolulu (2003) | The analog map is georeferenced and digitized. |
| Groundwater (No Pass Zone) | City & County of Honolulu (2003) | The analog map is georeferenced and digitized. |
| Land Use/Land Cover* | (i)1:100,000 Digital Geographic Information Retrieval and Analysis (GIRAS) files (1976) and (ii) Land Use Maps and Survey | Historical land use and land cover classification data |
| Reserves | Various sources by State of Hawaii (SOH), Department of Land and Natural Resources (DLNR), and Department of Forest and Wildlife (DOFAW) | |
| Critical Habitat | U.S. Fish and Wildlife Service (FWS) (2004) | The data sets define the boundaries for the habitat needed for the species to recover to a normal distribution. |
| Soil | Natural Resources Conservation Service (NRCS) (2007) | Soil Data Viewer is used for analyzing soil properties for landfill construction. |

* High Resolution Land Cover data (2005) by NOAA (National Oceanic & Atmospheric Administration) was produced for Oahu County. However, ArcGIS9.2 could not load the data with 11.2 megabytes.

Datasets of critical habitat define the boundaries for habitats needed by species to recover to a normal frequency distribution. Under the Endangered Species Act, the U.S. Fish and Wildlife Service (FWS) designates critical habitat for threatened and endangered species whenever it is determined to be prudent and determinable. Five areas of critical habitat are designated with approximately 66,350 acres for the Elepaio and for 99 plant species on the island of Oahu.

Data for reserves are compiled by various sources including State of Hawaii, Department of Land and Natural Resources (DLNR), Division of Forest and Wildlife (DOFAW). Activities within natural reserves (e.g., state forests, national parks and historic sites, state parks, and recreation areas) should be regulated in order to prevent environmental contamination.

Data for soil is sourced from USDA Natural Resources Conservation Service, Hawaii (2007). These data can be utilized in order to analyze geographic characteristics (Scenario 1) and to create thematic soil maps showing what map units are suitable or limited for a use of a sanitary landfill (Scenario 2).

GIS layers that are compatible with exclusionary criteria are shown in Figure 3.2. One should note that each layer (constraint map) can be integrated on each other or in any combination because of the assumption of equal weights for the exclusionary criteria.

As will be shown in Chapter 4, the GIS-based analysis is conducted on the City's 45 potential sites (Scenario 1) and the entire island of Oahu (Scenario 2). Together, both scenarios found 7 sites. Benefits transfer (BT) methods (e.g., meta-analysis and mean transfer value approaches) are utilized to measure social costs for the selected 7 target sites and rank these sites according to the social cost minimization. Before discussing benefits transfer (BT) methods, the following section will discuss meta-analysis based on HPM studies. The meta-analysis transfer functions are applied to measure social costs for the selected sites.

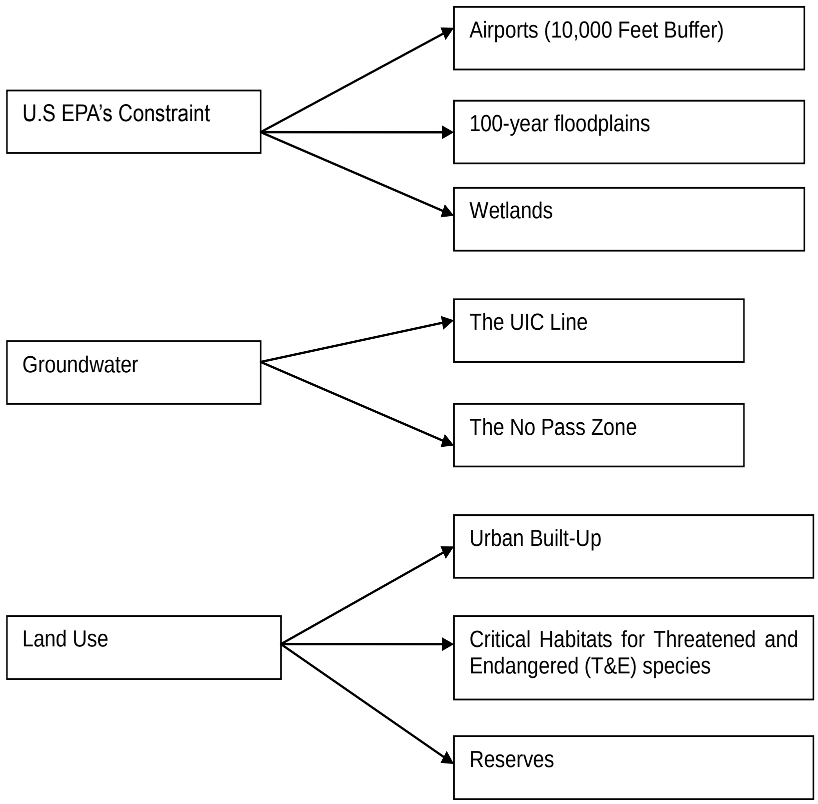



Figure 3.2. GIS layers based on restrictive or exclusionary Criteria

## 3.3. Meta-Analysis (MA)

Meta-analysis (MA) is a statistical method used to examine empirical results from common primary hedonic price method (HPM) studies (e.g., marginal willingness to pay for distance).[40] The meta-analysis (MA) examines systematic relationships between marginal willingness to pay (MWTP) for distance and explanatory variables including core economic variables (e.g., income and population density) and study design variables (e.g., functional forms, the number of observations, and standard errors of each selected estimate) (for a general

[40] Meta-analysis has been applied in non-market valuation studies based on HPM research: Smith and Huang (1995) on air pollution; Nelson (2004) on noise; Brisson and Pearce (1995), Ready (2005), and Walton et al. (2006) on disamenities from landfill sites. For technical aspects of meta-analysis, refer to Hedges and Olkin (1985) and Lipsey and Wilson (2001).

idea, see Stanley and Jarrell 1989; Bergstrom and Taylor 2006). Core economic variables based on economic theory are key factors to reflect characteristics for target sites, and study design variables adjust for different MWTP estimates among studies that result from dissimilar research designs (Bergstrom and Taylor 2006). The estimated meta-analysis functions are then utilized for measuring social costs for target sites.

### 3.3.1. Data

For commodity consistency, welfare change consistency, and theory consistency suggested by Bergstrom and Taylor (2006), this study focuses on HPM studies that estimate a marginal implicit price (MIP) or marginal willingness to pay (MWTP) for distance from a landfill. After excluding 2 observations which are theoretically inconsistent negative MWTP, nine HPM studies provide a total of 22 MWTP estimates (for HPM studies and data used for MA, see Appendix A of current research).[41] The objective of the meta-analysis (MA) develops provisional functions for benefits transfer (BT), which are later employed to estimate aggregate MWTP or social costs for target sites on Oahu.

The dependent variable ($MWTP_i$) is the ith marginal willingness to pay (MWTP) estimate drawn from primary HPM studies (i.e., the percentage change in housing values at mile increments from landfill sites):[42]

$$\mathrm{MWTP_i} \equiv \frac{\partial P_i}{\partial D} / \mathrm{P_i} \qquad (3\text{-}2)$$

where $\mathrm{P_i}$ is average housing values in the original model of ith MWTP estimate

[41] When 2 observations are excluded, regression fits and individual significances were improved. The estimated parameters had signs expected and reasonable diagnostic results (e.g., heteroskedasticity, normality, model specification, and multicollinearity).

[42] In equilibrium the marginal implicit prices (MIP) based on HPM represents household's MWTP, which requires assumptions without housing market distortions (e.g., full information on housing prices and attributes, zero transaction costs and moving costs, and instantaneous price adjustment to changes in either demand or supply) (Freeman 2003).

for i = 1 to n, and the variable D represents the distance from a landfill (Walton et al. 2006; Ready 2005).

Given insufficient studies to report the estimates, the current research includes multiple estimates from a single study (9 studies providing a total of MWTP estimates). The inclusion of multiple MWTP estimates from a single study is debatable. While a single estimate per primary study can create an independent set of MWTP estimates (recommended by Lipsey and Wilson 2001), this has a disadvantage of ignoring potential meaningful information and incurs an unacceptable small samples for the meta-analysis (MA). While the inclusion of multiple estimates from a single study can boost sample sizes, the data structure is similar to unbalanced panel data (for details, see Nelson & Kennedy 2009).[43] The issue and test for panel data are later discussed.

Before conducting the meta-analysis (MA), the homogeneity test checks the following null hypothesis (i.e., MWTP estimates are statistically identical):

$H_0$: $\mathrm{MWTP_1} = \ldots = \mathrm{MWTP_n} = \mathrm{MWTP}$  $\mathrm{H_i}$: $\mathrm{MWTP_i} \neq \mathrm{MWTP}$ for at least one i

where $MWTP_i$ is ith marginal willingness to pay (MWTP) estimate (i = 1, 2,…, n), and MWTP is the mean value (Bergh et al. 1997). If null hypothesis is not rejected, the mean value for MWTP can be applied to the target sites. However, rejecting the null hypothesis implies insignificantly different MWTP estimates. Variation in MWTP estimates may be then adjusted by explanatory variables (Lipsey and Wilson 2001).

In order to test for equivalence of MWTP estimates, the following Q statistic (suggested by Hedges and Olkin 1985) is used:

$$Q = \sum_{\mathrm{i}}^{\mathrm{n}} \mathrm{w_i}\ (\mathrm{D_i} - \overline{\mathrm{D}})^2 \qquad (3\text{-}3)$$

where $D_i$ is the ith MWTP estimate for i = 1 to n (the number of MWTP estimates). The variable $w_i$ is the individual weight for $D_i$ given by the inverse of the variance of each estimate (= $1/\mathrm{s_i^2}$), where $s_i^2$ is the estimated variance of $\mathrm{D_i}$. The variable $\overline{\mathrm{D}}$ is the weighted mean of MWTP estimates ($\sum_{\mathrm{i=1}}^{\mathrm{n}} \mathrm{w_i D_i} / \sum_{\mathrm{i}}^{\mathrm{n}} \mathrm{w_i}$), and its variance (V

[43] Through the survey of 130 meta-analyses, Nelson & Kennedy (2009) revealed that the median study employs three observations per primary study (mean is 6.5).

$(\bar{D}))$ is $1/\sum_i^N w_i$. The Q statistic is distributed as a chi-square with n-1 degrees of freedom, where n is the number of MWTP estimates. In the dataset, the weighted mean $(\bar{D})$ of MWTP is 3.72%, and its variance $(V\ (\bar{D}))$ is 0.178.[44] The null hypothesis is rejected at the 1% significance level (Q statistic is 167.62). As stated in the empirical findings (refer to Chapter 2 pp 47-53), it would be expected that MWTP estimates are insignificantly different.

The following explanatory variables are utilized to explain different MWTP estimates based on previous studies (for data description, see Table 3.3):

- Core economic variables: median household income (Y) (Smith and Huang 1995; Walton et al. 2006) and population density (POP) (Bergstrom et al. 2006; Walton et al. 2006).
- Study design variables: the number of observations (N), standard errors of each selected estimate (SE) (Stanley and Jarrell 1989), functional forms (FUNCTION) (Smith and Huang 1995), types of a landfill (MSW), and activity level of a landfill (ACTIVE) (Walton et al. 2005).

Data sources: MWTP and study design variables were based on other primary HPM studies obtained by journal searches, and several extensive literature surveys including reviews by Brisson and Pearce (1995), Ready (2005), Walton et al. (2006), Farber (1998), and Boyle and Kiel (2001). Data are shown in Appendix A.1 based on nine studies (Nelson et al. 1992 a, b; Nelson et al. 1997; Ready 2005; Lim and Missios 2003; Bouvier et al. 2000; Reichert et al. 1992; Kiel and McClain 1995; Thayer et al. 1992). These studies were selected following the standard for valid and reliable benefits transfer (Bergstrom and Taylor 2006; Smith et al. 2002): (i) welfare measure consistency (e.g., hedonic price methods), (ii) commodity consistency (e.g., distance from municipal solid waste landfills), and (iii) theory consistency (e.g., positive distant effects). All studies examined the impact of municipal solid waste (MSW) landfill sites on housing values by utilizing distance from the landfills except for Kiel and McClain

[44] Critical $\chi^2$ value ($\alpha = 0.05$) for 21 degrees of freedom is 32.671, and critical $\chi^2$ value ($\alpha = 0.01$) for 21 degrees of freedom are 38.93.

(1995) which examined the impact of an incinerator on housing values. The variable MSW was utilized in order to reflect the difference between MSW landfills and incinerators. If primary studies did not provide data (e.g., median income levels and population density at original primary study sites), supplemental data were collected from the U.S. Census.

Table 3.3. Description of Data for Meta-Analysis

| Category of Variables | Variables | Description | Expected Sign |
|---|---|---|---|
| Dependent Variable | MWTP | For each mile away from landfill sites, the percentage change in housing values | |
| Core Economic Variables (C) | Y | Median Household Income | Positive |
| | POP | Population density (population per square mile) | Positive |
| Study Design Variables | MSW | Dummy variable: 1 if a landfill disposes of MSW, 0 otherwise. | Mixed |
| | ACTIVE | Dummy variable: 1 if a landfill is active; 0 otherwise | Positive |
| | N | The number of observations from each selected study. | Mixed |
| | SE | Standard error of each selected estimate from primary research | Mixed |
| | FUNCTION | Dummy variable: 1 if a function is linear; 0 otherwise. | Mixed |

* If original studies do not report income and population density, the data are collected at the town levels from US census. Thus, figures may not perfectly match the exact area within 3-mile distance from landfill sites.

### 3.3.2. Estimation methods

#### Ordinary least squares (OLS)

The meta-analysis model examines systematic relationships between MWTP estimates and explanatory variables (see equation 2-11 of this study). If the error term is distributed with the mean zero and constant variance, using the ordinary least squares (OLS) has finite sample properties (e.g., unbiasedness and minimum sampling variance). According to the Gauss-Markov Theorem, the OLS estimator is the best linear unbiased estimator (BLUE) with minimum sampling variance (Johnston and Dinardo 1997). However, using OLS may incur some primal econometric concerns: (i) a panel structure of data that may occur due to multiple MWTP estimates from single studies and (ii) heteroskedasticity caused by primary studies with different data sets, different sample sizes, and different independent variables (Bergh et al. 1997). The OLS estimator would provide biased parameter estimates when used with a panel structure of data (Rosenberger and Loomis 2000b). Using OLS with heteroskedasticity provides unbiased parameter estimates but biased standard errors (Stanley and Jarrell 1989). These primal econometric problems need to be managed.

Unbalanced panel data model

Panel data are defined as regularly or irregularly repeated observations on each cross section unit (e.g., each study).[45] If each unit or panel (e.g., study j) has the same number of repeated observations, it is referred to as balanced panel data. If each study j has different number of observations, it is referred to as unbalanced panel data (Wooldridge 2002). The dataset in this research is similar to unbalanced panel data because nine studies (j = 1 to 9) provide 22 total observations (j = 1 to 4). A generic unbalanced panel model may be defined as

$$\mathrm{MWTP}_{ij} = \beta_0 + \beta_1\mathrm{INCOME}_{ij1} + \beta_2\mathrm{POP}_{ij2} + \beta_3\mathrm{N}_{i3} + \beta_4\mathrm{SE}_{ij4} + \beta_5\mathrm{FUNCTION}_{ij5} + \beta_6\mathrm{MSW}_{ij6} + \beta_7\mathrm{ACTIVE}_{ij7} + \varepsilon_i + u_{ij}, \qquad (3\text{-}4)$$

where j is the jth panel or study for j = 1 to 9, and i is the number of observations on the jth study for i = 1 to 4. The dependent variable $\mathrm{MWTP}_{ij}$ is the ith marginal willingness to pay for the jth study. Explanatory variables (e.g., INCOME, POP, N, SE, FUNCTION, MSW, and ACTIVE) are utilized for adjusting different $\mathrm{MWTP}_{ij}$. The variable $\beta = (\beta_0, \beta_1, \beta_2, \dots \beta_k)'$ is a vector of coefficients of the intercept and explanatory variables. The variable $\varepsilon_i$ is a common error across estimates, and $\mu_{ij}$ is the panel error that may occur when multiple estimates from the same

[45] Panel data are usually called longitudinal data or cross-sectional time series data where each unit or panel (e.g., study) is observed at several periods. Panel data contain more information than a single cross section.

source (study or panel) incur cross sectional correlation or heteroskedasticity (for panel stratification in the meta-analysis, see Rosenberger and Loomis 2000b).

In the panel data structure, using OLS may provide biased parameter estimates (Johnston and Dinardo 1997). Several methods may manage this issue (e.g., a fixed effect model and a random effect model). While the fixed effect model allows correlation between the unobserved panel effect ($u_{ij}$) and the explanatory variables, a random effect model treats $u_{ij}$ as a random variable with zero covariance between explanatory variables and $u_{ij}$ (for details, see Woodridge 2002; Johnston and Dinardo 1997). Before selecting a suitable method, a test for panel effects is suggested if a single study provides multiple observations (Rosenberger and Loomis 2000b).[46]

## Robust standard errors and generalized least squares (GLS)

Heteroskedasticity in meta-analysis is a primal problem, which results from the different findings on the same research problem (Santos 2007). Although the inclusion of study design variables (e.g., N, SE, FUNCTION, MSW, and ACTIVE) and key economic variables (e.g., Y and POP) adjust for differences in MWTP estimates, heteroskedasticity often persists (Walton et al. 2006; Smith and Huang 1995). With heteroskedasticity, parameter estimates using OLS are unbiased. However, standard errors for the coefficients are biased, and test statistics based on these biased standard errors (e.g., t statistics and F statistics) are invalid (Walton et al. 1996; Nelson and Kennedy 2009). This heteroskedasticity can be managed in several ways (e.g., generalized least squares (GLS), robust standard errors, and data transformation).

For convenience, the linear equation with n sample observations is defined as follows:

$$y = X\beta + u \qquad (3\text{-}5)$$

[46] The panel effect can be avoided by providing a single estimate per study because i = j and $\mu_{ij}$ collapses into $\varepsilon_i$ (Lipsey and Wilson 2001).

where y = $\begin{pmatrix} y_1 \\ y_2 \\ \vdots \\ y_n \end{pmatrix}$, X = $\begin{pmatrix} 1 & X_{11} \cdots & X_{1k} \\ \vdots & \vdots \quad \ddots & \vdots \\ 1 & X_{n1} \cdots & X_{nk} \end{pmatrix}$, β = $\begin{pmatrix} \beta_0 \\ \beta_1 \\ \vdots \\ \beta_k \end{pmatrix}$, and u = $\begin{pmatrix} \mu_1 \\ \mu_2 \\ \vdots \\ \mu_n \end{pmatrix}$

y is a n×1 vector of MWTP estimates, X is a n×(k+1) matrix of explanatory variables and a column of ones to allow for the intercept, β is a $(k+1) \times 1$ vector of coefficients, and u is a n×1 vector of disturbances. Unlike the assumption of homoskedasticity (equal variance i.e., $\sigma^2 I$ where I is an nxn identity matrix), the error term $u$ is heteroskedastic (i.e., $\sigma^2\Omega$ where $\Omega$ is a known nxn positive-definite and nonidentical matrix). With homoskedasticity, the OLS estimator and its variance-covariance term are $\hat{\beta} = [X'X]^{-1}X'y$ and $V(\hat{\beta}) = \sigma^2 I$. However, with heteroskedasticity, the OLS estimator and its variance-covariance term provide $\hat{\beta} = [X'X]^{-1}X'y$ and and $V(\hat{\beta}) = [X'X]^{-1}X'\sigma^2\Omega X[X'X]^{-1}$. With heteroskedasticity, OLS based on erroneous assumption of homoskedasticity provides biased standard errors for OLS coefficients (Johnston and Dinardo 1997).

There are several ways to remedy this heteroskedasticity (e.g., data transformation, GLS or feasible GLS, and robust standard errors). One method is to transform data (the semi-log or log-linear forms) to remedy heteroskedasticity and non-normality of residuals (Wooldridge 2003).

The other method, GLS estimator can correct for heteroskedasticity from a transformed model in which the error terms meet classical OLS assumptions. For example, a nonsingular, symmetric, and idempotent matrix P can be found such that $P'P = \Omega^{-1}$ and $\Omega = (PP')^{-1}$. When P is premultiplied in the regression model, $Py = PX\beta + Pu$, where $E(Pu) = 0$ and $V(Pu) = \sigma^2 P\Omega P' = \sigma^2 I$. The GLS estimator and its covariance matrix are as follows:

$$\hat{\beta} = [X'\Omega X]^{-1}\Omega^{-1}y \text{ and } V(\hat{\beta}) = \sigma^2[X'\Omega X]^{-1} \qquad (3\text{-}6)$$

The GLS estimator is the best linear unbiased estimator (BLUE) that contributes to valid hypothesis testing (Johnston and Dinardo 1997). While $\Omega$ is usually unknown, the feasible GLS estimator is used to estimate the unknown component ($\Omega$). The feasible GLS estimator shares the same asymptotic properties as the GLS estimator (e.g., consistency, asymptotic normality, and asymptotic efficiency if errors are normally distributed (for the process of feasible GLS, see Wooldridge 2002; 2003).

Robust standard error estimators are often utilized to correct for heteroskedasticity (Smith and Huang 1995; Wooldridge 2002). The estimated variance-covariance matrix of $\hat{\beta}$ and the robust standard error estimators are as follows:

The estimated var $(\widehat{\beta})$ = $[X'X]^{-1}X'\sigma^2\hat{\Omega}X[X'X]^{-1}$ (3-7)

$\sigma^2\hat{\Omega} = diag\{e_1^2, e_2^2, \cdots, e_n^2\}$

where var $(\widehat{\beta})$ is the variance-covariance term of $\widehat{\beta}$, $e_i$ is the OLS residuals for i= 1 to n, and diag denotes diagonal terms of the estimated variance-covariance matrix for $\widehat{\beta}$. The square root of the elements on the principal diagonal of estimated var $(\widehat{\beta})$ are called the White standard errors or Huber-standard errors (for details, see White 1980). Robust standard errors method is particularly useful without requiring any specific assumptions about the form of the heteroskedasticity. With a large sample size, robust standard errors are suitable even under homoskedasticity because this method provides asymptotically valid t statistics, F statistics, and significance levels for hypothesis testing (Johnston and Dinardo 1997). Although the properties are unknown in small samples, researchers often utilize the robust standard error for a comparison purpose.

### 3.3.3. Tests for panel effects and heteroskedasticity

#### Test for panel effects

As suggested by Rosenberger and Loomis (2000b; 2003), panel effects were tested by modeling each study as a panel for a fully specified model. The Breusch-Pagan (BP) Lagrange Multiplier test was utilized for testing for the null hypothesis that panel effects ($u_{ij}$) do not exist:

$H_0: u_{ij} = 0$ versus $H_a: u_{ij} \neq 0$.

The Breusch–Pagan (BP) statistics for single linear and semi-log (dependent variable) forms failed to reject the null hypothesis at the 5% level (p- values are respectively 0.1552 and 0.0850). The statistically insignificant panel effects allow the OLS to be used for both functional forms. The Hausman test for choosing a fixed effect model versus a random effect model is not applicable (see Figure 3.9).[47]

Test for heteroskedasticity

It is important to test for heteroskedasticity, which is a primal problem in meta-analysis. There are several ways to test for heteroskedasticity (e.g., the White test, the Breusch-Pagan (BP) test, and Breusch-Pagan / Cook-Weisberg test) (Wooldridge 2003). The Breusch-Pagan/Cook-Weisberg (BP-CW) test was used for testing for the null hypothesis of homoskedasticity. The BP-CW test for a simple linear form including all variables rejects the null hypothesis at the 1% level (p-value $\leq$ 0.0000). Heteroskedasticity seems to be present (see Table 3.4).

There is no precedent for choice of a functional form when using the meta-analysis (Rosenberger and Loomis 2000b). However, data transformation (the semi-log or log-linear forms) is one way to remedy heteroskedasticity and non-normality of residuals (Wooldridge 2003). The BP-CW test for a semi-log (dependent) form including all variables fails to reject the null hypothesis (p-value = 0.8424) (see Table 3.4). Thus, the semi-log form is homoskedastic.

This study will utilize the OLS method since panel test results show that panel effects are statistically insignificant at 5% level. As shown by the heteroskedasticity test, the simple linear form was heteroskedastic. Thus, a semi-log form is employed to correct for heteroskedasticity. For a comparison purpose, robust standard errors are also utilized.

[47] Rosenberger and Loomis (2000b; 2003) also found that panel effects were insignificant.

Table 3.4. Heteroskedasticity Test

| | Test Name | $\chi^2$ statistic | p-value | Degrees of Freedom | $H_0$: Homoskedasticity $H_a$: Heteroskedasticiy |
|---|---|---|---|---|---|
| Simple Linear | BP-CW | 31.35 | 0.0000 | 1 | Reject $H_0$ |
| Semi-Log (LogMWTP) | BP-CW | 0.04 | 0.8424 | 1 | Fail to reject $H_0$ |

* Critical $\chi^2$ value (α = 0.05) is 3.84 for DF = 1.

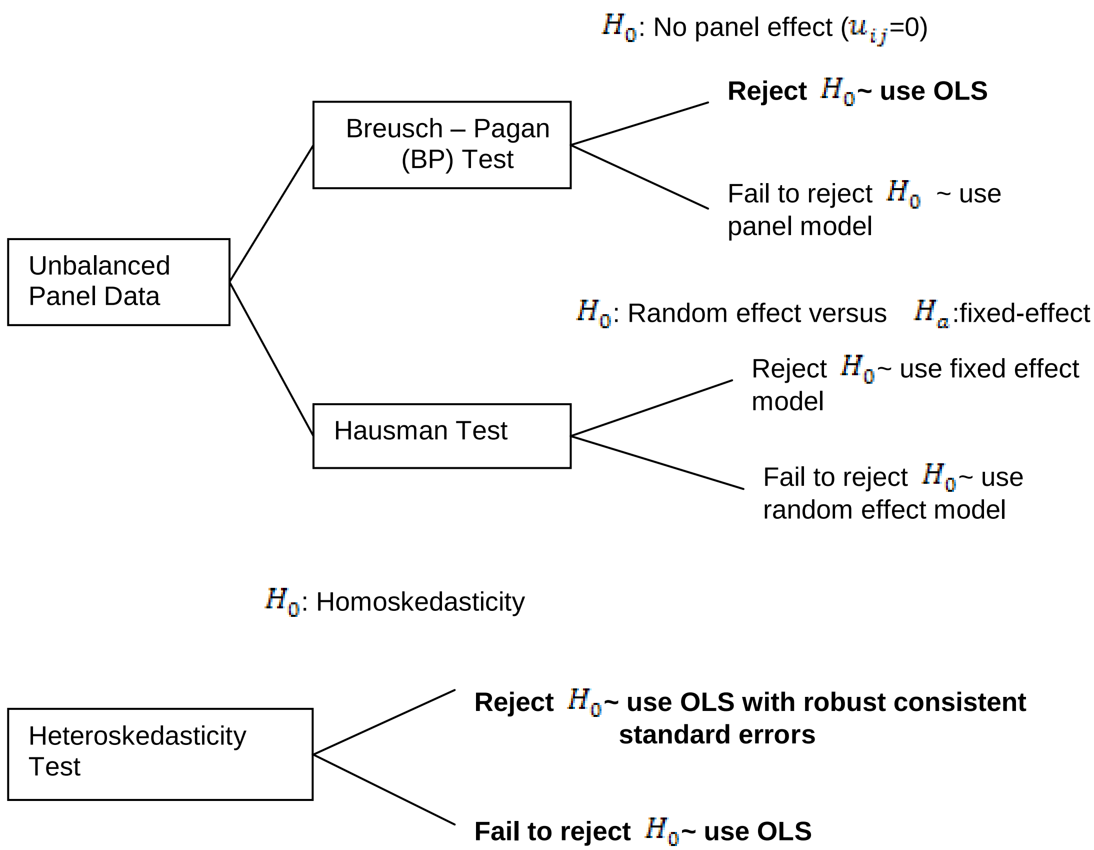



Figure 3.3. Estimation method choice

### 3.3.4. Tests for hypotheses and diagnostics

Hypothesis tests concerning parameters strength the confidence of OLS models. The F test assesses the null hypothesis that all coefficients (β) on the model's explanatory variables equal zero i.e., $H_0$: $\beta_1 = \beta_2 = \cdots = \beta_k = 0$ for all coefficients (Wooldridge 2003). This test will assure overall significance of the regression model. The t-test tests the null hypothesis that each individual coefficient equals zero i.e., $\mathrm{H}_0$: $\beta_j$ =0 for each j = 1, 2, …, k (Hamilton 2004).

Given a small sample size, conducting various diagnostic tests is important for meta-analysis equations (suggested by Walton et al. 2006). Together with the $R^2$ for the overall fit of the regression and hypothesis tests (F tests and t tests), diagnostic tests (e.g., skewness-kurtosis normality test, Ramsey's RESET test for the specification error bias, heteroskedasticity test, and multicollinearity assessment) are reported in Chapter 4.

Together with homoskedasticity, normality of residuals should be assured for valid hypothesis tests (e.g., F and t-tests). Among several tests for normality (e.g., the Jarque-Bera test, the Shapiro-Wilk test, and the skewness-kurtosis test), this study utilizes a skewness-kurtosis test evaluating the null hypothesis that residuals are normally distributed. Skewness characterizes the degree of asymmetry of a distribution around its mean. Kurtosis characterizes the relative peakedness or flatness of a distribution compared to the normal distribution (Hamilton 2004). The p-values for skewness-kurtosis normality tests are reported.

An important issue when working with small samples is the potential for multicollinearity between explanatory variables (Walton et al. 2006). High correlation between explanatory variables in small samples can produce possible concerns: (i) substantially higher standard errors with lower t statistics (a greater chance of falsely accepting the null hypothesis in standard significance tests); (ii) unexpected changes in coefficient magnitudes or signs; and (iii) statistically insignificant coefficients despite a high $\mathrm{R}^2$ coefficient (Hamilton 2004). A number of tests to detect multicollinearity exist (e.g., Durbin-Watson tests, VIF, Tolerance, and a correlation matrix between estimated coefficients). One test variance inflation factor (VIF) is utilized in this research to measure the degree to which the variance and standard error of an estimated coefficient increase because of the inclusion of the explanatory variable (i.e., the higher the value of VIF, the greater is the degree of multicollinearity) (Wooldridge 2000). With over 10 of variance inflation factor (VIF) values, the variable could be collinear with other variables (Hamilton 2004). The correlation matrix for explanatory variables is also examined to determine the presence and severity of multicollinearity.

If a relevant functional form is not used on the model, the estimates are biased (a functional form misspecification). The associated standard errors, t statistics, and F statistics are then incorrectly specified (Wooldridge 2002, 2003). This study employs the Ramsey regression specification error test (RESET), which utilizes powers of the fitted values of $\hat{y}$. For example, the dependent variable y is regressed on the x variables and the second, third, and fourth powers of $\hat{y}$ (predicted values). An F test checks the null hypothesis that all three coefficients on those powers of $\hat{y}$ equal zero. If this null hypothesis is rejected, further polynomial terms could improve the model (Hamiltion 2004). The p values for the RESET test are then reported.

Meta-analysis models (e.g., a fully specified model and a restricted model) with reasonable diagnostic test results are developed to measure social costs for target sites on the island of Oahu. Results from the models are compared regarding their performance in benefits transfer (BT) analyses.

### 3.4. Benefits transfer (BT) for measuring disamenity effects from landfill sites

Planners usually estimate direct costs for landfill site selection (e.g., costs of site development, operation, and site acquisition), but they do not consider external costs for proposed landfills. If external or social costs are high, they will likely underestimate costs of the landfills and possibly locate the landfill in a higher overall cost location. The measurement of external costs helps planners to locate a landfill or allocate more efficiently solid waste resources. Table 3.5 shows direct costs for the selected 4 sites (Scenario 1 in Chapter 4) on the island of Oahu. These direct costs range from $15 million to $82 million ($0.87 million to

$8.87 million per year) (for details, refer to the Attachment D, City and County of Honolulu 2003).

Benefits transfer (BT) is a research method which employs data or functions from other primary studies to a different or transfer context in order to calculate values (Desvousges et al. 1998). The location where the primary research was conducted is referred to as the study site, and the location where a new policy is implemented or value measurement is utilized is referred to as the policy or target site (e.g., the 7 sites selected from the GIS analysis) (Rosenberger and Loomis 2003). When planners have encountered difficulty in conducting primary hedonic price method (HPM) or contingent valuation method (CVM) research (e.g., time and monetary constraints), benefits transfer (BT) can provide a possible measure which approximates the impacts of proposed landfill sites and ranks the selected target sites. BT also saves time and reduces monetary budgets by utilizing information or empirical findings from other primary studies.

Table 3.5. Estimated Direct Cost of Potential Landfill Sites on Oahu ($): Present Value*

| Target Sites | Landfill Capacity (Years) | Cost of Site Acquisition | Cost of Development | Cost of Operation | Total | Annual Value |
|---|---|---|---|---|---|---|
| Site A | 13 | $56,735,035 | $10,428,190 | $12,831,855 | $79,995,080 | $8,879,852 |
| Site B | 25 | $14,265,477 | $10,822,033 | $10,430,069 | $35,517,579 | $2,667,430 |
| Site C | 16 | $530,744 | $8,814,909 | $15,243,642 | $24,589,295 | $2,130,928 |
| Site D | 20 | 0 | $2,887,128 | $12,233,359 | $15,120,487 | $877,582 |

Source: City and County of Honolulu (2003)
*Annual values were calculated by utilizing an equivalent annual value (EAV) method for comparing other projects with unequal time lengths adjusted in 2008 year based on (i) a 3% real discount rate recommended by U.S. EPA. (Freeman 2003); and (ii) the Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars
** Sites E, F, and G in Scenario 2 in Chapter 4 do not have data or information about direct costs.

In order to measure social costs, this study utilizes two approaches to benefits transfer (BT) analysis: (i) a meta-analysis (MA) based on primary HPM studies and (ii) a mean transfer value approach from primary CVM research. Either CVM or HPM research can act as a reliability check on the other by ranking the selected 7 target sites according to social cost minimization (Brisson and Pearce 1995). A resident's decrease in income in order to avoid a proposed landfill reflects the resident's WTP or marginal WTP, and a sum of these estimates represents society's costs (social costs). Present values of total values represent a stream of social costs over time. In order to compare sites or projects with unequal time lengths, equivalent annual value (EAV) method is utilized to transform the present values to an annual basis (Boardman et al. 2006). For a theoretical basis, refer to Appendix B of this study. For example, total WTP or MWTP represent social costs for target sites A to F, and larger total WTP or marginal WTP induce larger social costs. Thus, the selected 7 sites are ranked where site #1 has the lowest total value, and #7 has the highest total values (Swallow et al. 1992).

### 3.4.1. Mean transfer value for WTP approach

A mean transfer value for WTP approach based on CVM research is utilized for comparing a MA-BT method. The major steps for a mean transfer value for WTP approach adapted by Rosenberger and Loomis (2003) are as follows: (i) define the transfer context or target sites; (ii) find original research outcomes through a literature review; (iii) screen the unsuitable studies for the transfer context; (iv) select a mean value for WTP; and (v) transfer the values.

Step (i): define the transfer context

The transfer context is defined as the 7 target sites on Oahu (determined from the GIS analysis), and measuring social costs for each target site is needed. In order to compare this mean transfer value approach with the meta-analysis transfer function, each target site requires information including a mean transfer value for WTP per household, income levels, the number of affected households, and the length of landfill life in year. Following Brisson and Pearce (1995), it is assumed that the impacts of the proposed landfill site are within a 3-mile distance of the target sites. Thus, ArcGIS 9.2 identified census data (e.g., income levels and the number of households) based on census tracts within a 3-mile distance of each target site on Oahu (for data, see Table 3.6).[48] The length of landfill life in year is given by the City and County of Honolulu. If the length of landfill life is unknown, it is calculated following the City's method (see p 113 in Chapter 4). Transfer information needed for the 7 target sites on Oahu includes a mean value for WTP per household to avoid a proposed landfill from other primary research, income levels at each study site adjusted for income differences between study and target sites, and a base year for adjusting monetary values over time.

Steps (ii) and (iii): review literature to find the original research outcomes and screen the unsuitable studies for the target sites on the island of Oahu

The proposed landfill can contribute to negative external effects (e.g., disamenities, air pollution, land pollution, and water pollution, sedimentation, and global warming). While benefits provided by a new landfill site are shared by all citizens, harm from a landfill concentrates on nearby residents (Kunreuther and Kleindorfer 1986; Mitchell and Carson 1986). Despite various environmental impacts, disamenity is often considered as one component of externality effects because the effects from landfills are site specific.

In order to estimate disamenity costs or external costs from landfills, a revealed preference method (e.g., HPM (hedonic price method) and/ or a stated preference method (e.g., CVM (contingent valuation method)) is used (Brisson and Pearce 1995). While few studies (Roberts et al. 1991; Smith and Desvousges 1986b; Opaluch et al. 1993) conducted CVM, most studies utilized HPM in order to analyze how housing prices reflect the individual's willingness to pay (WTP) for avoiding disamenities form a landfill by using distance from a

[48] Utilizing GIS can improve accuracy of BT by providing data or analyzing site characteristics. For use of GIS in BT methods, see Bateman et al. (2005) and Lovett et al. (1997).

landfill site. Conducting primary HPM is not possible for the 7 target sites without landfills. Although CVM can measure the impacts of proposed landfills, performing CVM for each target site requires substantial time and large money. Thus, this study employs benefits transfer (BT) methods because of time and monetary budget constraints for conducting primary valuation research.[49]

A literature review is utilized to find a CVM study which fits the target sites on Oahu and can be compared with the MA-BT approach. A few studies were found (Roberts et al. 1991; Smith and Desvousges 1986; Opaluch et al. 1993). See Chapter 2 for summaries of these studies. Smith and Desvousges (1986b) estimated consumer surplus per household on a hazardous waste site. Opaluch et al. (1993) developed weights for site characteristics (e.g., groundwater quality, wildlife habitats, and types of access road) and location characteristics (e.g., presence of schools and parks within a 4 square mile area of the landfill) based on the contingent choice method. However, they did not provide information on WTP to avoid a landfill site. Application of these weights to the target sites on Oahu is not feasible because of inaccessible data. It will be also difficult to compare social costs from non-monetary characteristics estimated by the MA functions based on the HPM research. The study by Roberts et al. (1991) was the most relevant to the target sites since they provided a mean value for WTP to avoid a proposed municipal solid waste (MSW) landfill and suitable information about the study year and income levels. The annual mean value for WTP per household at the study site (Knox County, Tennessee) was $410 with 95 percent confidence intervals of $310 to $543. Per capita income and median household income were respectively $28,526 and $48,619.[50]

Steps (iv) and (v): select a mean value for WTP and transfer the values

---

[49] HPM studies are suitable for meta-analysis for benefits transfer (MA-BT) because most studies have utilized HPM.

[50] The Honolulu Consumer Price Index (CPI) is used to adjust monetary values for 2008 US dollars.

Equation (2-8) in Chapter 2 for the mean transfer value for WTP approach is utilized to measure annual mean value for WTP per household. For sensitivity analysis, this study considers both cases for e=1 and e=0.3 (see pp 61-62). The primary CVM study by Roberts et al. (1991) provides information about $WTP_s$ and $Y_s$, and Table 3.6 provides data for $Y_p$ at target sites. When $WTP_s$, $Y_p$, and $Y_s$ are inserted into equation (2-8), the annual mean value for WTP per household at each target site can be estimated.

In order to determine annual aggregate WTP at the target sites, the following formula is used (Roberts et al 1991):

Aggregate $\mathrm{WTP_p}(t) = \mathrm{WTP_p}(t) * N$ (3-8)

where $\mathrm{WTP_p}(t)$ is the estimated average annual WTP per household for the target sites in the 2008 year, and N is the number of affected households for the target sites (for data at target sites, see Table 3.6). Multiplying the target site's $\mathrm{WTP}_p(t)$ by the number of affected households (N) yields estimates of annual total $\mathrm{WTP_p}(t)$ at the specific time. The aggregate annual values are utilized in order to compare those of target sites or projects with unequal time lengths.

The length of landfill life (in terms of year) can affect social cost differences since a future value is worth less today. Present values (PV) of a stream of total $\mathrm{WTP_p}(t)$ over time are calculated as follows (Roberts et al. 1991):

$PV = \sum_{i=0}^{T} \frac{1}{(1 + r)^{t}} *$ aggregate $\mathrm{WTP_p}(t)$ (3-9)

where aggregate $WTP_p(t)$ is annual aggregate WTP in year t for t = 1 to T, r is a discount rate (a 3% discount rate), and T is the terminal year for each target site. For a comparison purpose, the present values are also used to rank target sites (for details, see Appendix B).

Table 3.6 shows data for the target sites on Oahu. One interesting observation is that the average median household income levels are much larger than the average per capita income levels. This can be explained by: (i) more working people compared to the average household and/ or (ii) greater income inequality.

Table 3.6. Census Data for Target Sites (2008 US dollars)*

| Target Sites | Years of Life (Lyear) | Number of Household (N) | Per Capita Income (PCI) | Median Household Income (Y) | Median Housing Prices (MHP) | Population Density (POP) ** | Number of Member in Households |
|---|---|---|---|---|---|---|---|
| Site A | 13 | 22,774 | 37,880 | 93,144 | 463,208 | 1587.29 | 3.06 |
| Site B | 25 | 12,376 | 22,906 | 74,400 | 257,494 | 606.37 | 3.77 |
| Site C | 16 | 9,600 | 19,129 | 66,980 | 215,390 | 604.75 | 4.13 |
| Site D | 20 | 9,635 | 22,979 | 74,448 | 269,558 | 517.40 | 3.82 |
| Site E | 20 | 8,789 | 24,347 | 76,988 | 265,607 | 513.67 | 3.69 |
| Site F | 11 | 6,960 | 27,570 | 74,726 | 363,136 | 386.83 | 3.16 |
| Site G | 25 | 19,171 | 34,843 | 88,709 | 447,726 | 1911.16 | 3.13 |

Source: U.S. Census (2000) and City $ County of Honolulu (2003)
* ArcGIS9 identified census data based on census tract within a 3-mile distance of each target site (N, PCI, Y, and MHP). PCI, Y, and MHP are mean values based on the census tracts. The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars.
*** Population densities are people per square mile.

### 3.4.2. Meta-analysis for benefits transfer (MA-BT) approach

This study utilizes a meta-analysis for benefits transfer (MA-BT) approach (suggested by Bergstrom and Taylor 2006) (for theory and practice about the MA-BT model, see Appendix C). The MA-BT method assumes the existence of (i) an underlying valuation function which accounts for target site characteristics and (ii) an envelope function for individual site specific valuation functions estimated from different studies (Rosenberger and Phipps 2007).

An underlying meta-analysis valuation function has a relationship between marginal willingness to pay (MWTP) for distance (average percentage change in housing values) and explanatory variables. Different MWTP estimates are often due to dissimilar research designs (e.g., N, SE, and Function) and characteristics of target sites (e.g., Y and POP). Core economic variables (e.g., Y and POP) are key factors for benefits transfer (BT) reflecting differences between study and/or target sites.[51] Study design variables (e.g., N, SE, FUNCTION, MSW, and ACTIVE) i.e., explanatory variables except for the core economic variables help to improve the predictive power of the MA-BT model by explaining for different

[51] Bergstrom and Taylor (2006) suggested that core economic variables based on economic theory should be included and consistently defined across studies. For this study, median household income and population density across the studies are utilized.

MWTP estimates among studies (Bergstrom and Taylor 2006). By holding the effect of these study design variables (e.g., N, SE, and FUNCTION) constant (use of mean values for these variables in the data), target sites are assumed to have same study design variables (Smith et al. 2002). The use of core economic variables for each target site reflects differences between this study and / or target sites.

Two meta-analysis (MA) models with reasonable diagnostic test results (see Table 4.9 in Chapter 4) are utilized for BT: (i) a fully specified regression (MA model 1), and (ii) a restricted regression excluding MSW and ACTIVE (MA model 2). After inserting the mean values of study design variables (N, SE, and FUNCTION) into the MA models and core economic variables for each target site, average MWTP per household for each target site is measured.[52] The following formula is utilized in order to determine total MWTP estimates for target sites (Freeman 2003):

Aggregate MWTP = B * N (3-10)

where B is average marginal willingness to pay per household (i.e., average housing value changes per each mile away from the target site), and N is the number of households. Total MWTP estimates for each target are calculated when B (average marginal willingness to pay for distance) is multiplied by the number of households (N). Average marginal willingness to pay (B) can be calculated when average MWTP per household (%) estimated by meta-analysis is multiplied by average of median housing values and divided by 100 i.e., $(\frac{\partial P}{\partial D}/P)*P*(1/100)$, where $(\frac{\partial P}{\partial D}/P)$ is average percentage changes in housing values per each mile away from landfills, D is distance form landfills, and P is average of median housing values (for data, see Table 3.6).

Present values of social costs stream over time are considered as follows (Roberts et al. 1991; Pearce et al. 2006):

[52] For MA model 1, the variables MSW and ACTIVE from target sites on Oahu are employed.

$$PV = \sum_{i=0}^{T} \frac{1}{(1 + r)^{t}} * \text{ annual aggregate } \mathrm{MWTP_p}(t) \qquad (3\text{-}11)$$

where PV is a present value of the stream of annual aggregate $MWTP_p$(t) over time for t=1 toT, r is a real discount rate (3%), and T is a terminal year for each target site (landfill's life). For a comparison of other projects or target sites with unequal time length, the equivalent annual value (EAV) method is employed, which transforms the present value to an annual basis (Boardman et al. 2006) (refer to Appendix B of this study). Target sites are ranked according to the social cost minimization.

### 3.4.3. Reliability and validity on meta-analysis models

A fundamental issue in benefits transfer (BT) is validity and/or reliability of transfer values. While validity requires statistical tests (e.g., a parametric t-test and a non-parametric sign rank test) to examine mean differences or differences between transfer values and original values, reliability checks their similarity by utilizing a transfer error measured by an absolute percentage difference (Navrud and Ready 2007).[53] This study utilizes (i) parametric (a t-test) and non-parametric (a sign rank test) tests to check for validity and (ii) an absolute percentage difference to check for reliability (Brouwer and Spaninks 1999; Shrestha and Loomis 2001).

In order to evaluate meta-analysis (MA) transfer functions in terms of validity and reliability criteria, this study utilizes the method used by Lindhjem and Navrud (2008). All data except for those of the target site j are utilized to obtain the N-j meta-analysis transfer function that predicts the transfer value at the jth excluded site (original site). Based on the N-j transfer function, the estimated jth

[53] This reliability check does not require statistical hypothesis tests but transfer error measured by an absolute percentage difference (smaller TE performs better).

transfer value is compared with the jth original value using the primary HPM research.[54]

Table 3.7. Tests for reliability and validity

| | Criteria | Test Statistics |
|---|---|---|
| Test for validity (statistical difference) | $H_0$: mean differences ($\mu_D$) between transfer vales and original values = 0 | 1) Paired t-test |
| | $H_0$: differences between transfer values and original values =0 | 2) Sign rank test test |
| Check for reliability (similarity) | Smaller $\delta$ performs better | Absolute percentage difference ($\delta$) |

First, a parametric t-test and a non-parametric sign rank test are utilized to examine whether transfer values estimated by the N-j transfer functions are statistically different from original values. While a t-test is a common parametric test assuming that population is normally distributed, a sign rank test is a non-parametric test without assuming a normal distribution (Hamilton 2004). Both tests have the null hypotheses of mean differences or differences between transfer values and original values. Statistical significance indicates rejection of the null hypothesis. In terms of validity criteria, the null hypotheses should not be rejected i.e., transfer values are insignificantly different from original values (see Table 3.7).

Second, the N-j transfer function is utilized to calculate a transfer error (TE) measured by an absolute percentage difference in order to check similarity between transfer values and primary values (smaller TE performs better) (Shrestha et al. 2007). One important assumption is that the value estimated from primary or original research is the true value (original or primary value) for the selected target site. The transfer value incurs the associated error ($\varepsilon$) i.e., $\mathrm{WTP_T} = \mathrm{WTP_B} + \varepsilon$ where $\mathrm{WTP_T}$ is the transfer value from the study site to the target site, and $\mathrm{WTP_B}$ is the original value estimated by using primary research (Rosenberger and Loomis 2003). The following equation is employed in order to

[54] Instead of setting the methodological variables (e.g., MSW, ACTIVE, and FUNCTION) at mean values, these variables are set to the same as for the benchmark estimate.

measure transfer error (the absolute percentage difference) (Lindhjem and Navrud 2008; Rosenberger and Loomis 2003):[55]

$$\delta = \frac{|(WTP_T - WTP_B)|}{WTP_B} * 100 \qquad (3\text{-}12)$$

where $WTP_T$ is the transfer value from original sites to target sites, $WTP_B$ is the original value at target sites, and $\delta$ is the transfer error (TE) measured by the absolute percentage difference between the transferred value and the original value. In terms of reliability criteria, smaller $\delta$ suggests better performance.

For a comparison, the mean MWTP for the N-j data used in meta-analysis is also used in calculating TE for each site and mean TE. A comparison between BT methods (e.g., direct mean transfer value versus meta-analysis transfer function approaches) may provide valuable information on process reliability and the level of error. Sensitivity of TE and mean TE is assessed by sorting the TE with income levels (Y) and population densities (POP) respectively. Each subgroup consists of eleven transfer errors. The mean TE for each subgroup with eleven transfer errors is calculated.

Chapter 4 reports the results for this study. First, the GIS-based method was conducted on (i) the City's 45 potential sites (Scenario 1) and (ii) the entire island of Oahu (Scenario 2). Second, benefits transfer (BT) methods (e.g., meta-analysis and mean transfer value approaches) were utilized to measure social costs for the selected sites and to rank these sites. Meta-analysis models were evaluated in terms of sensitivity, validity, and reliability criteria.

[55] Although transfer error (TE) is not a statistical hypothesis test, it is widely used for analyzing the similarity of transfer values and original values (Loomis 1992; Rosenberger and Loomis 2000b; Rosenberger and Loomis 2003; Shrestha and Loomis 2001). In terms of validity or reliability criteria, potential use of the MA-BT approach is suggested (Desvousges et al. 1998; Kirchhoff et al. 1997; Rosenberger and Phipps 2007).

# CHAPTER 4. RESULTS

## 4.1. GIS analysis

The GIS-based method was conducted on (i) the City's 45 potential sites (Scenario 1) and (ii) the entire island of Oahu (Scenario 2). The 7 sites fulfilling constraints (e.g., airports, wetlands, floodplains, land use, groundwater, and landfill capacity) were selected. Benefits transfer (BT) approaches (e.g., meta-analysis (MA) and mean transfer value (MTV) approaches) were then utilized to measure social costs for the selected 7 target. Meta-analysis models were evaluated in terms of sensitivity, validity, and reliability criteria. Sensitivity analysis examined sensitivity of aggregate values estimated by met-analysis models in response to changes in a selected variable (e.g., income, distance from target sites, discount rates, and lengths of landfill life) ceteris paribus. In terms of validity criteria, a parametric t-test and a non-parametric sign rank test were utilized to examine statistical differences between transfer values and original values. In terms of reliability criteria, a transfer error (TE) measured by an absolute percentage difference was utilized to check similarity between transfer values and original values (smaller TE performs better).

### 4.1.1. Scenario 1: City's 45 potential landfill sites

Before the GIS-based analysis, proposed landfill sites are assumed to (i) meet restrictions on fault areas, seismic impact zones, and unstable areas, (ii) place equal weights on exclusionary criteria, and (iii) have a given landfill life.

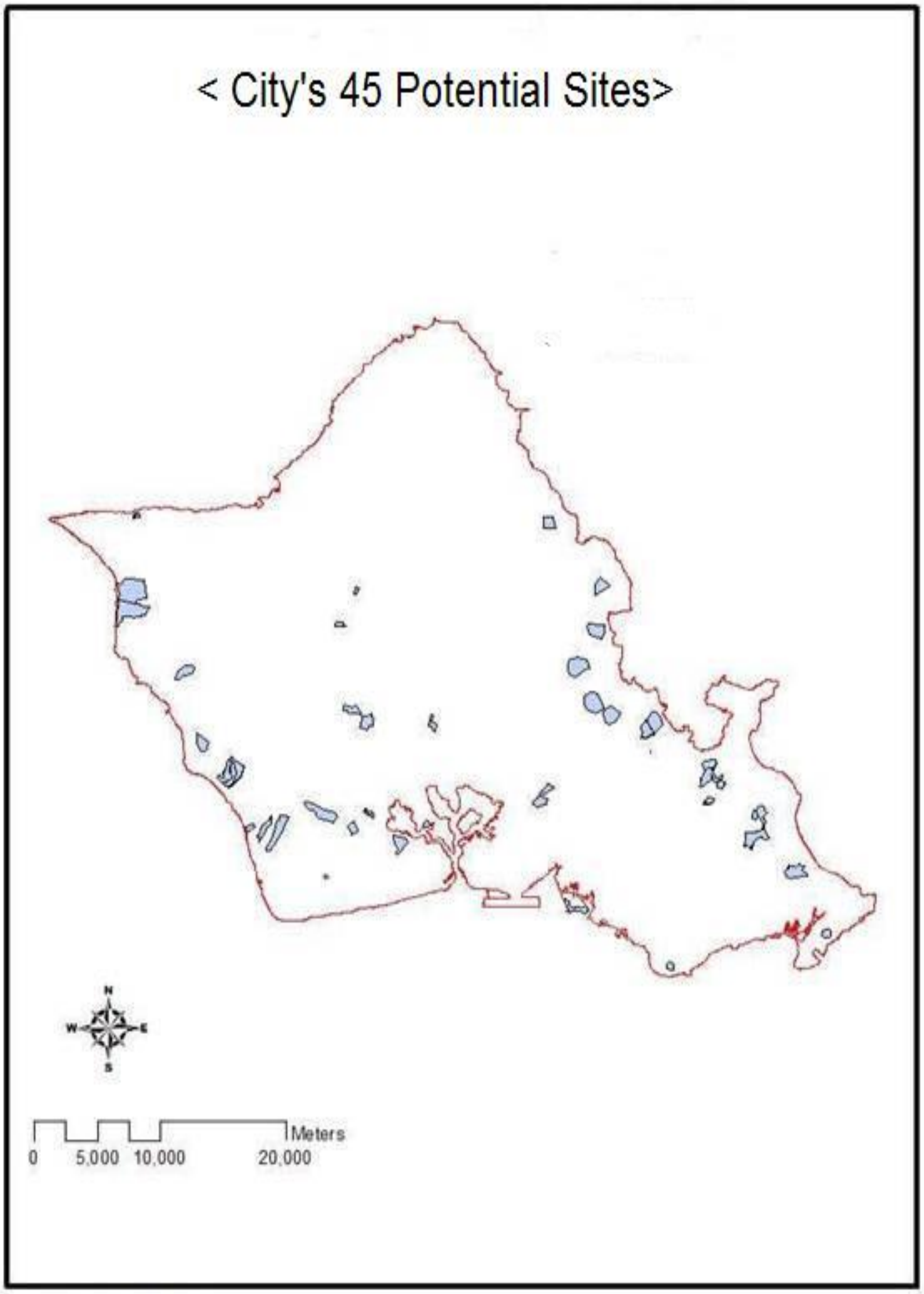
- The island of Oahu is not associated with a particular fault zone and seismic activity (City & County of Honolulu 2006).

- The structure of landfills can endure the effects of ground motion from earthquakes and geological destabilizing events (e.g., sinkholes).
- Restrictive criteria are considered as equally important. Because determining weights is quite controversial and occasionally subjective, placing weights on criteria is not suitable for the preliminary screening phase (for placing weights on criteria, see Siddiqui et al. 1996; Kontos et al. 2003; Sener et al. 2006).
- The City estimated the length of landfill life in years when the approximate site volume (in cubic yards) is divided by the annual projected disposal volume. Because of the complicated process and the length of time to select a new landfill site, potential landfill sites must have at least a 10 year landfill life (City & County of Honolulu 2002, 2003).

The GIS-based preliminary screening (suggested by Daneshvar et al. 2003) was implemented on the City's 45 potential sites as shown in Figure 4.1 (for the process and data description, see pp 74-79 in Chapter 3). Figure 4.2 shows the constraint map of U.S. EPA's restrictions including (i) 10,000 feet (or 3030 meter) buffer areas around airports or airfields, (ii) wetlands, and (iii) 100-year floodplains. This constraint map eliminates 9 sites and reduces the number of potential landfill sites from 45 to 36.

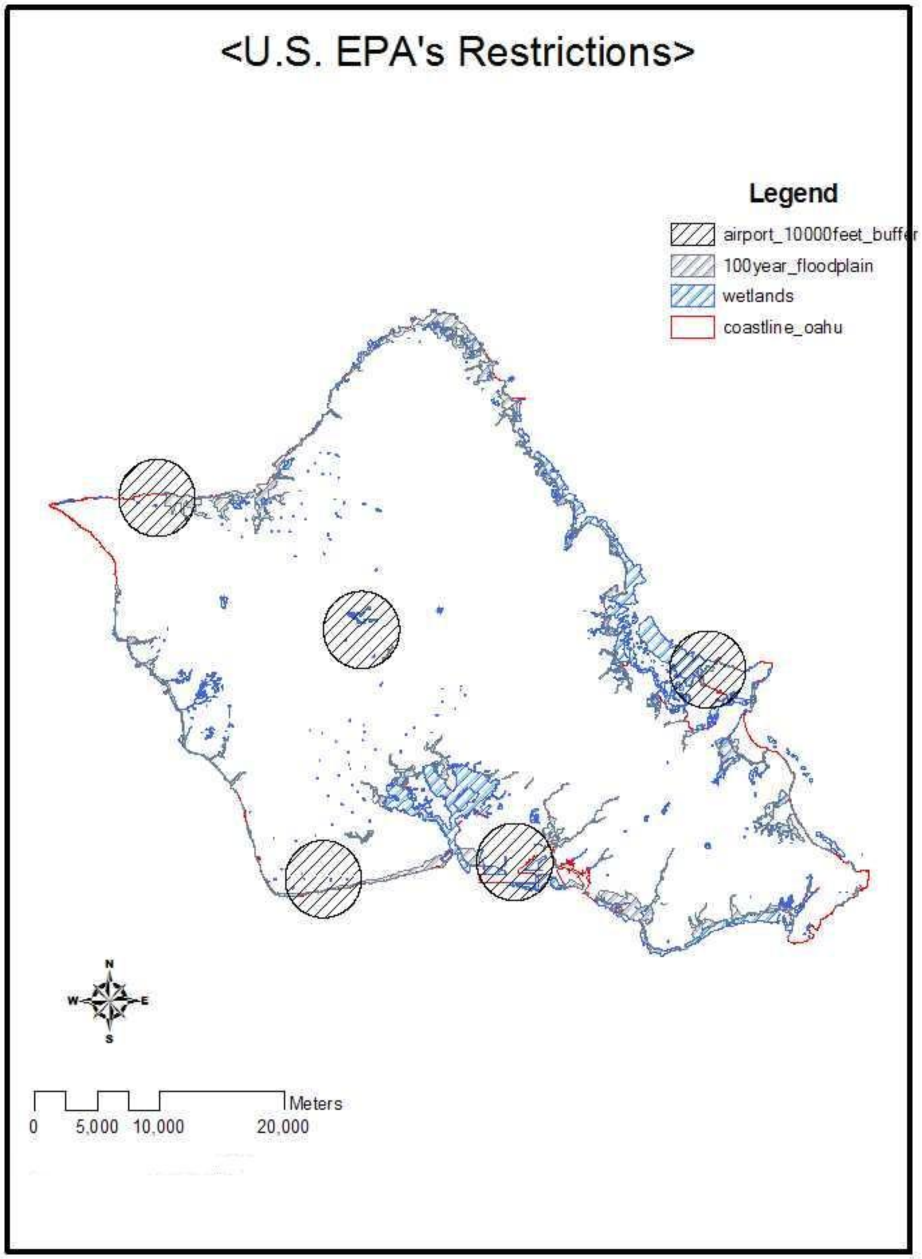
Figure 4.3 shows the constraint map of land use including (i) urban or built-up land (e.g., residential and commercial areas), (ii) critical habitats including threatened and endangered (T&E) plants and Elepaio, and (iii) reserves (e.g., state forests, national parks and historic sites, state parks, and recreation areas). The constraint map of urban land reduces the number of sites from 36 to 24.

Figure 4.4 shows the constraint map of groundwater incorporating (i) the Underground Injection Control (UIC) line and (ii) the No Pass Zone. In order to protect groundwater resources, these two restrictive criteria reduce the number of potential sites from 24 to 10.




<UTM Zone 4N, NAD83>

Figure 4.1. City’s 45 Potential landfill sites




<UTM Zone 4N, NAD83>

Figure 4.2. U.S. EPA’s constraint map

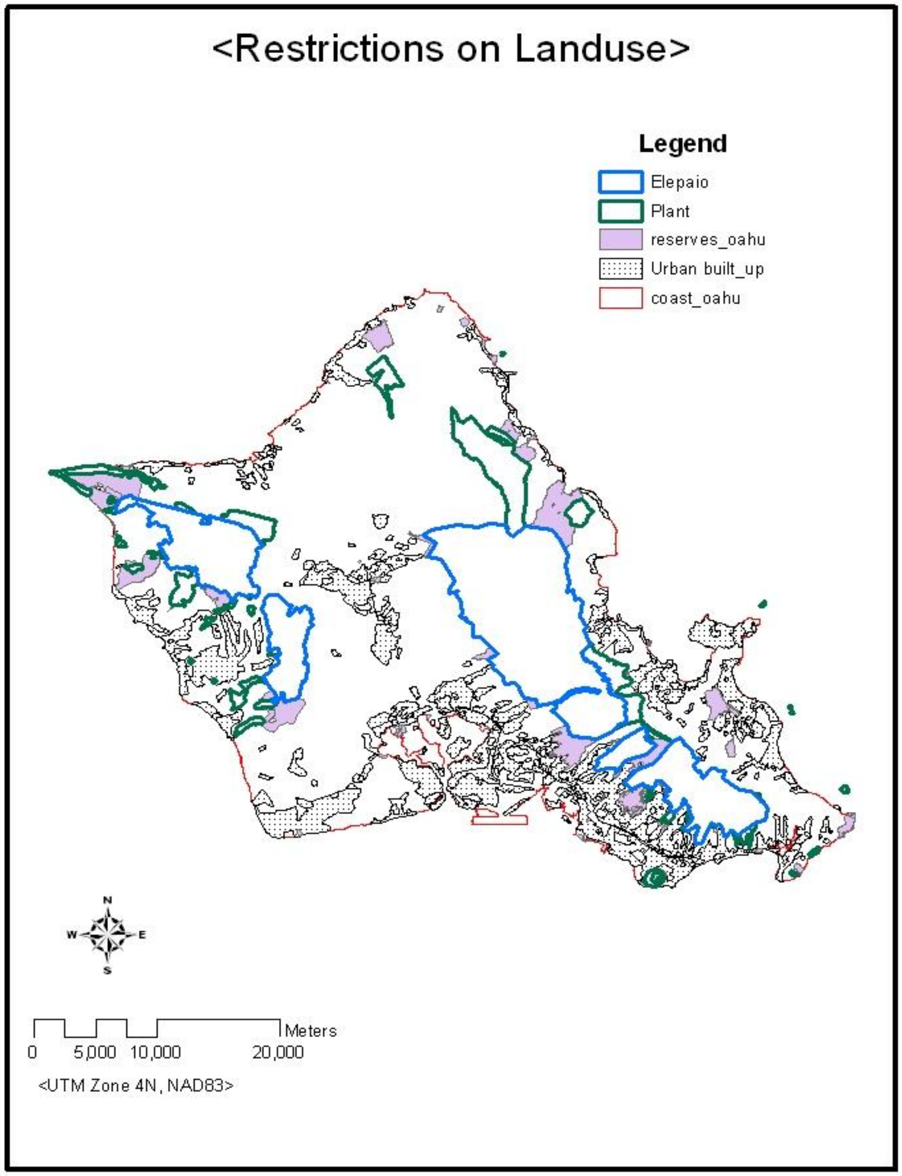



Figure 4.3. The constraint map of land use

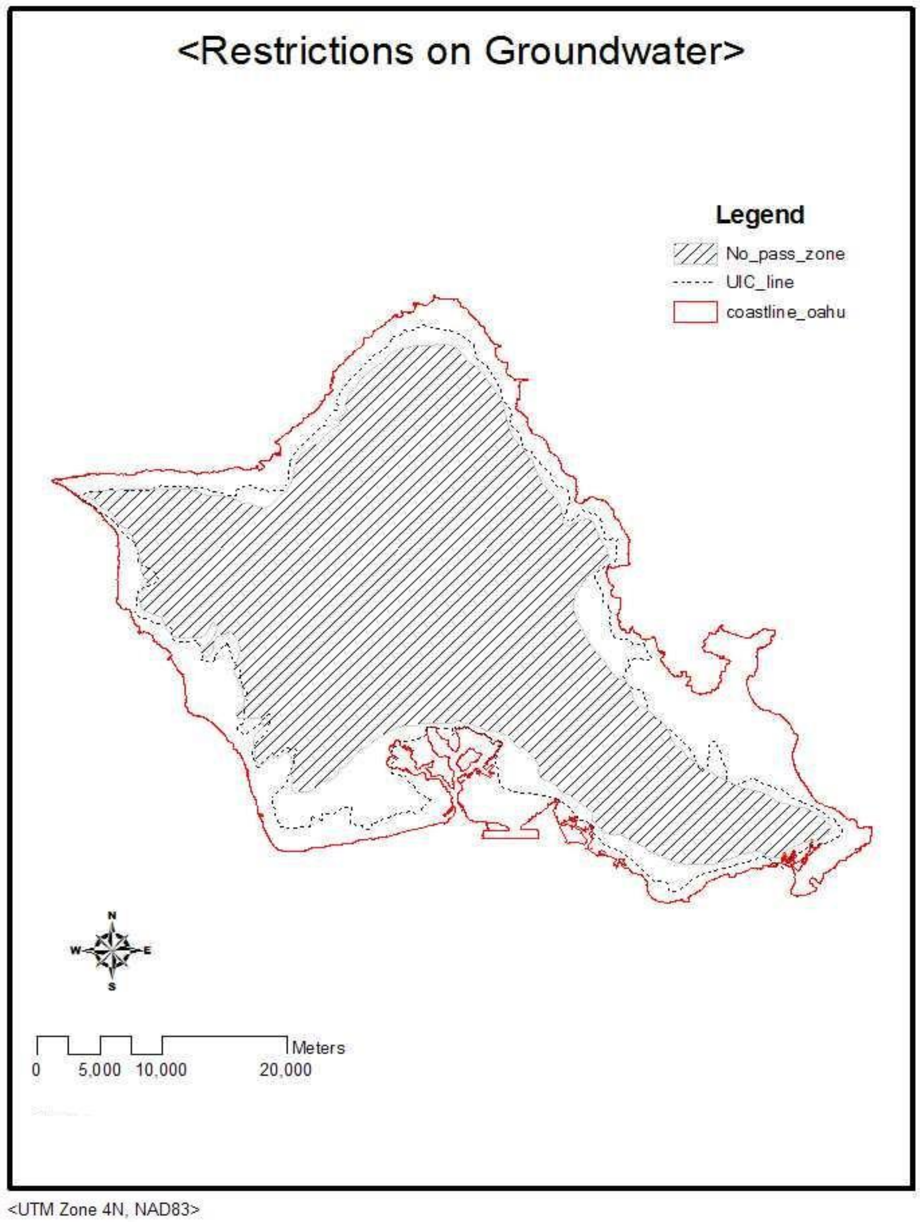



Figure 4.4. The constraint map of groundwater

Landfill capacity (or the length of landfill life in year) of each potential landfill site was estimated by the City (for details, refer to Appendix H in the City & County of Honolulu 2002). The City estimated the site volume for each site and utilized the annual projected waste volume disposed at the existing Waimanalo Gulch Sanitary Landfill and the amount of cover material needed (a 2008 base year). The number of years for capacity available for each site was estimated by dividing the site volume with the annual projected disposal volume.[56] Following the City's restriction on landfill capacity (at least a 10 year landfill life), six sites were eliminated (from 10 to 4). This is the process by which the City's 4 sites were selected.

Soil data was utilized in order to analyze geographic characteristics (e.g., soil properties). Table 4.1 shows soil types and the availability of cover materials. Soil types on the four selected sites do not indicate serious concerns about geological stability such as fractured bedrock and karst topography. Cover materials are available in all selected sites. The availability of cover materials (e.g., silt and clay soils) can reduce operating costs of proposed landfills.

Site A: Papaa clay (PYF) is the dominant soil type on Site A. The surface layers consist of clays. Below these clays is silty clay loam. Kawaihapai silty clay loam (KlbC) and Alaeloa silty clay (ALF) cover small parts of the site. Cover material is also available.

Site B: Lualualei extremely stony clay (LPE) and stony steep land (rSY) are major soil types for Site B. Lualualei extremely stony clay (LPE) is moderately sloping to steep, and the surface consists of stony clay. Stony steep land (rSY) consists of a mass of boulders and stones, and a small amount of soil provides a foothold for plants. Cover material is also available.

---

[56] The capacity or volume of the existing WGSL landfill was estimated by analyzing site characteristics, slope, and area available for development. The capacity of each target site except for the existing site was approximated by utilizing site area multiplied by height (assuming a 4:1 ratio of waste to daily cover). The projected waste volume was 600,000 cubic yards (or 459,000 cubic meters) per year.

Site C: Soil types on Site C consist of Mamala stony silty clay loam (MnC), Lualualei extremely stony clay (LPE), and stony steep land (rSY). MnC has flat areas. In most places, the slope does not exceed 6%. LPE is moderately sloping to steep, and the surface consists of stony clay. Stony steep land (rSY) consists of a mass of boulders and stones, and a small amount of soil provides a foothold for plants. Cover material is also available.

Site D: Rock land (rRK) and stony steep land (rSY) are major soil types for Site D. For rRK, exposed rock covers 25-90% of the surface and includes very shallow soils. Stony steep land (rSY) consists of a mass of boulders and stones, and a small amount of soil provides a foothold for plants. Cover material is also available.

Table 4.1. Soil Types and Availability of Cover Material at Selected Sites

| Sites (p) | Soil Classification | Cover Material |
|---|---|---|
| Site A | Papaa clay, 35-70% slopes (PYF) | Available on site |
| | Kawaihapai silty clay loam, 0-15% slopes (KIbC) | |
| | Alaeloa silty clay, 40-70% slopes (ALF) | |
| Site B | Stony steep land (rSY) | Available on site |
| | Mahana-Badland complex (MBL) | |
| | Lualualei extremely stony clay, 3 to 35% slopes (LPE) | |
| | Helemano silty clay, 40-90% slopes (MBL) | |
| Site C | Mamala stony silty clay loam, 0-12% slopes (MnC) | Available on site |
| | Lualualei extremely stony clay, 3-35% (LPE) | |
| | Rock land (rRK) | |
| Site D | Rock land (rRK) | Available on site |
| | Stony steep land (rsy) | |
| | Lualualei extremely stony clay, 3-35% slopes (LPE) | |

Source: City and County of Honolulu (2002, 2006)

The GIS-based analysis and the restriction on landfill capacity selected 4 sites (called Sites A, B, C, and D) out of the 45 potential sites. These 4 sites were also part of the 8 sites that the City and County of Honolulu selected after preliminary screening. This can be logically explained since this research used the City's 45 potential sites with the same restrictions. However, findings from the GIS analysis differ from those of the City. First, the application of restrictive criteria has different results. The GIS analysis (Scenario 1) has a smaller number of sites (4 sites) than the City (8 sites). The number of sites can be different between the two approaches when exclusionary criteria are applied. Table 4.2 illustrates the differences in attrition for the preliminary screening phase between the GIS analysis and the City's approach. Table 4.3 exhibits one example of using U.S. EPA's restrictions (e.g., airports, floodplains, and wetlands) in the preliminary screening process that shows differences in the number of sites eliminated from these two approaches.[57]

Second, the application of restrictions on groundwater and land use is different between the two approaches. While the Board of Water Supply (BWS) reviewed possible sites for protecting groundwater, this study created constraint maps (GIS data) for the No Pass Zone and the Underground Injection Control (UIC) line. Additionally, while the City removed potential sites in developed areas (e.g., areas with buildings or residential houses) and closed landfill sites as restrictions on land use, the GIS analysis utilized constraint maps based on digital data incorporating build-up land, critical habitats, and reserves.

Table 4.2. Comparing Attrition of Sites between GIS Analysis and the City's Approach

| | C&C of Honolulu | | GIS Analysis | |
|---|---|---|---|---|
| Restrictive Criteria | Before application of criteria | After application of criteria | Before application of criteria | After application of criteria |
| US EPA | 45 | 40 (-5) | 45 | 36 (-9) |
| (i) Airports | | (-4) | | (-4) |
| (ii) Floodplains | | (0) | | (-1) |
| (iii) Wetlands | | (-1) | | (-4) |
| Land use | 40 | 34 (-6) | 36 | 24 (-12) |
| Groundwater | 34 | 16 (-18) | 24 | 10 (-14) |
| Landfill Capacity | 16 | **8 (-8)** | 10 | **4 (-6)** |

Source: Report of Mayor's Advisory Committee on Landfill Selection (2003)
1) Parenthesis ( ) indicates the number of attrition of sites.

[57] It should be noted that data quality problems (e.g., digitization and map scale to provide information) may occur in the GIS analysis. For ensuring improved accuracy, field validation and corroboration from other research results are recommended.

Table 4.3. Comparing the Application of U.S. EPA's Restrictive Criteria between the GIS Analysis and the City's Approach

| | Airports | Floodplains | Wetlands |
|---|---|---|---|
| Site 1 | O X | | |
| Site 2 | | | O X |
| Site 3 | | | O |
| Site 4 | O X | | |
| Site 5 | O | | O |
| Site 6 | O X | | |
| Site 7 | X | O X | |
| Site 8 | | | O |

1) While O indicates the sites eliminated after applying U.S. EPA's restrictions with GIS analysis, X indicates the sites excluded after the application of the City's approach.
2) The names of sites are not revealed because the process of site selection is a sensitive issue.

It is interesting to note that despite different orderings in applying restrictions, the final selected sites did not change. The assumption of equal weights for criteria allows each constraint map to be integrated with other maps in different combinations. Final selected sites are outside of the integration (union) of all the constraint maps (i.e., these sites satisfy all exclusionary criteria). One should be cautious about placing different weights on criteria since different ordering of restrictions can lead to different results (see Luthbom and Lagerkvist 2003 which show that weighting and/or using different criteria can produce different results). Thus, the assumption of equal weights for restrictive criteria is recommended only for preliminary screening.

### 4.1.2. Scenario 2: Entire island of Oahu

Rather than the City's 45 potential landfill sites, the GIS-based analysis was applied to the entire island of Oahu. Some assumptions used for finding potential landfill sites:

- Soil properties. Potential landfill sites meet soil properties in regard to flooding, ponding, slope, and depth to permafrost, a water table, and bedrock or a cemented pan (see Table 4.4).[58]

[58] The Soil Data Viewer is utilized for creating thematic soil maps showing what map units are suitable or unsuitable for a sanitary landfill. Each soil map unit may contain multiple soil

- Landfill site size. Target sites should have at least 90 acres (0.36$\mathrm{km}^2$) of land available in order to verify minimum life (in years) based on the extended areas of the current Waimanalo Gulch Sanitary Landfill (WGSL) site. The length of landfill life in years is later estimated for further economic analysis by utilizing the City's approach.
- The number of parcels. Each area should not exceed 3 parcels based on a unique Tax Map Key (TMK) number (City and County of Honolulu 2003). From the TMK one can obtain information about parcels and site areas (maximum 3 parcels).

Table 4.4 shows constraints on soil properties.[59] Flooding can contaminate areas downstream from the landfill. If permeability is too rapid or if on fractured bedrock or if on a fractured cemented pan, or if the water table is close to the surface, leachate can contaminate the water supply. Ponding limits the installation and function of most land use applications. Slopes over 15% can also cause leachate to flow along the surface of the soils in steeper areas and cause difficult permeability problems. Figure 4.5 describes the soils constraint map. The very limited area is considered as unsuitable for landfill selection.

Table 4.4. Constraints of Soil Properties

| Factors | Constraints |
|---|---|
| Flooding | frequency |
| Ponding | frequency |
| Slope | >15% |
| Seepage (Permeability) | > 14.0 micrometers/second |
| Depth to permafrost (permanently frozen soil layer) | < 50 cm |
| Depth to bedrock | <100 cm |
| Depth to cemented pan | <100 cm |
| Depth to saturated zone | <180 cm |

components, and the Soil Data Viewer makes it easy to compute a single value for a map unit and display results.

[59] The Natural Resources Conservation Service (NRCS) categorizes soils into the following interpretive ratings: (i) not limiting (rating index = 0), (ii) somewhat limiting (0 < rating index < 1), and (iii) very limiting (rating index = 1). The very limited rating is considered as exclusionary criteria for landfill selection. Because these criteria for soil properties are too strict to determine potential sites, this study considers sites with rating indexes < 2/3.

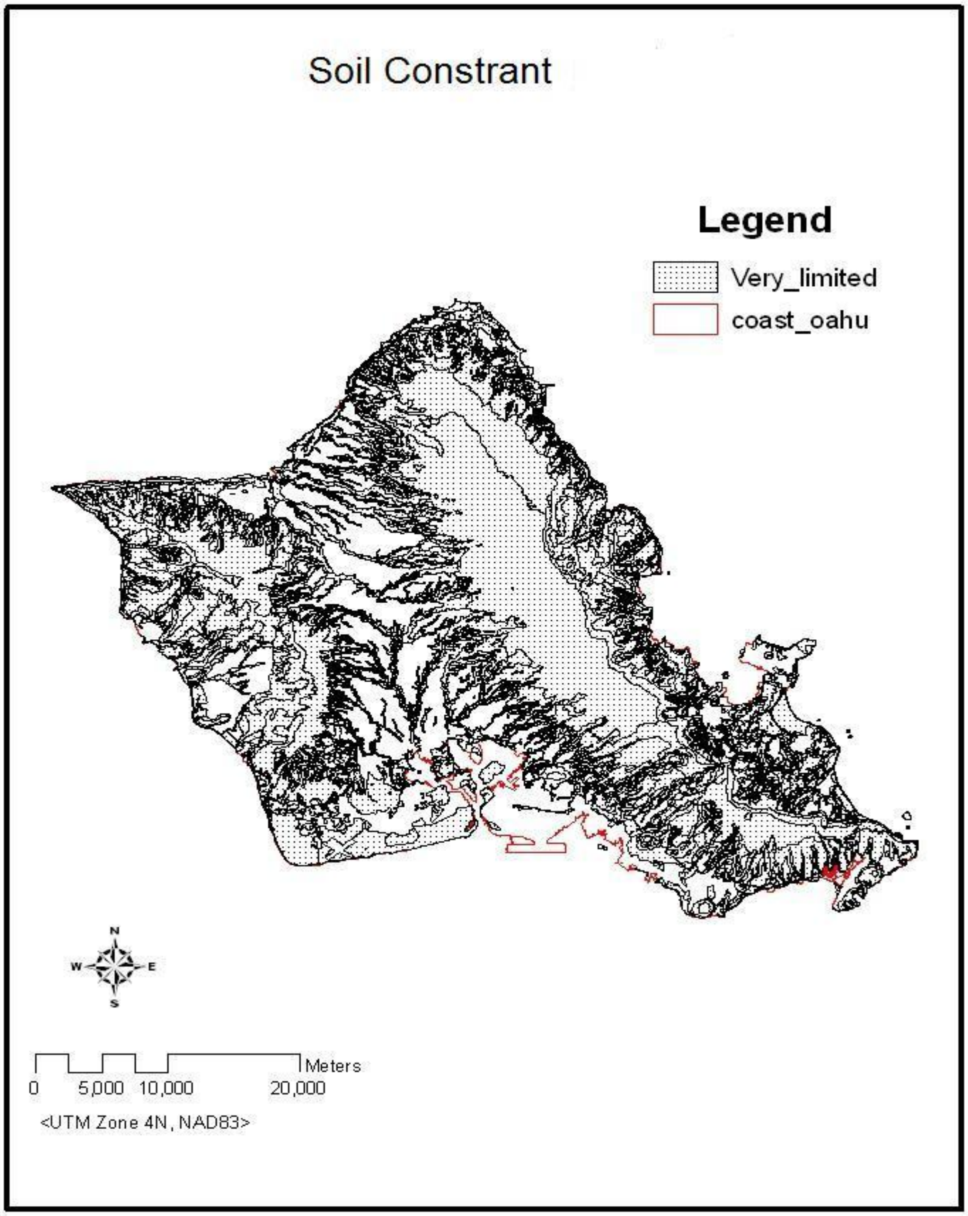



Figure 4.5. The soils constraint map

Ensuring sufficient landfill area is important for landfill selection (Tchobanoglous et al. 1993). Because calculating landfill capacity or volume requires analysis of site characteristics, slope, and area available for development, some landfill site selection studies e.g., Baban and Flannagan 1998; Siddiqui et al. 1996; Kao et al. 1996; Kontos et al. 2003, 2005; and Vasiloglou 2004 assumed that the landfill capacity was given or not needed. Although little evidence is known about the relationship between landfill life and landfill site area, some studies (Baban and Flannagan 1998; Siddiqui et al. 1996; Guam EPA 2004; Santa Cruz County 2003) utilized the site area in order to verify minimum life (see Table 4.5). This study assumes that each site requires at least 90 acres (0.36$km^2$) of land area based on the extended area of the existing Waimanalo Landfill. Following the City's approach, landfill life in years for each site, determined from a GIS analysis, is later estimated by utilizing the following factors: (i) the annual volume of municipal solid waste (MSW) disposed at the existing Waimanalo Gulch Sanitary Landfill (WGSL) site; (ii) landfill site volume (multiplying elevation height by site area); and (iii) length of landfill life in years when (ii) is divided by (i) (for technical details, refer to Tchobanoglous et al. 1993).[60]

Table 4.5. Landfill Capacity (Life in years) and Site Area

| | Place | Landfill Life (years) | Area (acres) |
|---|---|---|---|
| This study | Oahu County, HI | 10 | 90 (0.36$km^2$) |
| Siddiqui et al. 1996 | Cleveland County, Oklahoma | 20 | 200 (0.8$km^2$) |
| Kontos et al. 2003 | Lesvos, Greece | * | 50 (0.2 $km^2$) |
| Baban and Flannagan 1998 | Warwickshire, United Kingdom | * | 125 (0.5 $km^2$) |
| Guam EPA. 2004 | National (USA) | 50 | 500 to 1000 (2 $km^2$ to 4 $km^2$) |
| | Guam | 30 | 100 (0.4$km^2$ ) |
| Santa Cruz County 2003 | Santa Cruz, CA | 20 | 100 (0.4$km^2$) |

* The research did not specify this information.

[60] Site area and height for each target site are sourced from Oahu parcels and topographic layers. A ratio of waste to daily cover (4:1) is assumed (Bagchi 1990; City and County of Honolulu 2002).

A preliminary list of 38 potential sites shown in Figure 4.6 is based on these three assumptions. The 34 sites are different from those of the City. The GIS-based screening with the same exclusionary criteria used in Scenario 1 was conducted on these 38 sites. The U.S. EPA's constraint map shown in Figure 4.2 eliminates 15 sites and reduces the number of potential landfill sites from 38 to 23. The land use constraint map shown in Figure 4.3 reduces the number of sites from 23 to 17. The groundwater constraint map in Figure 4.4 reduces the number of potential sites from 17 to 4. For further economic analysis, an landfill life year for each target site (remaining 4 sites) was estimated following the City's method. Each site satisfies a minimum 10 year of landfill capacity (for landfill life in years for target sites, see Table 3.6 in Chapter 3).

Of the 4 remaining sites, one site (Site C) is in the same location as in Scenario 1, and the other three sites (Sites E, F, and G) are in different locations. This can be explained by the application of GIS analysis to the entire island of Oahu (E, F, and G are not part of the City's original 45 sites).[61]

Together, both scenarios find a total of 7 target sites that satisfy criteria (Sites A to D from Scenario 1 and Sites C, and E to G from Scenario 2) shown in Figure 4.7. Benefits transfer methods were utilized to measure social costs for these 7 sites and to rank the selected target sites according to social cost minimization.

---

[61] One concern is that because sites E (about 260 meter) and F (about 30 meter) are close to the Ocean, these two sites have the possibility of leaching. Further technical research is needed to examine the potential impact of leaching (e.g., the underlying bedrock of these sites that protect the waste extracts from leaching).

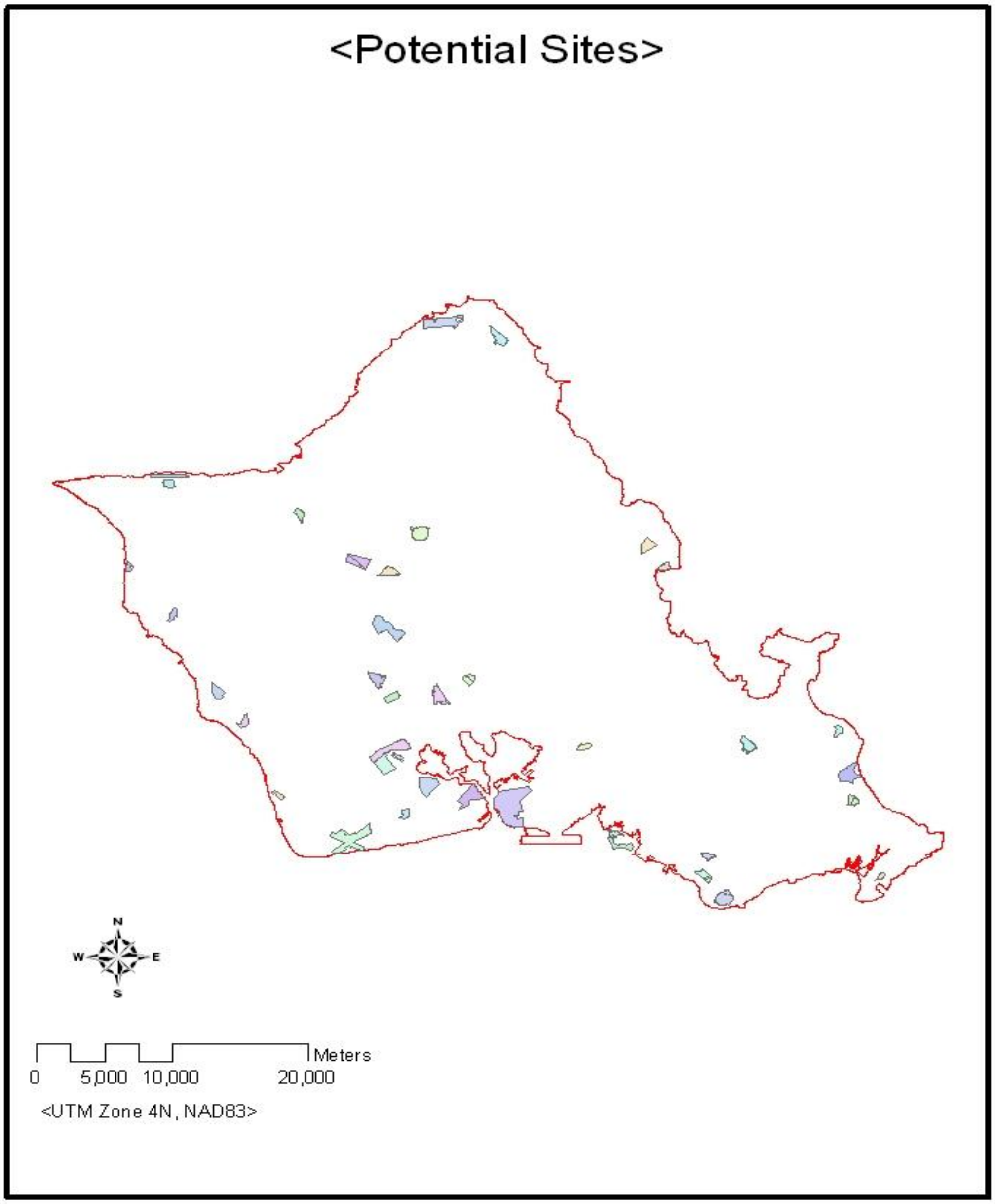



Figure 4.6. A preliminary list of potential landfill sites

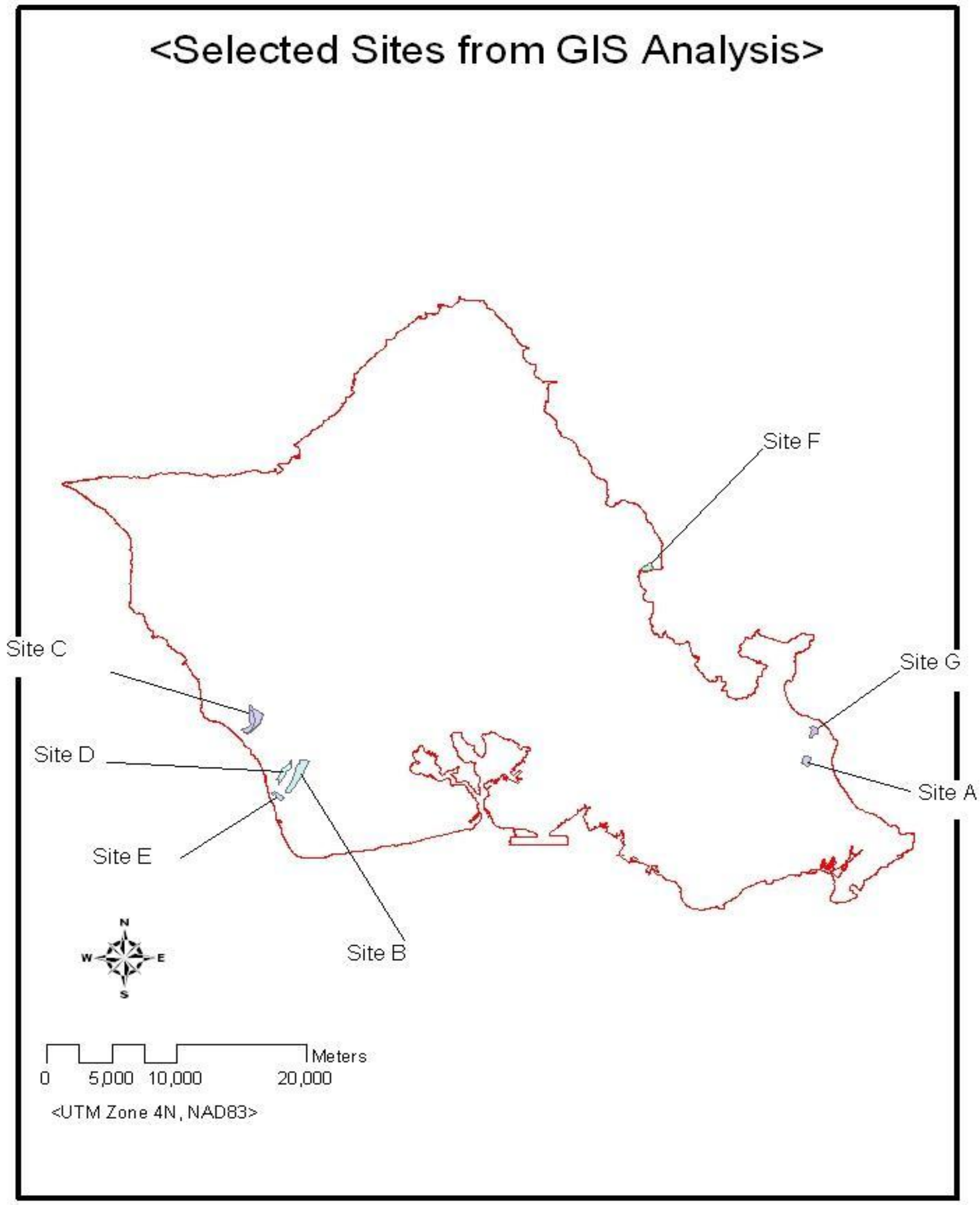



Figure 4.7. Selected sites from GIS analysis

## 4.2. Economic analysis

This section provides results from meta-analysis (MA) and benefits transfer (BT) methods (e.g., meta-analysis for benefits transfer (MA-BT) and mean transfer value (MTV) approaches) used for measuring social costs for the seven sites. MA first was conducted to find suitable functions. The estimated MA functions were then utilized in order to calculate social costs. For a comparison purpose, the mean transfer value approach was also conducted. MA models were evaluated in terms of sensitivity, validity, and reliability criteria.

### 4.2.1. Results of meta-analysis (MA)

Based on earlier tests for panel effects and heteroskedasticity (see pp 89-90), the ordinary least squares (OLS) method was employed because no statistically significant panel effects were found. Since the simple linear form was heteroskedastic, a semi-log form (log transformation of the dependent variable) was utilized to correct for heteroskedasticity. For comparison purposes, robust standard errors were also utilized to correct for heteroskedasticity.[62] Results for a wide range of diagnostic tests (e.g., heteroskedasticity test, normality test, model specification test, and multicollinearity assessment) are reported (for estimation methods, data description, and tests see pp 79-90 in Chapter 3).

For finding a suitable function for benefits transfer (BT), different MA models were specified: (i) different functional forms (e.g., a simple linear form versus a semi-log form) and (ii) a fully specified model including all independent variables versus a restricted model based on statistical significance of independent variables or presence of econometric problems (e.g., multi-collinearity). Variables MSW and ACTIVE were dropped since these variables were collinear or have

[62] While robust standard errors have asymptotic properties (large samples), small sample properties are unknown (Johnston and Dinardo 1997). However, researchers often utilize the robust standard error for a comparison purpose.

statistically insignificant coefficients at the 20% level (Rosenberger and Loomis 2000 a, b, 2003; Shrestha et al. 2007).[63]

Table 4.6 describes results based on a simple linear form. The ordinary least squares (OLS) method using robust standard errors corrects for heteroskedasticity. Regression (1) is a fully specified model including all variables. The joint F tests are significant at the 5% level. Only the coefficient on N is significant at the 10% level. Model specification is accepted at the 5 % level. However, one problem is non-normality of residuals i.e., the null hypothesis of non-normal residuals is rejected at the 1% level (p-value=0.0008). Finding non-normal residuals suggests that p-values, F statistics, and t statistics for OLS are invalid. Another potential problem is multicollinearity. Although VIFs do not show severe multicollinearity, the correlation matrix for explanatory variables indicates possible multicollinearity, especially involving MSW and ACTIVE (see Table 4.7). The correlation coefficients for explanatory variables MSW and POP, MSW and FUNCTION are -0.7899 and -0.8355 respectively. Also, the correlation coefficients for explanatory variables ACTIVE and SE, ACTIVE and FUNCTION, ACTIVE and MSW are 0.7608, -0.919, and 0.7629 respectively.

In regression (2), the model is respecified by omitting MSW and ACTIVE for reducing possible multicollinearity. The signs of coefficients FUNCTION changes from negative to positive. Although VIFs do not detect severe multicollinearity, the correlation matrix for explanatory variables still shows slight multicollinearity (see Table 4.8). The correlation coefficients for explanatory variables SE and Y is 0.8497. The correlation coefficients for FUNCTION and Y, FUNCTION and SE are -0.7564 and -0.7706 respectively. The null hypothesis that residuals are normally distributed (p-value=0.0001) is rejected at the 1 % level. Non-normal

[63] Although most studies use a 1% or 5% significant level, for benefits transfer, retaining significant variables at the 20% level is often recommended since this optimization can reduce transfer error for benefits transfer. However, retaining core economic variables (e.g., Y and POP) at the 20% level avoids misspecification that can occur when relevant core variables are omitted from the model.

residuals persist. Using OLS invalidates test statistics. Overall, OLS using the robust standard errors for the simple linear functional form does not perform well.

Table 4.6. Results of OLS Using Robust Standard Errors

| | Dependent Variables: MWTP | |
|---|---|---|
| | Regression (1) | Regression (2) |
| C (Constant) | -11.2967900<br>(0.269) | -10.0586<br>(0.265) |
| Y | 0.0001023<br>(0.209) | 0.0001336°<br>(0.139) |
| POP | -0.0008198<br>(0.388) | -0.0004680<br>(0.504) |
| N | 0.0005823*<br>(0.090) | 0.0007534**<br>(0.016) |
| SE | 2.5248770<br>(0.273) | 3.0036810<br>(0.215) |
| FUNCTION | -0.5980758<br>(0.868) | 1.4768370<br>(0.348) |
| MSW | 2.9111430<br>(0.356) | Omitted |
| ACTIVE | 4.0990610<br>(0.423) | Omitted |
| Observations | 22 | 22 |
| R square | 0.4120 | 0.3702 |
| F test (p value) | 0.0120 | 0.0095 |
| Normality (p value) | 0.0008 | 0.0001 |
| Heteroskedasticity | R.E | R.E |
| MULTICOLLINEARITY (VIF) | MSW (2.29), POP (2.21)<br>ACTIVE (2.1), FUNCTION (2.05)<br>N (1.98), SE (1.67),<br>Y (1.54) | POP (1.97), N (1.72)<br>Y(1.34), FUNCTION (1.20)<br>SE (1.13) |
| Ramsey RESET test (P value) | 0.3836 | 0.4480 |

* The 1, 5, 10, and 20 % statistically significance levels are respectively shown as ***, **, *, and °. All values in parentheses () under the estimated coefficients are p-value.
**Robust standard errors (RE) correct for heteroskedasticity.

Table 4.7. Correlation Matrix based on Linear Model (Regression 1)

| | Y | POP | N | SE | FUNCTION | MSW | ACTIVE |
|---|---|---|---|---|---|---|---|
| Y | 1 | | | | | | |
| POP | 0.1482 | 1 | | | | | |
| N | -0.2327 | -0.2410 | 1 | | | | |
| SE | 0.4785 | 0.0339 | -0.6408 | 1 | | | |
| FUNCTION | -0.3849 | 0.5352 | 0.4664 | -0.7180 | 1 | | |
| MSW | 0.1989 | -0.7899 | -0.1954 | 0.3100 | -0.8355 | 1 | |
| ACTIVE | 0.2143 | -0.4841 | -0.5285 | 0.7608 | -0.9190 | 0.7629 | 1 |

Table 4.8. Correlation Matrix based on Linear Model (Regression 2)

| | Y | POP | N | SE | FUNCTION |
|---|---|---|---|---|---|
| Y | 1 | | | | |
| POP | 0.3258 | 1 | | | |
| N | -0.0897 | -0.6300 | 1 | | |
| SE | 0.8497 | 0.1710 | -0.0991 | 1 | |
| FUNCTION | -0.7564 | -0.1206 | 0.0901 | -0.7706 | 1 |

Table 4.9 shows OLS results (equations (3) and (4)) of the semi-log functional form and robust standard errors regressions (3a) and (4a). Regression (3) is a fully specified model with all variables. Residuals are normally distributed at the 5 % level (p-value=0.9820). VIF and correlation matrix for explanatory variables do not show a severe degree of multicollinearity (for correlation matrix, see Table 4.10). Model specification (p-value=0.5383) is not rejected at the 5 % level. The BP-CW test for heteroskedasticity fails to reject the null hypothesis at the 5 % level. The explanatory power ($R^2$) of the regression is 0.5208. Unlike the simple linear form, the semi-log form is homoskedastic, and the residuals are normally distributed. However, the joint F test (p-value=0.1024) is insignificant at the 10 % level (i.e., overall insignificance of the regression model). The coefficient on Y is significant at the 5% level with a positive sign indicating that higher income levels are related to higher MWTP (see Smith and Huang 1995; Nelson et al. 1998). Other variables are statistically insignificant at the 10% level. The coefficients on N and SE are significant at the 20% level, but other variables (MSW, ACTIVE, and FUNCTION) are statistically insignificant at the 20% level.

Regression (3a), using robust standard errors, performs well for diagnostic tests. Joint and individual significances are improved. The joint F test (p-value = 0.0149) now becomes significant at the 5% level, and the coefficients on N and Y are significant at the 5% or 10% levels respectively. The coefficient on SE is significant at the 20% level, but the coefficients on MSW, ACTIVE, and FUNCTION are insignificant at the 20% level respectively.

Table 4.9. Results from OLS and Robust Consistent Standard Errors (Semi-Log Form)

| | Dependent variable: Log (MWTP) | | | |
|---|---|---|---|---|
| Independent Variables | Regression (3) | Robust Standard Error (3a) | Regression (4) | Robust Standard Error (4a) |
| C (constant) | -3.011537* | ° | -2.81639** | * |
| | (0.053) | (0.102) | (0.036) | (0.091) |
| Y | 0.0000454** | * | 0.0000455** | ** |
| | (0.030) | (0.063) | (0.014) | (0.032) |
| POP | 0.0000143 | | 0.0000306 | |
| | (0.932) | (0.933) | (0.836) | (0.821) |
| N | 0.0001193° | ** | 0.00013016* | ** |
| | (0.167) | (0.049) | (0.087) | (0.023) |
| SE | 0.3567514° | ° | 0.4041681** | * |
| | (0.125) | (0.163) | (0.030) | (0.100) |
| FUNCTION | 0.4325731 | | 0.5742518° | ** |
| | (0.463) | (0.346) | (0.182) | (0.045) |
| MSW | 0.2992723 | | Omitted | Omitted |
| | (0.711) | (0.548) | | |
| ACTIVE | 0.1702279 | | Omitted | Omitted |
| | (0.789) | (0.807) | | |
| Observations | 22 | | 22 | |
| R square | 0.5208 | | 0.5157 | |
| F test (p-value) | 0.1024 | 0.0149 | 0.0276 | 0.0037 |
| Normality (p-value) | 0.9820 | | 0.9706 | |
| Heteroskedasticity | 0.8424 | RE | 0.8609 | RE |
| Multicollinearity (VIF) | MSW (2.29), POP (2.21), ACTIVE (2.10), Function (2.05), N (1.98), SE (1.67), Y (1.54) | | POP (1.97), N (1.72), Y (1.34), Function (1.20), SE (1.13) | |
| Ramsey RESET test (p-value) | 0.5383 | | 0.4791 | |

* The 1, 5, 10, and 20 % statistically significance levels are respectively shown as ***, **, *, and °. All values in parentheses () under the estimated coefficients are p-value.
** Robust standard error methods were utilized for Regression (3a) and (4a).

In regression (4), the model is respecified by omitting statistically insignificant coefficients on MSW and ACTIVE at the 20% level in regression (3). For optimizing meta-analysis for transfer function, Rosenberger and Loomis (2001a and 2003) suggest drop statistically insignificant variables at the 20% level. Although POP is insignificant at the 20% level, it is a core economic variable and should be retained in order to avoid misspecification that can occur when relevant core variables are omitted from the model. Residuals are normally distributed at the 5 % level (p-value=0.9706). The F statistic (p-value=0.0276) rejects the null hypothesis at the 5 % level. The VIF and the correlation matrix do not detect severe multicollinearity (for the correlation matrix, see Table 4.11). The BP-CW test for heteroskedasticity (p value=0.8609) fails to reject the null hypothesis for constant variance at the 5 % level. Model specification (p-value=0.4791) is accepted at the 5 % level. The explanatory power ($R^2$) of the model is 0.5157, which is similar to regression (3) ($R^2$=0.5208). The coefficients on Y and SE are significant at the 5% level, and the coefficient on N is significant at the 10% level. The coefficient on FUNCTION is significant at the 20% level.

In regression (4a), the robust standard errors method was utilized. Diagnostic tests perform well. Individual significance is slightly improved. Other variables except POP are significant at the 5% or 10% level.

Table 4.10. Correlation Matrix based on Semi-Log Model (Regression 3)

| | Y | POP | N | SE | FUNCTION | MSW | ACTIVE |
|---|---|---|---|---|---|---|---|
| Y | 1 | | | | | | |
| POP | 0.4530 | 1 | | | | | |
| N | -0.0903 | -0.4171 | 1 | | | | |
| SE | 0.1429 | 0.2296 | 0.2672 | 1 | | | |
| FUNCTION | -0.1337 | -0.0332 | 0.4588 | 0.3874 | 1 | | |
| MSW | 0.0741 | -0.2012 | -0.3227 | -0.5695 | -0.5887 | 1 | |
| ACTIVE | -0.2687 | -0.3264 | -0.3076 | -0.3057 | -0.5304 | 0.5102 | 1 |

Table 4.11. Correlation Matrix based on Semi-Log Model (Regression 4)

| | Y | POP | N | SE | FUNCTION |
|---|---|---|---|---|---|
| Y | 1 | | | | |
| POP | 0.4266 | 1 | | | |
| N | -0.1448 | -0.5969 | 1 | | |
| SE | 0.2337 | 0.1423 | 0.1051 | 1 | |
| FUNCTION | -0.2609 | -0.3070 | 0.3156 | 0.0756 | 1 |

The results in Table (4-9) confirm previous findings. First, Y (income levels) has a statistically significant and positive effect on MWTP at the 5% level i.e., the higher income levels are related to the higher MWTP (Nelson et al. 1997 and Walton et al. 1996). Income is a core economic variable reflecting differences between this study and/or target sites. High-income households are willing to pay more to avoid landfills than low-income households are. When a landfill is selected, its presence decreases housing values. High-income households will escape it, and low-income households will flock to acquire cheap land and housing. In terms of environmental equity, compensation for impacted households and aid to reduce transaction costs for low-income households (e.g., providing legal service and information) can improve inequity (Banzhaf 2008).

Second, some study design variables (e.g., N, SE, and FUNCTION) have statistically significant effects on MWTP at the 5% or 10 % level (Walton et al. 2006; Smith and Huang 1995), which adjust for different MWTP. Although POP is insignificant at the 10% level, the positive sign is consistent with a priori expectation i.e, higher population densities are related to the higher MWTP (see Brander et al. 2006).

The two meta-analysis (MA) models 1 (regression 3) and 2 (regression 4) with reasonable diagnostic test results were utilized to measure social costs for the selected 7 target sites on Oahu. Meta-analysis models are evaluated in terms of sensitivity, reliability, and validity criteria.

### 4.2.2. Benefits transfer (BT)

#### The mean transfer value approach

A mean transfer value approach based on the contingent valuation method (CVM) research was utilized to measure social costs for a comparison purpose with those of meta-analysis (see pp 93-96 in Chapter 3 for more information on this approach and data used). Present values of aggregate annual WTP and aggregate annual values were adjusted for: (i) per capita income and (ii) median household income for both cases of e=0.3 and e=1.

Tables 4.12 and 4.13 show present values of aggregate annual WTP estimates and 95% confidence intervals with e=0.3 and e=1, which represent a simple statistical range in which true values would fall with 95%. Despite the variability in estimates, the rank order is consistent between them. One should be cautious that present values for target sites or projects with unequal lengths of landfill life cannot be compared directly. Annual aggregate values measured by an equivalent annual value method are recommended for target sites with unequal lengths of life.

Table 4.12. Present Values of Aggregate Annual WTP: e=0.3*

| Target Sites | Aggregate WTP (i) | Aggregate WTP (ii) |
|---|---|---|
| Site A | 112,570,166 (6) | 125,654,800 (6) |
| | [81,749,632 to 143,193,710] | [91,251,830 to 159,837,883] |
| Site B | 86,132,725 (5) | 104,517,006 (5) |
| | [62,550,486 to 109,564,238] | [75,901,343 to 132,949,772] |
| Site C | 45,659,674 (2) | 56,668,176 (2) |
| | [33,158,533 to 58,080,914] | [41,153,022 to 72,084,165] |
| Site D | 57,346,299 (4) | 69,533,549 (4) |
| | [41,645,482 to 72,946,764] | [50,495,991 to 88,449,429] |
| Site E | 53,226,394 (3) | 64,069,812 (3) |
| | [38,653,564 to 67,706,081] | [46,528,168 to 81,499,339] |
| Site F | 27,210,082 (1) | 31,273,236 (1) |
| | [19,760,246 to 34,612,302] | [22,710,951 to 39,780,795] |
| Site G | 151,302,716 (7) | 170,675,058 (7) |
| | [109,877,615 to 192,463,048] | [ 123,946,012 to 217,105,434] |

* The Honolulu Consumer Price Index (CPI) inflates money values to 2008 US dollars. The 3% discount rate was used for discounting.
** All values in the bracket [ ] indicate 95% confidence level, and all values in the parenthesis ( ) indicate the ranked order.
*** Aggregate WTP (i) is adjusted for per capita income levels, and aggregate WTP (ii) is adjusted for median household income levels.

Table 4.13. Present Values of Aggregate Annual WTP: e=1*

| Target Sites | Aggregate WTP (i) | Aggregate WTP (ii) |
|---|---|---|
| Site A | 137,289,692 (6) | 198,072,352 (6) |
| | [99,701,210 to 174,637,926] | [143,842,213 to 251,955,877] |
| Site B | 73,868,874 (5) | 140,774,520 (5) |
| | [53,644,349 to 93,964,134] | [102,231,929 to 179,070,765] |
| Site C | 34,518,106 (2) | 70,914,502 (2) |
| | [25,067,410 to 43,908,399] | [51,498,853 to 90,206,056] |
| Site D | 49,290,625 (4) | 93,697,427 (4) |
| | [35,795,368 to 62,699,628] | [68,044,051 to 119,186,839] |
| Site E | 47,639,268 (3) | 88,386,604 (3) |
| | [34,596,134 to 60,599,036] | [64,187,276 to 112,431,261] |
| Site F | 26,568,172 (1) | 42,251,123 (1) |
| | [19,294,085 to 33,795,768] | [30,683,207 to 53,745,102] |
| Site G | 174,009,454 (7) | 260,006,350 (7) |
| | [126,367,485 to 221,346,917] | [ 188,819,331 to 330,738,376] |

* The Honolulu Consumer Price Index (CPI) inflates money values to 2008 US dollars. The 3% discount rate was used for discounting.
** All values in the bracket [ ] indicate 95% confidence level, and all values in the parenthesis ( ) indicate the ranked order.
*** Aggregate WTP (i) is adjusted for per capita income levels, and aggregate WTP (ii) is adjusted for median household income levels.

Tables 4.14 and 4.15 summarize aggregate annual values and 95% confidence intervals with e=0.3 and e=1. Aggregate annual values vary, and the rank order can also differ due to different adjustment factors (e.g., average per capita income versus median household income), market conditions (e.g., income levels and the number of households), different income elasticities of WTP (e.g., e=0.3 and 1), and different lengths for landfill life. Aggregate annual WTP estimates (ii) with e=1 (column 3 in Table 4.15) reflect differences between the target sites compared to others. One should be cautious since aggregate annual WTP estimates for e=0.3 (in reference to Sites C, D, E, and F) are similar. Sites C, D, and E (the leeward side) are located nearby, which have similar market conditions (e.g., income levels and the number of households). Although Site F (near Waikane) is far away from these sites, this site has similar per capita income levels and the number of households.

Table 4.14. Aggregate Annual WTP for Target Sites: e=0.3*

| Target Sites (p) | Aggregate WTP (i) | Aggregate WTP (ii) |
|---|---|---|
| Site A | 10,584,921 (7) | 11,815,264 (7) |
| | [7,686,881 to 13,464,439] | [8,580,368 to 15,029,483] |
| Site B | 4,946,419 (5) | 6,002,189 (5) |
| | [3,592,141 to 6,292,041] | [4,358,853 to 7,635,022] |
| Site C | 3,635,005 (3) | 4,511,402 (3) |
| | [2,639,779 to 4,623,041] | [3,276,227 to 5,738,682] |
| Site D | 3,854,572 (4) | 4,673,747 (4) |
| | [2,799,231 to 4,903,168] | [3,394,124 to 5,945,191] |
| Site E | 3,577,650 (2) | 4,306,498 (2) |
| | [2,598,127 to 4,550,912] | [3,127,424 to 5,478,036] |
| Site F | 2,940,796 (1) | 3,379,932 (1) |
| | [2,135,637 to 3,740,809 | [2,454,542 to 4,299,407] |
| Site G | 8,688,993 (6) | 9,801,505 (6) |
| | [6,310,038 to 11,052,743] | [ 7,117,956 to 12,467,903] |

* The Honolulu Consumer Price Index (CPI) inflates money values to 2008 US dollars. The 3% discount rate was used for discounting.
** All values in the bracket [ ] indicate 95% confidence level, and all values in the parenthesis ( ) indicate the ranked order.
*** Aggregate WTP (i) is adjusted for per capita income levels, and aggregate WTP (ii) is adjusted for median household income levels.

Table 4.15. Aggregate Annual WTP for Target Sites: e=1*

| Target Sites (p) | Aggregate WTP (i) | Aggregate WTP (ii) |
|---|---|---|
| Site A | 12,909,287 (7) | 18,624,653 (7) |
| | [9,374,859 to 16,421,125] | [13,525,418 to 23,691,296] |
| Site B | 4,242,132 (5) | 8,084,381 (5) |
| | [3,080,681 to 5,396,160] | [5,870,962 to 10,283,653] |
| Site C | 2,748,016 (1) | 5,645,564 (2) |
| | [1,995,638 to 3,495,585] | [4,099,867 to 7,181,381] |
| Site D | 3,313,104 (4) | 6,297,939 (4) |
| | [2,406,011 to 4,214,400] | [4,573,629 to 8,011,228] |
| Site E | 3,202,107 (3) | 5,940,968 (3) |
| | [2,325,404 to 4,073,207] | [4,314,393 to 7,557,147] |
| Site F | 2,871,420 (2) | 4,566,394 (1) |
| | [2,085,255 to 3,652,560] | [3,316,163 to 5,808,633] |
| Site G | 9,992,992 (6) | 14,931,611 (6) |
| | [7,257,016 to 12,711,482] | [ 10,843,492 to 18,993,601] |

* The Honolulu Consumer Price Index (CPI) inflates money values to 2008 US dollars. The 3% discount rate was used for discounting.
** All values in the bracket [ ] indicate 95% confidence level, and all values in the parenthesis ( ) indicate the ranked order.
*** Aggregate WTP (i) is adjusted for per capita income levels, and aggregate WTP (ii) is adjusted for median household income levels.

The meta-analysis for benefits transfer (MA-BT) approach

Following the meta-analysis for benefits transfer (MA-BT) approach (suggested by Bergstrom and Taylor 2006), the two meta-analysis (MA) models in Table 4.9 were utilized to measure total MWTP: (i) a fully specified model 1 (MA model 1) and (ii) a restricted model (MA model 1) (for the procedure of measuring aggregate MWTP, see pp 97-99 in Chapter 3). By inserting the mean values of study design variables (e.g., N, SE, and FUNCTION) into the meta-analysis models, each target site is assumed to have the same study design variables.

Tables 4.16 and 4.17 show average MWTP (%) per household for MA models 1 and 2. MWTP per household is estimated by multiplying the coefficients of the transfer functions with data for target sites (e.g., Y, POP, MSW, and ACTIVE for MA model 1 and Y and POP for MA model 2) and taking the anti-log (for data, see Table 3.6 in Chapter 3). The results indicate that average MWTP estimates per household vary by the target sites due to differences in income levels, population densities, and model selection. These estimates lie in the 95%

confidence intervals based on Ready (2005), Walton et al. (2006), and this study (see Table 4.18). However, caution should be used for target sites A and G in Kailua with much higher MWTP (about 10.917% to 14.481%) than other target sites (3.995% to 6.878%), which may be attributed to higher income levels and population densities.[64]

Table 4.16. MWTP per Household for Target Sites (MA Model 1)

| Variables | Coefficients | Site A | Site B | Site C | Site D | Site E | Site F | Site G |
|---|---|---|---|---|---|---|---|---|
| Constant | -1.9637136 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Y | 0.0000454 | 93,144 | 74,440 | 66,980 | 74,448 | 76,988 | 74,726 | 88,709 |
| POP | 0.0000143 | 1,587.29 | 606.37 | 604.75 | 517.40 | 513.67 | 386.83 | 1,911.16 |
| MSW | 0.2992723 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| ACTIVE | 0.1702279 | 0 | 0 | 0 | 1 | 0 | 0 | 0 |
| Log MWTP* | | 2.587 | 1.860 | 1.385 | 1.893 | 1.853 | 1.732 | 2.432 |
| MWTP | | 13.290 | 6.424 | 3.995 | 6.640 | 6.285 | 5.661 | 10.917 |

*Log MWTP and MWTP are in percentage terms (%)

Table 4.17. MWTP per Household at Target Sites (MA Model 2)

| Variables | Coefficients | Site A | Site B | Site C | Site D | Site E | Site F | Site G |
|---|---|---|---|---|---|---|---|---|
| Constant | -1.6137703 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Y | 0.0000455 | 93,144 | 77,440 | 66,980 | 74,448 | 76,988 | 74,726 | 88,709 |
| POP | 0.0000306 | 1,587.29 | 606.37 | 604.75 | 517.40 | 513.67 | 386.83 | 1,911.16 |
| Log MWTP* | | 2.673 | 1.928 | 1.452 | 1.789 | 1.905 | 1.798 | 2.481 |
| MWTP | | 14.481 | 6.878 | 4.273 | 5.986 | 6.719 | 6.038 | 11.953 |

*Log MWTP and MWTP are in percentage terms (%).

Table 4.18. Mean Value Estimates and Confidence Interval for MWTP per Household

| | Mean MWTP | 95% confidence interval |
|---|---|---|
| Ready (2005) | 4.12 | [-1.7, 9.9] |
| Walton et al. (2006) | 6.75 | [0.65, 28.45] |
| This Study | 5.13 | [2.53, 7.73] |

* Values are in percentage terms (%)

Tables 4.19 and 4.20 summarize present values and annual values (2008 US dollars) for target sites on Oahu: (i) the mean transfer value for WTP

[64] A confidence interval represents a simple statistical range with some likelihood (e.g., 95% and 99%) for MWTP per household (Desvousges et al. 1998). Alternatively, a range of MWTP estimates from other studies may provide bounds on the probable MWTP for the target sites (Rosenberger and Loomis 2003). For example, when multiple study sites exist, a study site with the lowest MWTP becomes the lower bound of the transfer, and a study site with the highest MWTP becomes the upper bound of the transfer. For this study, the confidence interval is utilized.

approach adjusting for median household income with e=1 (based on CVM research);[65] (ii) the direct mean transfer value for MWTP approach from data used for MA; (iii) the fully specified model (MA model 1); and (iv) the restricted model (MA model 2). Their comparison shows the various estimates for target sites due to market conditions (e.g., income levels, housing values, population densities, and the number of households), different methods used (CVM and HPM), different benefits transfer (BT) approaches used (MA models and mean transfer value approaches), and differences in lengths of landfill life in years.

The results indicate that social costs are substantially high when compared with the City's direct costs. Social costs range from $42 million to $1,299 million (annually $4.6 million to $143 million) while direct costs for a proposed landfill on Oahu range from about $15 million to $82 million ($0.87 million to $8.87 million per year) (for direct costs, see Table 3.5). If planners fail to consider social costs, they will likely grossly underestimate the costs of the landfill and possibly locate the landfill in a higher overall cost location. Careful examination of social costs can help planners to locate a new landfill more efficiently.

The results between Tables 4.19 (present values) and 4.20 (annual aggregate values) show some divergence because target sites have different lengths of landfill life in years (for lengths of landfill life in years, see Table 3.6). One should be cautious that present values for target sites or projects with unequal lengths of landfill life cannot be compared directly. Annual aggregate values measured by an equivalent annual value method are recommended in order to compare sites or projects with unequal lengths of life. Most sites show relatively stable annual aggregate values while Sites A and G (in Kailua) have doubled or tripled respectively. These two sites were located near high populated residential areas, which have higher income levels, population densities, and housing values.

---

[65] For consistency measure, adjustment for median household income with e=1 is chosen. Meta-analysis based on HPM studies employed median household income, and hedonic price models assume the income elasticity of WTP is unity (e=1) (Navrud 2004).

Table 4.19. Present Values of Aggregate values for Target Sites on the Island of Oahu, Hawaii*

| Target Sites | Mean Transfer for WTP (i) | Direct Mean Transfer for MWTP (ii) | MA Model 1 (iii) | MA Model 2 (iv) | Sum of Ranking |
|---|---|---|---|---|---|
| Site A | $198,072,352 (6) | $460,628,468 (7) | $1,192,774,792 (7) | $1,299,709,140 (7) | 27 (7) |
| Site B | $140,774,520 (5) | $113,918,548 (5) | $142,585,365 (5) | $152,664,047 (5) | 20 (5) |
| Site C | $70,914,502 (2) | $85,449,527 (1) | $66,518,216 (1) | $71,143,732 (1) | 5 (1) |
| Site D | $93,697,427 (4) | $99,154,613 (3) | $128,285,258 (4) | $115,651,098 (2) | 13 (4) |
| Site E | $88,386,604 (3) | $89,122,671 (2) | $109,140,597 (2) | $116,673,518 (3) | 10 (2) |
| Site F | $42,251,123 (1) | $109,109,069 (4) | $120,354,772 (3) | $128,366,927 (4) | 12 (3) |
| Site G | $260,006,350 (7) | $306,834,188 (6) | $652,652,896 (6) | $714,611,651 (6) | 25 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars, and the 3% discount rate was used for discounting all values (Freeman 2003). The length of life for each target site was used as the time horizon.
** Numbers in parenthesis () next to the aggregated values indicate rank order.

Table 4.20. Annual Aggregate Values for Target Sites on the Island of Oahu, Hawaii*

| Target Sites | Mean Transfer for WTP (i) | Direct Mean Transfer for MWTP (ii) | MA Model 1 (iii) | MA Model 2 (iv) | Sum of Ranking |
|---|---|---|---|---|---|
| Site A | $18,624,653 (7) | $50,908,400 (7) | $131,824,801 (7) | $143,643,125 (7) | 28 (7) |
| Site B | $8,084,381 (5) | $9,392,471 (4) | $11,756,022 (4) | $12,586,999 (4) | 17 (5) |
| Site C | $5,645,564 (2) | $8,448,484 (2) | $6,576,725 (1) | $7,034,055 (1) | 6 (1) |
| Site D | $6,297,939 (4) | $8,959,514 (3) | $11,591,731 (3) | $10,450,121 (2) | 12 (3) |
| Site E | $5,940,968 (3) | $8,053,038 (1) | $9,861,838 (2) | $10,542,506 (3) | 9 (2) |
| Site F | $4,566,394 (1) | $14,019,215 (5) | $15,464,154 (5) | $16,493,620 (5) | 16 (4) |
| Site G | $14,931,611 (6) | $25,298,175 (6) | $53,810,585 (6) | $58,919,023 (6) | 24 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars. An equivalent annual value method was utilized, which transforms present values to an annual basis for a comparison of other projects with unequal time lengths. The length of life for each target site was used as the time horizon, and the 3% discount rate was used (Freeman 2003).
** Numbers in parenthesis () next to the aggregated values indicate rank order.

In terms of the sum of rankings (see Table 4.20), Sites E and F had lower social costs than the City's target sites A and B. The City's exclusion of these two sites can be explained by several reasons. Site E (1.8 km away from Makakilo city) is near roads (e.g., Ko Olina Lagoon & Roadway Easement) and parks (doughnut and Makaiwa Beach), and Site F (350 meter away from Waikane) is surrounded by parks (e.g., Sacred Falls and Kahana Valley State), the Ewa Forest Reserve, and the Moli'I pond. This site is far (distance) from urban Honolulu, has a narrow winding road (route 83), and culturally sensitive areas for native Hawaiians. Site G (in Kailua) has higher social costs than the City's Sites B, C, and D since this site was located near high populated residential areas, which have higher income levels, population densities, and housing values.

The direct mean transfer value approach (ii) and meta-analysis models 1 (iii) and 2 (iv) overstate the mean transfer value for WTP approach based on CVM research (i) (see Table 4.20). One possible explanation is that while the mean transfer values based on CVM research measure the household's WTP, the others (ii), (iii), and (iv) based on HPM studies measure marginal WTP (the effects of landfills on housing values). Theoretically, the marginal implicit price (MIP) based on HPM represents household's MWTP without housing market distortions (e.g., full information on housing prices and attributes, zero transaction costs and moving costs, and instantaneous price adjustment to changes in either demand or supply) (Freeman 2003). However, the diversion can occur due to market distortions. High demands for housing and/or high population densities on Oahu with the limited space available may lead this overstatement. For example, Site F is ranked first by the mean transfer value for WTP method based on CVM research (i) but fifth by all others (ii), (iii), and (iv), which may be attributed to market distortions (e.g., far distance from Honolulu with higher income levels and housing values).

One caution should be used that homeowners and renters can have different responses to a new landfill.[66] However, when changes in the environmental conditions in the future are expected, this expectation will be reflected in the sales price but not the rental price (Palmquist 2005).

### 4.2.3. Sensitivity Analysis

Sensitivity analysis examines sensitivity of results (e.g., aggregate MWTP estimated by met-analysis models) in response to changes in a selected variable (e.g., income, distance from target sites, discount rates, and lengths of landfill life) ceteris paribus. This sensitivity analysis can show how changes in a selected variable affect aggregate values and rank order for target sites, which can

[66] The sales price (asset price) is the capitalized value of the anticipated future services provided by the house while the rental price is the value of those services over the rental contract period (Palmquist 2005).

support robustness and credibility of the meta-analysis (MA) models and support decision-making. Because meta-analysis (MA) models generally perform better than mean transfer value approaches, sensitivity analysis focuses on MA models with respect to changes in income, distance from target sites, discount rates, and lengths of landfill life ceteris paribus.

Changes in median household income levels

Tables 4.21 and 4.22 provide present values of aggregate MWTP estimates for MA models 1 and 2 with changes in hypothetical income given other variables: (i) no change, (ii) a 50% increase in income, and (iii) a 100 % increase in income. Results indicate that higher income incurs higher social costs. In terms of rank order, when income levels rise by 50%, MA model 1 has consistent rank order for Site A (ranked seventh), Site B (ranked fifth), Site C (ranked first), and Site G (ranked sixth). The other sites vary in ranking. With a 100% increase in income, MA model 1 has consistent rank order for Sites A, B, C, F, and G. The other sites (D and E) vary in ranking (see Table 4.21). When income levels increase by 50%, rank order for MA model 2 is equivalent to the no change case. With a 100 % increase in income levels, MA model 2 has consistent rank order for Sites A, B, C, D, and G. Sites E and F have rankings that are interchanged (see Table 4.22). MA model 2 is less sensitive to changes in income levels than MA model 1.

Table 4.21. Present Values of Aggregate MWTP Estimates Contingent on Changes in Y (MA Model 1) (2008 US$)*

| Target Sites (p) | No Change | 50% Increase in Income | 100% Increase in Income |
|---|---|---|---|
| Site A | $1,192,774,792 (7) | $9,881,370,454 (7) | $81,860,781,032 (7) |
| Site B | $142,585,365 (5) | $674,201,375 (5) | $3,653,046,813 (5) |
| Site C | $66,518,216 (1) | $304,271,944 (1) | $1,391,820,461 (1) |
| Site D | $128,285,258 (4) | $587,676,867 (2) | $3,187,120,908 (2) |
| Site E | $109,140,597 (2) | $626,576,419 (3) | $3,597,176,690 (4) |
| Site F | $120,354,772 (3) | $656,367,471 (4) | $3,579,569,510 (3) |
| Site G | $652,652,896 (6) | $4,889,007,576 (6) | $36,623,440,675 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars, and the 3% discount rate was used for discounting all values (Freeman 2003). The length of life for each target site was used as the time horizon.

** Numbers in parenthesis () next to the aggregated values indicate rank order.

Table 4.22. Present Values of Aggregate MWTP Estimates Contingent on Changes in Y (MA model 2) (2008 US$)*

| Target Sites (p) | No Change | 50% Increase in Income | 100% Increase in Income |
|---|---|---|---|
| Site A | $1,299,709,140 (7) | $10,817,514,144 (7) | $90,034,461,323 (7) |
| Site B | $152,664,047 (5) | $724,331,919 (5) | $3,939,305,582 (5) |
| Site C | $71,143,732 (1) | $326,521,966 (1) | $1,498,608,400 (1) |
| Site D | $115,651,098 (2) | $630,462,629 (2) | $3,431,915,646 (2) |
| Site E | $116,673,518 (3) | $672,406,334 (3) | $3,875,174,815 (4) |
| Site F | $128,366,927 (4) | $702,683,160 (4) | $3,846,501,865 (3) |
| Site G | $714,611,651 (6) | $5,376,935,429 (6) | $40,457,547,218 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars, and the 3% discount rate was used for discounting all values (Freeman 2003). The length of life for each target site was used as the time horizon.
** Numbers in parenthesis () next to the aggregated values indicate rank order.

Tables 4.23 and 4.24 provide annual values of social costs for MA models 1 and 2 with the income changes ceteris paribus: (i) no change, (ii) a 50% increase in income, and (iii) a 100 % increase in income. In terms of rank order, MA model 1 has same rank order for Site A (ranked seventh), Site C (ranked first), Site F (ranked fifth), and Site G (ranked sixth). The other sites vary in ranking. When income rises by 100%, MA model 1 has same results as the 50% change in income. When income increases by 50%, MA model 2 has same rank order for Site A (ranked seventh), Site C (ranked first), Site D (ranked second), Site F (ranked fifth), and Site G (ranked sixth). The other sites vary in ranking. When income rises by 100%, MA model 2 has same results as the 50 % increase in income. Model 2 is (slightly) less sensitive to the changes in income levels.

Table 4.23. Annual Values of Aggregate MWTP Estimates Contingent on Changes in Y (MA model 1) (2008 US$)*

| Target Sites (p) | No Change | 50% Increase in Income | 100% Increase in Income |
|---|---|---|---|
| Site A | $131,824,801 (7) | $1,092,083,521 (7) | $9,047,207,613 (7) |
| Site B | $11,756,022 (4) | $55,587,235 (3) | $301,190,091 (3) |
| Site C | $6,576,725 (1) | $30,083,684 (1) | $137,610,740 (1) |
| Site D | $11,591,731 (3) | $53,101,910 (2) | $287,985,144 (2) |
| Site E | $9,861,838 (2) | $56,616,836 (4) | $325,037,385 (4) |
| Site F | $15,464,154 (5) | $84,335,397 (5) | $459,932,021 (5) |
| Site G | $53,810,585 (6) | $403,093,831 (6) | $3,019,566,401 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars. An equivalent annual value method was utilized, which transforms present values to an annual basis for a comparison of other projects with unequal time lengths. The length of life for each target site was used as the time horizon, and the 3% discount rate was used (Freeman 2003).
** Numbers in parenthesis () next to the aggregated values indicate rank order.

Table 4.24. Annual Values of Aggregate MWTP Estimates Contingent on Changes in Y (MA model 2) (2008 US$)*

| Target Sites (p) | No Change | 50% Increase in Income | 100% Increase in Income |
|---|---|---|---|
| Site A | $143,643,125 (7) | $1,195,545,597 (7) | $9,950,558,175 (7) |
| Site B | $12,586,999 (4) | $59,720,449 (3) | $324,791,843 (3) |
| Site C | $7,034,055 (1) | $32,283,567 (1) | $148,168,974 (1) |
| Site D | $10,450,121 (2) | $56,967,990 (2) | $310,104,558 (2) |
| Site E | $10,542,506 (3) | $60,757,982 (4) | $350,157,026 (4) |
| Site F | $16,493,620 (5) | $90,286,411 (5) | $494,229,647 (5) |
| Site G | $58,919,023 (6) | $443,322,999 (6) | $3,335,684,687 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars. An equivalent annual value method was utilized, which transforms present values to an annual basis for a comparison of other projects with unequal time lengths. The length of life for each target site was used as the time horizon, and the 3% discount rate was used (Freeman 2003).
** Numbers in parenthesis () next to the aggregated values indicate rank order.

### Changes in distance from target sites

As shown previously, the base model assumes that the effects on the impacted area are within a 3 mile-distance of each target site. Tables 4.25 and 4.26 provide present values of aggregate MWTP estimates with changes in distance from target sites (i.e., smaller or larger effects on the impacted area compared with the base model). The results indicate that the larger effects on the impacted area have higher social costs, which are due to an increase in the number of affected households when compared with the lesser effects on the impacted area. In terms of rank order, when the effects on the impacted area change from 3 miles to one mile, the MA model 1 has consistent rank order for Site A (ranked seventh), Site C (ranked first), and Site G (ranked sixth). The other sites vary in ranking. When distance changes from 3 miles to 5 miles, MA model 1 has consistent rank order for Site A (ranked seventh), Site C (ranked first), Site E (ranked second), and Site G (ranked sixth). MA model 2 has similar results. With a change in distance from 3 miles to one mile, MA model 2 has same results as MA model 1. When the effects on impacted area increase from 3 miles to 5 miles, MA model 2 has consistent rank order for Sites A, C, D, E, and G. Sites B and F have rankings that are interchanged.

Table 4.25. Present Values of Aggregate MWTP Estimates Contingent on Distance (MA model 1) (2008 US$)*

| Target Sites (p) | 1 mile | 3 mile (base model) | 5 mile |
|---|---|---|---|
| Site A | $590,191,792 (7) | $1,192,774,792 (7) | $2,756,548,131 (7) |
| Site B | $73,299,104 (4) | $142,585,365 (5) | $255,153,338 (4) |
| Site C | $23,024,762 (1) | $66,518,216 (1) | $85,347,812 (1) |
| Site D | $71,184,304 (3) | $128,285,258 (4) | $214,294,883 (3) |
| Site E | $78,344,455 (5) | $109,140,597 (2) | $210,942,262 (2) |
| Site F | $36,716,683 (2) | $120,354,772 (3) | $345,373,918 (5) |
| Site G | $485,108,481 (6) | $652,652,896 (6) | $1,705,000,525 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars, and the 3% discount rate was used for discounting all values (Freeman 2003). The length of life for each target site was used as the time horizon.
** Numbers in parenthesis () next to the aggregated values indicate rank order.

Table 4.26. Present Values of Aggregate MWTP Estimates Contingent on Distance (Meta-Model 2) (2008 US$)*

| Target Sites (p) | 1 mile | 3 mile (base model) | 5 mile |
|---|---|---|---|
| Site A | $638,625,945 (7) | $1,299,709,140 (7) | $3,022,164,668 (7) |
| Site B | $78,203,072 (4) | $152,664,047 (5) | $274,732,220 (4) |
| Site C | $24,674,501 (1) | $71,143,732 (1) | $91,118,452 (1) |
| Site D | $64,046,695 (3) | $115,651,098 (2) | $194,036,671 (2) |
| Site E | $83,585,975 (5) | $116,673,518 (3) | $226,508,251 (3) |
| Site F | $39,030,315 (2) | $128,366,927 (4) | $368,148,219 (5) |
| Site G | $535,627,319 (6) | $714,611,651 (6) | $1,869,563,573 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars, and the 3% discount rate was used for discounting all values (Freeman 2003). The length of life for each target site was used as the time horizon.
** Numbers in parenthesis () next to the aggregated values indicate rank order.

Tables 4.27 and 4.28 provide annual values with changes in distance from target sites. These results indicate that the larger effects on the impacted area have higher social costs, which are due to an increase in the size or number of affected households when compared with the smaller effects on the impacted area. When the effects on the impacted area change from 3 miles to one mile, the MA model 1 has consistent rank order for Site A (ranked seventh), Site C (ranked first), and Site G (ranked sixth). The other sites vary in ranking. When distance changes from 3 miles to 5 miles, rank order is equivalent to the no change case. With a change in distance from 3 miles to one mile, MA model 2 has same results as MA model 1. When the effects on impacted area increase from 3 miles to 5 miles, MA model 2 has consistent rank order for Sites A (ranked seventh), C (ranked first), D (ranked second), E (ranked third), and G (ranked sixth). The other sites vary in ranking. The interchange in rank order may be attributed to some sites (Sites B, D, E, and F) located close to each other and/or having similar market conditions (e.g., income levels, housing values, and the number of households). The sensitivity in rank order reduces with the larger effects on the impacted area since an increase in the number of household reduces variability.

Table 4.27. Annual Values of Aggregate MWTP Estimates Contingent on Distance (MA model 1) (2008 US$)*

| Target Sites (p) | 1 mile | 3 mile (base model) | 5 mile |
|---|---|---|---|
| Site A | $65,227,666 (7) | $131,824,801 (7) | $304,652,154 (7) |
| Site B | $6,043,438 (3) | $11,756,022 (4) | $21,037,140 (4) |
| Site C | $2,276,482 (1) | $6,576,725 (1) | $8,438,427 (1) |
| Site D | $6,432,144 (4) | $11,591,731 (3) | $19,363,477 (3) |
| Site E | $7,079,129 (5) | $9,861,838 (2) | $19,060,538 (2) |
| Site F | $4,717,656 (2) | $15,464,154 (5) | $44,376,432 (5) |
| Site G | $39,996,714 (6) | $53,810,585 (6) | $140,575,604 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars. An equivalent annual value method was utilized, which transforms present values to an annual basis for a comparison of other projects with unequal time lengths. The length of life for each target site was used as the time horizon, and the 3% discount rate was used (Freeman 2003).

** Numbers in parenthesis () next to the aggregated values indicate rank order.

Table 4.28. Annual Values of Aggregate MWTP Estimates Contingent on Distance (MA model 2) (2008 US$)*

| Target Sites (p) | 1 mile | 3 mile (base model) | 5 mile |
|---|---|---|---|
| Site A | $70,580,581 (7) | $1,299,709,140 (7) | $334,007,944 (7) |
| Site B | $6,447,766 (4) | $152,664,047 (5) | $22,651,399 (4) |
| Site C | $2,439,594 (1) | $71,143,732 (1) | $9,008,976 (1) |
| Site D | $5,787,197 (3) | $115,651,098 (2) | $17,532,965 (2) |
| Site E | $7,552,747 (5) | $116,673,518 (3) | $20,467,065 (3) |
| Site F | $5,014,930 (2) | $128,366,927 (4) | $47,302,659 (5) |
| Site G | $44,161,942 (6) | $714,611,651 (6) | $154,143,664 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars. An equivalent annual value method was utilized, which transforms present values to an annual basis for a comparison of other projects with unequal time lengths. The length of life for each target site was used as the time horizon, and the 3% discount rate was used (Freeman 2003).

** Numbers in parenthesis () next to the aggregated values indicate rank order.

Overall, MA models 1 and 2 have similar results in response to the changes in distance. Rank order is more sensitive to smaller impacted areas (i.e., closer distance to target sites) than for larger impacted areas. This is caused by (i) variability in the smaller impacted area due to fewer numbers of households and (ii) some sites are located close to each other having similar market conditions.

Changes in discount rates and length of landfill life (in years)

Tables 4.29, 4.30, and 4.31 show present values of aggregate MWTP estimates with different discount rates (3% and 7%) and lengths of landfill life (11, 25, and 50 years).[67] Shorter life lengths incur higher social costs, and higher discount rates have lower social costs. One interesting point is that the rank order for MA model 1 is consistent with different interest rates and different lengths of life. MA model 2 also has similar results i.e., rankings are consistent with Sites D and E interchanged with a landfill life of 25 years.

Table 4.29. Present Values of Aggregate MWTP Estimates with Different Discounting Rates (11 years)*

| | MA model 1 | | MA model 2 | |
|---|---|---|---|---|
| | 3% | 7% | 3% | 7% |
| Site A | $1,179,244,603 (7) | $955,709,909 (7) | $1,284,965,946 (7) | $1,041,390,975 (7) |
| Site B | $172,190,339 (5) | $139,550,363 (5) | $184,361,656 (5) | $149,414,516 (5) |
| Site C | $69,487,798 (1) | $56,315,863 (1) | $74,319,812 (1) | $60,231,932 (1) |
| Site D | $145,060,317 (4) | $117,563,042 (4) | $130,774,068 (3) | $105,984,859 (3) |
| Site E | $123,412,229 (3) | $100,018,512 (3) | $131,930,183 (4) | $106,921,823 (4) |
| Site F | $120,354,458 (2) | $97,540,365 (2) | $128,366,592 (2) | $104,033,738 (2) |
| Site G | $788,163,098 (6) | $638,760,848 (6) | $862,986,337 (6) | $699,400,779 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars, and the 3% discount rate was used for discounting all values (Freeman 2003). The length of life for each target site was used as the time horizon.
** Numbers in parenthesis () next to the aggregated values indicate rank order.

[67] The 11 and 25 years are the minimum and maximum lengths from the Oahu study, and the 50 year length is from Roberts et al. (1991).

Table 4.30. Present Values of Aggregate MWTP Estimates with Different Discounting Rates (25 years)*

| | MA model 1 | | MA model 2 | |
|---|---|---|---|---|
| | 3% | 7% | 3% | 7% |
| Site A | $976,492,429 (7) | $653,510,987 (7) | $1,064,036,684 (7) | $712,099,391 (7) |
| Site B | $142,584,975 (5) | $95,424,035 (5) | $152,663,629 (5) | $102,169,107 (5) |
| Site C | $57,540,487 (1) | $38,508,584 (1) | $61,541,713 (1) | $41,186,378 (1) |
| Site D | $120,119,525 (4) | $80,389,184 (4) | $108,289,567 (3) | $72,472,064 (4) |
| Site E | $102,193,478 (3) | $68,392,297 (3) | $109,246,906 (4) | $73,112,757 (3) |
| Site F | $99,661,442 (2) | $66,697,749 (2) | $106,296,018 (2) | $71,137,895 (2) |
| Site G | $652,651,109 (6) | $436,782,363 (6) | $714,609,694 (6) | $478,247,729 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars, and the 3% discount rate was used for discounting all values (Freeman 2003). The length of life for each target site was used as the time horizon.
** Numbers in parenthesis () next to the aggregated values indicate rank order.

Table 4.31. Present Values of Aggregate MWTP Estimates with Different Discounting Rates (50 years)*

| | MA model 1 | | MA model 2 | |
|---|---|---|---|---|
| | 3% | 7% | 3% | 7% |
| Site A | $721,438,311 (7) | $386,958,068 (7) | $786,116,518 (7) | $421,649,536 (7) |
| Site B | $105,342,612 (5) | $56,502,646 (5) | $112,788,781 (5) | $60,496,550 (5) |
| Site C | $42,511,248 (1) | $22,801,770 (1) | $45,467,377 (1) | $24,387,350 (1) |
| Site D | $88,745,007 (4) | $47,600,184 (4) | $80,004,964 (3) | $42,912,285 (3) |
| Site E | $75,501,138 (3) | $40,496,567 (3) | $80,712,253 (4) | $43,291,653 (4) |
| Site F | $73,630,455 (2) | $39,493,188 (2) | $78,532,119 (2) | $42,122,295 (2) |
| Site G | $482,182,452 (6) | $258,628,336 (6) | $527,957,816 (6) | $283,180,881 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars, and the 3% discount rate was used for discounting all values (Freeman 2003). The length of life for each target site was used as the time horizon.
** Numbers in parenthesis () next to the aggregated values indicate rank order.
.

Tables 4.32, 4.33, and 4.34 show annual values of aggregate MWTP estimates with different discount rates (3% and 7%) and lengths of landfill life (11, 25, and 50 years). The rank order for each MA model is identical. A shorter length of life incurs higher annual social costs. However, higher discount rates (i.e., lower annuity factors) have higher annual social costs. The equivalent annual value (EAV) method transforms one time values to an annual basis, and annual values are measured when one time values are divided by the annuity factor. Higher discount rates incur lower annuity factors, which result in higher annual social costs.

Table 4.32. Annual Aggregate MWTP Estimates with Different Discounting Rates (11 years)*

| | MA model 1 | | MA model 2 | |
|---|---|---|---|---|
| | 3% | 7% | 3% | 7% |
| Site A | $151,519,667 (7) | $186,959,189 (7) | $165,103,670 (7) | $203,720,408 (7) |
| Site B | $22,124,522 (5) | $27,299,312 (5) | $23,688,399 (5) | $29,228,970 (5) |
| Site C | $8,928,400 (1) | $11,016,699 (1) | $9,549,260 (1) | $11,782,773 (1) |
| Site D | $18,638,619 (4) | $22,998,078 (4) | $16,802,997 (3) | $20,733,115 (3) |
| Site E | $15,857,083 (3) | $19,565,958 (3) | $16,951,545 (4) | $20,916,407 (4) |
| Site F | $15,464,194 (2) | $19,081,174 (2) | $16,493,663 (2) | $20,351,430 (2) |
| Site G | $101,270,093 (6) | $124,956,547 (6) | $110,884,037 (6) | $136,819,134 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars. An equivalent annual value method was utilized, which transforms present values to an annual basis for a comparison of other projects with unequal time lengths. The length of life for each target site was used as the time horizon, and the 3% discount rate was used (Freeman 2003).
** Numbers in parenthesis () next to the aggregated values indicate rank order.

Table 4.33. Annual Aggregate MWTP Estimates with Different Discounting Rates (25 years)*

| | MA model 1 | | MA model 2 | |
|---|---|---|---|---|
| | 3% | 7% | 3% | 7% |
| Site A | $80,511,274 (7) | $120,301,956 (7) | $87,729,251 (7) | $131,087,237 (7) |
| Site B | $11,756,054 (5) | $17,566,190 (5) | $12,587,034 (5) | $18,807,860 (5) |
| Site C | $4,744,182 (1) | $7,088,875 (1) | $5,074,081 (1) | $7,581,819 (1) |
| Site D | $9,903,790 (4) | $14,798,490 (4) | $8,928,416 (3) | $13,341,063 (3) |
| Site E | $8,425,797 (3) | $12,590,036 (3) | $9,007,349 (4) | $13,459,005 (4) |
| Site F | $8,217,032 (2) | $12,278,095 (2) | $8,764,049 (2) | $13,095,461 (2) |
| Site G | $53,810,732 (6) | $80,405,339 (6) | $58,919,184 (6) | $88,038,515 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars. An equivalent annual value method was utilized, which transforms present values to an annual basis for a comparison of other projects with unequal time lengths. The length of life for each target site was used as the time horizon, and the 3% discount rate was used (Freeman 2003).
** Numbers in parenthesis () next to the aggregated values indicate rank order.

Table 4.34. Annual Aggregate MWTP Estimates with Different Discounting Rates (50 years)*

| | MA model 1 | | MA model 2 | |
|---|---|---|---|---|
| | 3% | 7% | 3% | 7% |
| Site A | $54,487,438 (7) | $101,585,490 (7) | $59,372,332 (7) | $110,692,807 (7) |
| Site B | $7,956,119 (5) | $14,833,258 (5) | $8,518,499 (5) | $15,881,751 (5) |
| Site C | $3,210,710 (1) | $5,985,995 (1) | $3,433,975 (1) | $6,402,246 (1) |
| Site D | $6,702,566 (4) | $12,496,155 (4) | $6,042,465 (3) | $11,265,473 (3) |
| Site E | $5,702,308 (3) | $10,631,290 (3) | $6,095,883 (4) | $11,365,066 (4) |
| Site F | $5,561,023 (2) | $10,367,880 (2) | $5,931,226 (2) | $11,058,082 (2) |
| Site G | $36,417,370 (6) | $67,895,952 (6) | $39,874,606 (6) | $74,341,566 (6) |

* The Honolulu Consumer Price Index (CPI) adjusts money values to 2008 US dollars. An equivalent annual value method was utilized, which transforms present values to an annual basis for a comparison of other projects with unequal time lengths. The length of life for each target site was used as the time horizon, and the 3% discount rate was used (Freeman 2003).
** Numbers in parenthesis () next to the aggregated values indicate rank order.

The sensitivity analysis results indicate that income levels, distance from the landfill, discount rates, and length of landfill life in years affect the level of social costs. MA models 1 and 2 have similar results in response to these changes. In terms of rank order, MA models have consistent rank order for Sites A (ranked seventh), C (ranked first), and G (ranked sixth). The other sites vary in ranking. This interchange may be attributed to some sites (Sites B, D, E, and F) having similar social costs. Sites B, D, and E (in leeward side) are located close to each other and have similar market conditions as income levels, housing values, and the number of households. Although Site F (near Waikane) is far away from these three sites, this site has similar income levels and housing values as Sites B, D, and E. It should be noted that the rank order for MA model 1 is consistent with different interest rates for 11, 25, and 50 years (each target site with the same life in years). MA model 2 has similar results with MA model 1 i.e., others are consistent with Site D and Site E interchanged for a landfill life of 25 years.

The MA models are good at picking out the high and the low i.e., Site C (ranked first) always comes out on top; sites A (ranked seventh) and G (ranked sixth) are always on the bottom. However, it is not good at differentiating the 4 remaining sites with similar market conditions in the middle. The MA models can be a valuable way to identify the top or low few, which allows decision-makers to use scarce resources more effectively to focus subsequent detailed analysis on a smaller number of options.

### 4.2.4. Reliability and validity checks

Meta-analysis (MA) models were evaluated in terms of validity and reliability criteria. While validity requires statistical tests (e.g., a parametric t-test and a non-parametric sign rank test) to examine mean differences or differences between transfer values and original values, reliability checks their similarity by utilizing a transfer error measured by an absolute percentage difference (Navrud and Ready 2007). The N-j MA transfer method suggested by Lindhjem and Nvrud (2008) was utilized to check validity and reliability for MA models and direct mean transfer values for MWTP approach (for the procedure see pp 99-101 in Chapter 3).

A t-test and a sign rank test were employed to examine whether transfer values estimated by meta-analysis models were statistically different from original values. While a t-test is a common parametric test assuming that population is normally distributed, a sign rank test without assuming a normal distribution of population is often utilized when non-normality occurs in small samples (Brouwer and Spaninks 1999; Shrestha and Loomis 2001). Equation (3-12) measures transfer errors measured by the absolute percentage difference between transfer values and original or primary values. In terms of reliability, smaller TE is preferred. These transfer errors were utilized to check reliability in terms of assessing their sensitivity. Transfer errors were sorted according to income levels and population densities.

Table 4.35 reports t and sign rank test results. For the MA models and the direct transfer value approach, the null hypotheses of mean differences or differences between transfer values and original values are not rejected at the 5 % level, which indicates that transfer values estimated by mean MWTP and MA transfer functions are insignificantly different from original values.

Table 4.35. Paired t-test and Sign Rank Test*

| Test | Mean MWTP** | MA model 1 | MA model 2 |
|---|---|---|---|
| Paired t-statistics | 0.5000 | 0.9105 | 0.8865 |
| Sign rank test statistics | 0.1485 | 0.9612 | 0.8838 |

* All values are p-values.

** Mean values used in meta-analysis (MA) was utilized.

Table 4.36 shows transfer error (TE), average TE, the range of TE, and mean TE for each subgroup based on the mean MWTP approach using mean MWTP used in meta-analysis, MA model 1, and MA model 2.[68] The transfer error (TE) is highly sensitive to the benefits transfer (BT) methods used and to market conditions (e.g., population densities and income levels). Overall, in terms of the

[68] These are based on HPM studies used for meta-analysis.

mean TE, MA models have lower mean TE than the direct mean transfer for the MWTP approach, and MA model 2 has lower mean TE than MA model 1. Mean TE for target sites with higher income levels is relatively small compared with other subgroups. These findings are consistent with Rosenberger and Loomis (2003) and Rosenberger and Phipps (2007).[69] In terms of reliability (mean TE), MA models are preferred to the direct mean transfer value approach, and MA model 2 is preferred to MA model 1.

Table 4.36. TE and Mean TE: Income Levels (Y) and Population Densities (POP)*

| | Mean MWTP (i) | MA model 1 (ii) | MA model 2 (iii) |
|---|---|---|---|
| Low POP | 127.72** | 64.64 | 61.47 |
| | [13.32, 722.42]*** | [3.08, 287.66] | [2.73, 293.79] |
| High POP | 238.13 | 186.54 | 155.02 |
| | [19.69, 1687.46] | [33.94, 651.31] | [10.56, 522.17] |
| Low Y | 291.73 | 182.09 | 155.99 |
| | [13.32, 1687.46] | [11.87, 651.31] | [10.85, 522.17] |
| High Y | 73.1 | 69.09 | 55.61 |
| | [18.04,211.51] | [3.08, 193.82] | [2.73, 192.58] |
| Mean $\delta$ (n-j) | 193.74 | 125.59 | 107.45 |
| | [13.32, 1687.46] | [3.08, 651.31] | [2.72, 522.17] |

* Eleven observations for each subgroup were selected for sensitivity testing of the meta-models
** Numbers indicates the mean transfer error in percentage terms (%).
*** Numbers in bracket indicate the range of transfer error mean..

Example 1 of reliability check: three target sites in Minnesota (MN) that conducted primary HPM

A comparison between primary values based on original hedonic price method (HPM) studies and transfer values based on benefits transfer (BT) methods allows checking reliability results (Rosenberger and Loomis 2003; Lindhjem and Navrud 2008; Navrud and Ready 2007). In order to evaluate the meta-analysis (MA) transfer functions based on HPM studies, conducting primary HPM is not possible for the target sites on Oahu without landfills. Three target

[69] One should caution that the results for meta-analysis (MA) models may be sensitive to the dropped observation for each target site. Different results could occur if the sample size increases.

sites (Ramsey, Eden Prairie, and Oakgrove in MN) were chosen following criteria for valid and reliable benefits transfer (Bergstom and Taylor 2006): (i) welfare change consistency (e.g., HPM), (ii) commodity consistency (e.g., distance from municipal solid waste landfills), and (iii) theory consistency (e.g., positive distance effects). Target sites provide original MWTP using primary HPM and are similar to the Oahu case since (i) target sites are in the same state and (ii) the target sites have similar high income levels. The target sites have relatively lower TE because the target sites in state have similar characteristics (e.g., income levels and population densities). Generally, values resulted from in-state transfers with similar characteristics performs better than across states (Rosenberger and Phipps 2007).

Table 4.37 summarizes original estimates, transfer values, transfer error (TE), and average TE for the three target sites. TE for direct mean transfer values ranges from 18.04 % to 113.80% (mean transfer error = 50.51%). The TE for MA model 1 ranges from 11.87% to 42.97% (mean transfer error = 29.60%), and the TE error for MA model 2 ranges from 10.85% to 45.08% (mean transfer error = 27.10%). MA models have lower mean TE than the direct mean transfer value for the MWTP approach and MA model 2 has lower mean TE than MA model 1.

Table 4.37. Transfer Error (TE) for Mean MWTP Transfer and Meta-Analysis Transfer Functions

| Primary HPM Research | Target Sites (j) | Original Value (j) | Mean MWTP Estimates (-j) | MA Model 1 (-j) | MA Model 2 (-j) |
|---|---|---|---|---|---|
| Nelson et al. (1992a) | Ramsey, MN (Site 1) | 6.20* | 5.09 (18.04)** | 3.56 (42.97) | 3.38 (45.08) |
| Nelson et al. (1998) | Eden Prairie, MN (Site 2) | 4.32 | 5.17 (19.70) | 5.79 (33.98) | 5.40 (25.00) |
| Ready (2005) | Oakgrove, MN (Site 3) | 2.46 | 4.88 (113.80) | 2.17 (11.87) | 2.19 (10.85) |
| Mean TE | | | 50.51 | 29.60 | 26.98 |

* Values are MWTP estimates in percentage terms.
** Numbers in parenthesis () indicates transfer errors in percentage terms

Table 4.38. Data for Reliability Test

| Location | Number of Household | Per Capita Income | Median Household Income | Median Housing Values | Population Density (population per square mile) |
|---|---|---|---|---|---|
| Ramsey, MN (Site 1) | 5,946 | $32,579 | $86,256 | $179,456 | 430 |
| Eden Prairie, MN (Site 2) | 20,457 | $48,579 | $97,934 | $247,936 | 1213 |
| Oakgrove, MN (Site 3) | 2,200 | $28,744 | $87,732 | $193,297 | 204.9 |
| Carter Community in Knox County, TN[70] | 798 | $27,942 | $46,829 | $123,155 | 751 |

Source: U.S. Census (2000)
* Data of the town or city levels where municipal solid landfills are located. Thus, the figures may not exactly match the exact area within a 3-mile distance of each target site.
** The Honolulu Consumer Price Index (CPI) adjusts all money values to 2008 US dollars.

For comparison purposes, present values and annual values of aggregate values are based on four methods: (i) the mean transfer value for WTP approach based on the CVM research; (ii) the direct mean transfer value for MWTP for each target site; (iii) the N-j mean transfer value for MWTP except for the jth target site; and (iv) the N-j MA transfer function (MA model 2). MA model 2 was utilized because MA model 2 had lower mean transfer errors than MA model 1, and sensitivity analysis results were similar for both MA models.[71]

Utilizing equations (3-8) and (3-9) in Chapter 3 one obtains aggregate WTP estimates (present values). Utilizing equations (3-10) and (3-11) in Chapter 3 one obtains three types of aggregate MWTP. Because aggregate MWTP is based on HPM studies, the equivalent annual value method was utilized to compare those of other projects or target sites with (see Appendix B) (For data see Table 4.38).

Tables 4.39, 4.40, and 4.41 summarize present values with the different lengths of landfill life (11, 25, and 50 years).[72] Tables 4.42, 4.43, and 4.44 provide annual values. The results support previous findings: (i) values are various with methods used (HPM versus CVM), market conditions (e.g., income

[70] The number of households is from the Carter community, but other data are from Knox County.
[71] MA model 2 has similar regression fits as MA model 1, but MA model 2 performs better than MA model 1 in terms of individual significance and mean square errors
[72] A 13 year and a 25 year is the minimum and maximum year from the island of Oahu case, and a 50 year is from Roberts et al. (1991).

levels, population densities, the number of households, and housing values), benefits transfer (BT) methods, discount rates (3% and 7%), and the length of landfill life in years, and (ii) the rank order is consistent between BT methods. One should be cautious with the assumption of the same length of year. If each target site has different lengths of life, different results can occur (refer to the Oahu case).

Table 4.39. Present Values of Aggregate Values (Life = 11 years)

| Target Sites | Mean Transfer For WTP* | Primary MWTP | N-j Mean MWTP | N-j MA Model 2 |
|---|---|---|---|---|
| Site 1 | $42,662,394 | $46,079,808 | $45,595,051 | $34,106,534 |
| | ($34,575,416) | ($30,838,602) | ($36,952,166) | ($27,641,384) |
| Site 2 | $166,650,430 | $152,616,457 | $220,568,343 | $230,380,861 |
| | ($135,060,586) | ($102,137,537) | ($178,757,953) | ($186,710,434) |
| Site 3 | $16,055,052 | $7,286,500 | $17,455,758 | $13,735,679 |
| | ($13,011,696) | ($4,876,441) | ($14,146,888) | ($11,131,977) |

* Mean transfer for WTP is adjusted for the median household income with e=1.
** The Consumer Price Index (CPI) and a 3% discount rate adjust money values to 2008 US dollars. For sensitivity analysis, all values in parenthesis () utilized a 7% discount rate. .

Table 4.40. Present Values of Aggregate Values (Life = 25 years)

| Target Sites | Mean Transfer For WTP* | Primary MWTP | N-j Mean MWTP | N-j MA Model 2 |
|---|---|---|---|---|
| Site 1 | $80,289,273 | $46,079,808 | $37,755,714 | $28,242,463 |
| | ($53,733,056) | ($30,838,602) | ($25,267,757) | ($18,901,078) |
| Site 2 | $313,630,828 | $152,616,457 | $182,645,158 | $190,770,572 |
| | ($209,895,321) | ($102,137,537) | ($122,234,043) | ($127,671,921) |
| Site 3 | $30,215,099 | $7,286,500 | $14,454,521 | $11,374,049 |
| | ($20,221,251) | ($4,876,441) | ($9,673,591) | ($7,612,006) |

* Mean transfer for WTP is adjusted for the median household income with e=1.
** The Consumer Price Index (CPI) and a 3% discount rate adjust money values to 2008 US dollars. For sensitivity analysis, all values in parenthesis () utilized a 7% discount rate. .

Table 4.41. Present Values of Aggregate Values (Life = 50 years)

| Target Sites | Mean Transfer For WTP* | Primary MWTP | N-j Mean MWTP | N-j MA Model 2 |
|---|---|---|---|---|
| Site 1 | $118,636,368 | $34,044,032 | $27,894,142 | $20,865,697 |
| | ($63,633,022) | ($18,260,207) | ($14,961,589) | ($11,191,740) |
| Site 2 | $463,424,575 | $112,753,930 | $134,939,310 | $140,942,413 |
| | ($248,567,169) | ($60,477,857) | ($72,377,435) | ($75,597,321) |
| Site 3 | $44,646,183 | $5,383,309 | $10,679,084 | $8,403,214 |
| | ($23,946,886) | ($2,887,447) | ($5,727,944) | ($4,507,234) |

* Mean transfer for WTP is adjusted for the median household income with e=1.
** The Consumer Price Index (CPI) and a 3% discount rate adjust money values to 2008 US dollars. For sensitivity analysis, all values in parenthesis () utilized a 7% discount rate. .

Table 4.42. Annual Values of Aggregate Values (Life = 11 years)

| Target Sites | Mean Transfer For WTP* | Primary MWTP | N-j Mean MWTP | N-j MA Model 2 |
|---|---|---|---|---|
| Site 1 | $4,610,855 | $7,150,078 | $5,858,451 | $4,382,306 |
| | | ($8,822,438) | ($7,228,707) | ($5,407,301) |
| Site 2 | $18,011,200 | $23,681,079 | $28,340,551 | $29,601,349 |
| | | ($29,219,939) | ($34,969,232) | ($36,524,924) |
| Site 3 | $1,735,194 | $1,130,626 | $2,242,869 | $1,764,880 |
| | | ($1,395,079) | ($2,767,462) | ($2,177,675) |

* Mean transfer for WTP is adjusted for the median household income with e=1.
** The Consumer Price Index (CPI) and a 3% discount rate adjust money values to 2008 US dollars. For sensitivity analysis, all values in parenthesis () utilized a 7% discount rate. An equivalent annual value method was utilized, which transforms present values to annual values.

Table 4.43. Annual Values of Aggregate Values (Life = 25 years)

| Target Sites | Mean Transfer For WTP* | Primary MWTP | N-j Mean MWTP | N-j MA Model 2 |
|---|---|---|---|---|
| Site 1 | $4,610,855 | $3,799,255 | $3,112,938 | $2,328,576 |
| | | ($5,676,942) | ($4,651,430) | ($3,479,416) |
| Site 2 | $18,011,200 | $12,583,145 | $15,058,995 | $15,728,931 |
| | | ($18,802,049) | ($22,501,526) | ($23,502,561) |
| Site 3 | $1,735,194 | $600,768 | $1,191,767 | $937,784 |
| | | ($897,683) | ($1,780,769) | ($1,401,261) |

* Mean transfer for WTP is adjusted for the median household income with e=1.
** The Consumer Price Index (CPI) and a 3% discount rate adjust money values to 2008 US dollars. For sensitivity analysis, all values in parenthesis () utilized a 7% discount rate. An equivalent annual value method was utilized, which transforms present values to an annual basis for a comparison of other projects with unequal time lengths.
.

Table 4.44. Annual Values of Aggregate Values (Life = 50 years)

| Target Sites | Mean Transfer For WTP* | Primary MWTP | N-j Mean MWTP | N-j MA Model 2 |
|---|---|---|---|---|
| Site 1 | $4,610,855 | $3,799,255 | $3,112,938 | $2,328,576 |
| | | ($5,676,942) | ($4,651,430) | ($3,479,416) |
| Site 2 | $18,011,200 | $12,583,145 | $15,058,995 | $15,728,931 |
| | | ($18,802,049) | ($22,501,526) | ($23,502,561) |
| Site 3 | $1,735,194 | $600,768 | $1,191,767 | $937,784 |
| | | ($897,683) | ($1,780,769) | ($1,401,261) |

* Mean transfer for WTP is adjusted for the median household income with e=1.
** The Consumer Price Index (CPI) and a 3% discount rate adjust money values to 2008 US dollars. For sensitivity analysis, all values in parenthesis () utilized a 7% discount rate. An equivalent annual value method was utilized, which transforms present values to to annual values

Example 2 of reliability check: One target site in Knox County where primary contingent valuation method (CVM) was conducted

A comparison of aggregate values based on the meta-analysis (MA) transfer function based on HPM studies with those of the primary CVM may act as another reliability check (Brisson and Pearce 1995; Walton et al. 2006). The target site is the proposed landfill site in Knox County, Tennessee where Roberts et al. (1991) conducted a primary CVM. They estimated an annual WTP per household of $410. When WTP per household is multiplied by the number of households (795), the annual aggregate value of $327,180 is obtained.[73] Values estimated by the MA transfer function (MA model 2) are compared with those of the primary CVM research. The MA transfer function and equations (3-10) and (3-11) in Chapter 3 are used for measuring aggregate MWTP at the target site (for data, see Table 4.38). Present values are considered reflecting social cost differences over time. For a comparison of annual values estimated by Roberts et al. (1991), the equivalent annual value (EAV) method was utilized assuming discount rates of 3% and 7% and useful life periods of 11, 25, and 50 years (see Appendix B).

Tables 4.45 and 4.46 show present values and annual values of aggregate values based on (i) the primary CVM study by Roberts et al. (1991) and (ii) the meta-analysis model based on the HPM research. The values also vary with methods used (HPM versus CVM), market conditions (e.g., income levels, population densities, the number of households, and housing values), discount rates, and landfill life in years. With a longer landfill life, present values based on CVM increase, but present values from MA based on HPM decrease.[74] While higher discount rates decrease present values for the two methods, higher discount rates increase annual values from MA. The equivalent value method measures annual values when one time values divided by the annuity factor. Higher discount rates incur lower annuity factors, which result in higher annual social costs.

---

[73] An annual mean value for WTP per household was $410 with 95 percent confidence limits of $310 to $543, and the number of the affected residents were 795 (Roberts et al. 1991). The Honolulu Consumer Price Index (CPI) is used to adjust monetary values for 2008 US dollars.
[74] While CVM research measures annual costs, MA based on HPM measures total costs at a specific time (i.e., impacts of landfills on housing values). With the different time periods, the difference between social costs may occur.

Table 4.45. Present Values of Social Costs for Carter Community in Knox County, TN*

| | Primary CVM Study by Roberts et al. (1991) | | MA Model 2 | |
|---|---|---|---|---|
| Life Lengths | 3% | 7% | 3% | 7% |
| 11 Years | $3,027,266 [$2,288,908, $4,009,281] | $2,453,425 [$1,855,028, $3,249,292] | $1,418,452 | $1,149,574 |
| 25 Years | $5,697,218 [$4,307,653 $7,545,340] | $3,812,825 [$2,882,868, 5,049,668] | $1,174,572 | $786,074 |
| 50 Years | $8,418,276 [$6,365,038,$11,149,083] | $4,515,313 [$3,414,017, $5,980.037] | $867,781 | $465,452 |

* The Honolulu Consumer Price Index (CPI) inflates money values to 2008 US dollars. The 3% and 7% discount rates and useful life periods of 11, 25, and 50 years were utilized for discounting.
** All values in the bracket [] indicates 95% confidence intervals.

Table 4.46. Annual Values of Social Costs for Carter Community in Knox County, TN*

| | Primary CVM Study by Roberts et al. (1991) | MA Model 2 | |
|---|---|---|---|
| Life Year | Annual Aggregate Values | 3% | 7% |
| 11 Year | $327,180 [$247,380, $43,314]** | $182,255 | $224,884 |
| 25 Year | $327,180 [$247,380, $43,314] | $96,843 | $144,705 |
| 50 Year | $327,180 [$247,380, $43,314] | $65,540 | $122,192 |

* The Honolulu Consumer Price Index (CPI) inflates money values to 2008 US dollars. The equivalent annual value (EAV) method was utilized assuming the 3% and 7% discount rates and useful life periods of 11, 25, and 50 years.
** The bracket [] indicates 95% confidence intervals.

Equation (3-12) in Chapter 3 was utilized to measure absolute percentage differences (i.e., transfer errors) between the values reported in Tables 4.45 and 4.46. Table 4.47 shows the variability of transfer errors. These errors range from 53.14% to 89.69% (mean transfer error = 74.07%) for present values and from 31.27% to 79.97% (mean transfer error = 57.39%) for annual values. Transfer errors are affected by discount rates and the length of landfill life, which is consistent with Walton et al. (2006). One interesting point is that transfer errors for present values are the same, even though the two benefits transfer methods utilize the different discount rates (3% or 7%) for the same landfill life period. Overall, annual values have lower mean TE than present values, and longer lengths of landfill life incur higher transfer errors.

Table 4.47. Absolute Percentage Difference (Transfer Error)

| | Present Values | | Annual Values | |
|---|---|---|---|---|
| Useful Life Periods | 3% discount rate | 7% discount rate | 3% discount rate | 7% discount rate |
| 11 Years | 53.14*<br>[38.03, 64.62]** | 53.14<br>[38.03, 64.62] | 44.30<br>[26.33 57.94] | 31.27<br>[9.11, 48.11] |
| 25 Years | 79.38<br>[72.72, 84.43] | 79.38<br>[72.72, 84.43] | 70.40<br>[60.85 77.65] | 55.77<br>[41.50, 66.61] |
| 50 Years | 89.69<br>[86.37, 92.22] | 89.69<br>[86.37, 92.22] | 79.97<br>[73.51 84.87] | 62.65<br>[50.61, 71.80] |
| Mean Transfer Error (MTE) | 74.07 | | 57.39 | |

* Values indicate Transfer error (TE) measured by the absolute percentage error
** All values in the bracket [] indicates TE of the 95% confidence intervals.

The present values and/or annual values of the aggregate MWTP estimates measured by the meta-analysis function (the impacts of landfills on housing values) underestimate those of the aggregate WTP based on primary CVM research (see Walton et al. 2006). Unlike the understatement, the meta-analysis function (MA model 2) for the Oahu case overestimates the mean transfer value for WTP approach based on CVM research (see Table 4.19). One possible explanation is that while the MA function based on HPM studies measures marginal WTP (the effects of landfills on housing values), the mean transfer values based on CVM research measure the individual's WTP. The diversion can occur due to market conditions (e.g., the number of households, income levels, housing values, and population densities) and different landfill life lengths in years.

Overall, social costs are high and vary by target sites due to different market conditions, different methods used, different models used, and differences in discount rates and lengths of landfill life in years. In terms of sensitivity, validity, and reliability criteria, the meta-analysis models perform better than the direct mean transfer value approach.

Although environmental justice or equity and participation are important in the process of landfill site selection, this study did not manage the related issues. This study assumes that compensation for impacted households and aid to reduce transaction costs for low-income households (e.g., providing legal service and information) can improve inequity. Managing participation is not desirable for this preliminary study, which has several reasons (e.g., need for rapid decision-making and lack of knowledge of complex issues on the part of the participants).

The following chapter will discuss (i) a review of this study, (ii) policy implications, and (iii) conclusion and discussion.

# CHAPTER 5. CONCLUSION AND DISCUSSION

## 5.1. Review of this study

Chapter 1 provides background information about the existing waste management situation in the City & County of Honolulu: (i) the waste management system, (ii) the existing municipal solid (MSW) landfill, and (iii) the current problems for landfill site selection. This chapter also briefly discusses the objectives and methodology for this study.

Chapter 2 provides: (i) definitions of terms and basic concepts involving waste management and the landfill site; (ii) anticipated impacts of landfills on the environment and nearby community (e.g., physical, chemical, and biological degradation); (iii) a discussion of the proper operation and effective management necessary to mitigate these adverse impacts; (iv) the landfill site selection process with the use of a GIS including a process of site selection and exclusionary criteria for landfill site selection; (v) environmental justice and participation; and (vi) economic valuation methods (e.g., hedonic price methods, contingent valuation methods, and benefits transfer methods) to measure external effects from landfills.

Chapter 3 describes the proposed integrative approach that links a GIS-based analysis with an economic framework. The basic framework minimizes social costs given constraints (exclusionary criteria) with a two-part process: (i) a GIS-based screening of possible landfill sites that satisfy constraints (e.g., airports, wetlands, floodplains, lane use, and groundwater) and (ii) benefits transfer (BT) methods (e.g., meta-analysis and mean transfer value approaches) that rank the remaining selected sites according to social cost minimization. This chapter discusses: (i) the procedure and data used for the GIS-based preliminary screening; (ii) the process and data utilized for benefits transfer (BT) methods; and (iii) the procedure to check validity and reliability of BT methods.

Chapter 4 presents the results of this study. First, the GIS-based analysis was conducted on the City's 45 potential sites (Scenario 1) and the entire island of Oahu (Scenario 2). Second, meta-analysis (MA) models 1 (a fully specified model) and 2 (a restricted model) with reasonable diagnostic test results and the mean transfer value for WTP or MWTP approach were utilized to measure social costs for the selected target sites and to rank these target sites according to social cost minimization. MA models were evaluated in terms of sensitivity, validity, and reliability criteria. Sensitivity analysis examined sensitivity of aggregate values in response to changes in a selected variable (e.g., income, distance from target sites, discount rates, and lengths of landfill life) ceteris paribus. In order to check validity (statistical difference between transfer values and original values), a parametric t-test and a non-parametric sign rank test were utilized. In order to check reliability (similarity between transfer values and original values), a transfer error measured by an absolute percentage difference was utilized.

In Scenario 1, the GIS analysis selected 4 sites (called A, B, C, and D). These 4 sites were also part of the 8 sites that the City & County of Honolulu selected after preliminary screening. This can be logically explained since this study used the City's 45 potential sites with the same restrictions. However, findings from the GIS analysis differ from those of the City i.e., a smaller number of sites (4 sites) than the the City (8 sites). In Scenario 2, 4 sites were also selected. One site (Site C) is the same as in Scenario 1, and the other three sites (Sites E, F, and G) are in different locations. This can be explained by the application of GIS analysis to the entire island of Oahu (Sites E, F, and G are not part of the City's original 45 sites). Together, both scenarios found a total of 7 sites that satisfy the criteria. Benefits transfer (BT) methods were utilized to measure social costs for the selected 7 sites and rank these sites according to social cost minimization.

The results of benefits transfer approaches show that social costs are high when compared with the City's direct costs. If planners fail to consider social costs, they will likely grossly underestimate the costs of the landfill and possibly locate the landfill in a higher overall cost location.

The results between present values and annual aggregate values show some divergence because target sites have different lengths of landfill life in years (for lengths of landfill life in years, see Table 3.6). Present values for target sites or projects with unequal lengths of landfill life cannot be compared directly. Annual aggregate values measured by an equivalent annual value method are recommended in order to compare target sites or projects with unequal lengths of life (see Tables 4.19 and 4.20). Most sites show relatively stable annual aggregate values while Sites A and G (in Kailua) have doubled or tripled respectively. These two sites were located near high populated residential areas, which have higher income levels, population densities, and housing values (see Table 4.20).

In terms of the sum of rankings (see Table 4.20), Sites E and F had lower social costs than the City's target sites A and B. The City's exclusion of these two sites can be explained by several reasons. Site E (1.8 km away from Makakilo city) is near roads (e.g., Ko Olina Lagoon & Roadway Easement) and parks (doughnut and Makaiwa Beach). Site F (350 meter from Waikane) is surrounded by parks (e.g., Sacred Falls and Kahana Valley State), the Ewa Forest Reserve, and the Moli'I pond. This site is far (distance) from urban Honolulu, has a narrow winding road (route 83), and located near culturally sensitive areas for native Hawaiians. Site G (in Kailua) has higher social costs than the City's target Sites B, C, and D because this site is located near high populated residential areas, which have higher income levels, population densities, and housing values.

Site F is ranked first by the mean transfer value for WTP method but fifth by all others. This can be explained by different methods used: (i) the mean transfer value for WTP approach based on CVM research, and (ii) the others (e.g., meta-analysis models and the direct mean transfer value approach) based on HPM studies. Theoretically, the marginal implicit price (MIP) based on HPM represents household's MWTP without housing market distortions (e.g., full information on housing prices and attributes, zero transaction costs and moving costs, and instantaneous price adjustment to changes in either demand or supply) (Freeman 2003). Although MWTP should be similar to WTP based on CVM, practically, the diversion can occur due to market distortions. For example, high demands for housing and/or high population densities on Oahu with the limited space available may incur this diversion.

The sensitivity analysis results indicate that income levels, distance from the landfill, discount rates, and lengths of landfill life affect the level of social costs. Meta-analysis (MA) models 1 and 2 have similar results in response to these changes. In terms of rank order, MA models 1 and 2 have consistent rank order for Sites A (ranked seventh), C (ranked first), and G (ranked sixth). The other sites (Sites B, D, E, and F) vary in ranking. This interchange may be attributed to some sites (Sites B, D, E, and F) having similar social costs. Sites B, D, and E (leeward side) are located close to each other, which have similar market conditions (e.g., income levels, housing values, and the number of households). Although Site F (near Waikane) is far away from these three sites, this site has similar income levels and housing values as Sites B, D, and E. Caution should be used that the rank order for MA model 1 is consistent as with different interest rates for lengths of life equal to 11, 25, and 50 years. MA model 2 has also similar results i.e., others are consistent with Sites D and E interchanged for the length of landfill life in 25 years.

The validity test results indicate that transfer values based on MA models 1 and 2 and original values were insignificantly different. The results of the reliability check indicate that MA models are preferred to the direct mean transfer for the MWTP approach, and MA model 2 is preferred to MA model 1. These findings are consistent with Rosenberger and Loomis (2003) and Rosenberger and Phipps (2007). Overall, in terms of sensitivity, validity, and reliability criteria, MA models perform better than the direct mean transfer value approach.

## 5.2. Policy implications

From findings of this study, the following policy issues can be drawn concerning: (i) the importance of examining social costs for proposed landfill sites, (ii) differences in social costs with different discount rates and the length of landfill life, (iii) differences in social costs due to market conditions (e.g., income levels, housing values, population densities, and the number of households), (iv) the significance of developing a method for landfill site selection, (v) the practical use of GIS, and (vi) the applicability of the meta-analysis for benefits transfer approach (BT). Policy implications and/ or recommendations are suggested as follows.

### 5.2.1. Social costs

The results indicate that external or social costs are substantially high when compared with the City's direct costs. Social costs range from $42 million to $1,299 million (annually $4.6 million to $143 million) while direct costs for a proposed landfill on Oahu range from about $15 million to $82 million ($0.87 million to $8.87 million per year) (for these costs, see Tables 3.5, 4.19, and 4.20). If planners fail to consider social costs, they will likely grossly underestimate the costs of the landfill and possibly locate the landfill in a higher overall cost location. Careful examination of social costs can help planners to locate a new landfill or allocate solid waste resources more efficiently.

### 5.2.2. Landfill capacity or length of landfill life in years

As the results indicate, landfill life lengths contribute to differences in social costs and/or direct costs. In terms of rank order, if each target site has the same length of life, the rank order is consistent between BT methods. However, if each target site has a different length of life, different results occur. The City misinformed residents by not considering this point. In order to compare sites or projects with unequal time lengths, planners need to examine the equivalent annual value (EAV) method, which transforms the present value to an annual basis (Boardman et al. 2006).

### 5.2.3. Market conditions

The results indicate that differences in social costs are affected by market conditions (e.g., income levels, population densities, housing values, and the number of households). One policy consideration is that higher income levels are positively related to higher marginal WTP or social costs. High income residents with higher income levels tend to be more sensitive to landfill location than low-income levels.[75] A proposed landfill site can discourage new investments (e.g., golf clubs, residential houses, condos, and resorts) to accommodate the demands by higher or affluent households. Thus, a regional development plan should be considered together with site selection. Another consideration is that areas with higher population densities incur higher social costs. In order to reduce these social costs, a proposed landfill should not be located near (within 3 mile) or in highly populated residential areas.

### 5.2.4. Availability of an integrative approach

The City's approach incorporates a preliminary screening phase (e.g., preliminary screening and ranking based on the landfill's length of life in years) and a further assessment phase to identify the Committee's preference for specific candidate sites (see Table 3.1). Unlike the City's approach, this study

[75] Residents with higher income levels may have greater substitution for landfill location or react to the proposed landfill or extension of the existing landfill than residents with lower income levels. For example, a group of homeowners living in the Ko Olina resort area filed a lawsuit against the City and County of Honolulu in regards to a proposal to expand the existing landfill in 2003. Compensation for affected residents and providing legal advice and information for low-income households may improve inequity related to landfill site selection.

provides an integrative approach that links a GIS-based method with an economic framework. The GIS-based method acts as a preliminary screening procedure that eliminates unsuitable areas from consideration as possible landfill sites and analyzes geographical characteristics (e.g., soil properties). The economic analysis can provide the basis for measuring the impacts of proposed landfill sites (social costs) for proposed target sites, which has advantages (e.g., replicability, transparency, and ease of comparison between target sites or other projects). The findings of this study suggest the applicability of the integrative approach for landfill site selection.

### 5.2.5. Practical use of a GIS

A geographic information system (GIS) can assist in data collection by identifying census tract data (e.g., income levels, population densities, the number of households, and housing values) within a 3-mile distance from each target site (for sensitivity analysis, data within 1, 3, and 5 mile distances respectively). Together with secondary data collection, the use of a GIS is helpful in analyzing geographical characteristics (e.g., soil properties). The use of a GIS also enables planners to process a large amount of data in a short time and reduce the effort required for information collection and processing.

### 5.2.6. Applicability of a meta-analysis for benefits transfer approach

The meta-analysis for benefits transfer (MA-BT) approach provides a possible measure that approximates the impacts of proposed landfill sites and ranks the selected target sites. The results indicate that meta-analysis models are preferred to the direct mean transfer value approach in terms of sensitivity, validity, and reliability criteria. This approach will enable planners to save time and reduce monetary budgets by utilizing information or empirical findings from other primary hedonic price method (HPM) studies.

## 5.3. Conclusion and discussion

As stated earlier, this study develops an integrative approach for landfill site selection that links a GIS analysis with an economic analysis. The basic framework minimizes social costs given constraints (exclusionary criteria) with a two-part process: (i) a GIS-based screening of possible landfill sites that satisfy constraints and (ii) benefits transfer (BT) methods (e.g., meta-analysis (MA) and mean transfer value approaches) that rank the remaining selected sites according to social cost minimization. Meta-analysis models were evaluated in terms of sensitivity, validity, and reliability criteria.

The GIS-based analysis was applied to: (i) the City's 45 potential sites (Scenario 1) and (ii) the entire island of Oahu (Scenario 2). Together, both scenarios found a total of 7 sites that satisfied the given criteria. Benefits transfer (BT) methods (e.g., meta-analysis and mean transfer value approaches) were utilized to measure social costs and to rank the selected 7 target sites. The results clearly demonstrate that social costs are substantially high compared with the City's direct costs and vary by target sites due to different market conditions, different methods used, different models used, and differences in discount rates and lengths of landfill life in years. In terms of sensitivity, reliability, and validity criteria, meta-analysis models are preferred to the direct mean transfer value approach.

This study will enable planners and residents to better understand the process of landfill site selection and anticipated impacts of proposed landfill sites. Unlike the City's approach, this study provides the integrative method by linking a GIS analysis with an economic analysis. The GIS-based method acts as a preliminary screening procedure and helps data collection by census tract data. Economic analysis provides a theoretical framework (e.g., social cost minimization given exclusionary criteria) and a basis for measuring the impacts of proposed landfill sites (social costs), which has advantages (e.g., replicability, transparency, and ease of comparison between target sites or other projects).

This method also saves time and reduces a monetary budget. The integrative approach provides a potential method for site selection and can be applied to select other sites (e.g., hazardous waste sites and incinerators). In subsequent sections, limits of the study, contributions of this study, and issues raised are discussed.

### 5.3.1. Study limitations

In reality, the above results were obtained given certain data limitations. One should note that GIS-based analysis can produce some error from digitization, map scale, and temporal changes. For ensuring improved accuracy, field validation and checking with other research results are recommended. A periodic upgrade of the GIS data is also recommended to capture temporal changes in geographic characteristics.

This study is concerned with a small sample size (9 studies providing 22 observations) of meta-analysis. Given the small sample size, this study utilized two meta-analysis (MA) models with reasonable diagnostic results (e.g., normality, heteroskedasticity, multicollinearity, and model specification) in performing benefits transfer (BT). The results were reasonable in terms of sensitivity, validity, and reliability criteria. The lack of data (reliable data) limits the scope of the meta-analysis. Without the addition of new studies, the ability to develop robust and valid benefit transfer functions may be limited. Support and funding to conduct primary research should be continued (Rosenberger and Phipps 2007).

One should use caution in discussing the reliability of benefits transfer (BT) because there are no guidelines to determine the acceptable level of transfer errors. Primary hedonic price method (HPM) studies were selected for the meta-analysis for benefits transfer approach following the standard for valid and reliable benefits transfer (Bergstrom and Taylor 2006; Smith et al. 2002): (i) welfare measure consistency (e.g., hedonic price methods), (ii) commodity consistency (e.g., distance from municipal solid waste landfills), and (iii) theory consistency (e.g., positive distant effects). The transfer errors are relatively reasonable compared with other studies (for summary of studies to conduct benefits transfer reliability checks, see Rosenberger and Loomis 2007). Despite probable transfer errors from benefits transfer (BT), policy-makers generally accept the use of benefits transfer (BT) as an appropriate method for gaining more knowledge about WTP in order to avoid negative externalities or disamenities from target sites or providing an initial assessment of ranking or screening sites (for the stylized cases involving the level of BT accuracy, see Brookshire 1992). Together with the acceptable level of accuracy, planners should consider the use of benefits transfer (BT).

### 5.3.2. Contributions

In terms of contributions to the frontier of knowledge, this research stretches over the following fields: GIS, natural resources and environmental management, resource and environmental economics, and related subject areas.

This study contributes to the frontier of knowledge by developing an integrative approach linking a GIS-based analysis with a meta-analysis for benefits transfer (MA-BT) approach. Although GIS has been widely used for landfill site selection (Siddiqui et al. 1996; Baban and Flannagan 1998; Kontos et al. 2003, 2005; and Daneshvar et al. 2005, Sener et al. 2006; Hasan et al. 2009), there is a tendency to disregard economic analysis.

A few studies (Swallow et al. 1992; Opaluch et al. 1993) utilized a GIS analysis with an economic framework for landfill site selection. These studies employed a GIS for data collection and a direct survey method to evaluate public preferences for landfill site selection. However, the direct survey method requires substantial time and a large monetary budget. Although few studies (Brisson and Pearce 1995; Ready 2005; Walton et al. 2006) have utilized meta-analysis (MA) to examine factors that affect different impacts of landfills on housing values, they did not employ meta-analysis for landfill site selection. To the author's knowledge, this is the first study that links a GIS analysis with a meta-analysis for benefits transfer (MA-BT) approach to analyze landfill site selection.

This research is also the first formal study on landfill site selection for the island of Oahu which: (i) compared study results with those of a recent City and County of Honolulu study, (ii) evaluated the process of landfill site selection, and (iii) checked the results in terms of sensitivity, reliability, and validity criteria. The important message is that social costs are substantial and vary by target sites according to market conditions, primary valuation methods used, meta-analysis models used, and the length of landfill life.

### 5.3.3. Discussion

A few issues are raised involving externalities, economically optimal costs, scarcity of landfill capacity, and equity and participation. These issues are important in terms of economic analysis. Further research improves the analysis by: (i) addressing the issues that this study did not examine and (ii) expanding the scope of economic analysis.

#### Externalities

This study focuses on cost-effectiveness (e.g., social cost minimization given constraints). Given clear ownership, no externalities, and the existence of viable markets, a competitive market process establishes an economically efficient solid waste resource allocation (Moncur and Pollock 1988). However, planners find difficulty in managing external effects from a proposed landfill with no market. Although this study measures the impact of landfills on housing values (i.e., marginal WTP for distance) or WTP to avoid the landfill, landfills induce various externalities (e.g., air pollution, land pollution, water pollution, and sedimentation and global climate change). See Chapter 2 of this study and other studies such as Methodex 2007 and Porter 2002, which have examined external effects of landfills. Future research is needed to further analyze these external effects.

### Economically optimal costs

It should be noted that financial costs used in most projects differ from economically optimal costs derived by optimization procedures. Malarin and Vaughan (1997) measured private costs from constrained cost minimization. Social costs incorporate private costs and external costs, and this study implicitly assumes that private costs are given (for the measurement of costs related to landfill sites, see Hirshfeld et al. 1992; Repetto 1992). Future research should analyze: (i) a comparison of financial private costs and optimal private costs and (ii) a comparison of social costs and external costs plus financial private costs or private costs. These comparisons will indicate whether the proposed landfill is being implemented cost-effectively. This information is important because planners can anticipate the approximate costs of the landfill.

### Scarcity of landfill capacity

This research considers physical landfill capacity as an important exclusionary criterion and utilizes the length of landfill life in order to measure social costs. However, this study did not analyze scarcity of landfill capacity in terms of economic analysis. Optimizing over a useful life period requires a dynamic model that employs optimal depletion theory. Given scarce landfill capacity, the optimization problem is to choose the waste disposal rates (extraction rates of the capacity) and alternative methods (e.g., the introduction of a new landfill, enhanced recycling, and transshipment off-island) in order to maximize net benefits of the residents on Oahu (for the application of optimal control theory to landfills, see Ready and Ready 1995; Highfill and McAsey 2001).

Future research is needed to compare costs between using the existing WGSL landfill and utilizing other methods including a new landfill site, enhanced recycling, and transshipment off-island.

Environmental justice or equity and participation

This study focuses on cost-effectiveness but does not resolve equity issues. While benefits provided by a new landfill site are shared by all citizens, harm from a landfill (e.g., reduction in housing values) concentrate on nearby residents i.e., unequal distribution of wealth. In terms of the Kaldor-Hicks criterion, if winners from a new landfill site compensate losers and there is a still net gain, a new landfill site may improve the social welfare. The use of compensation packages can assist in gaining landfill approval and improving inequality (for economic approaches that discuss economic efficiency and equity, see Kunreuther and Kleindorfer 1986; Mitchell and Carson 1986). [76] Although monetary payments are theoretically preferred to payment in-kinds, Kunreuther and Easterling (1996) found that monetary payments are viewed as less desirable than in-kind compensation (e.g., state money for schools and road improvements), free garbage collection, reimbursement for medical costs, and property value guarantees. Research is needed to analyze the level of compensation and the structure of incentive that can reduce strong opposition.

Participation is important in the process of landfill site selection because lack of participation can incur strong opposition to a proposed landfill and delay the process or increase costs. When various interest groups involved in the process of landfill site selection participate, this project is more likely to be successful (Pearce et al. 2006). Environmental and social equity is becoming an important consideration in evaluating resource use. All relevant policy appraisal

[76] It is assumed that a landfill should be located at a certain target site since social benefits obtained by the landfill are greater than social costs. For example, despite enhancement of alternative methods (e.g., source reduction, recycling, and incinerators), the landfill will be an essential component in the City's waste management plan (City and County of Honolulu 2008a).

processes (e.g., environmental impact statements) require addressing this issue. Environmental and social justice planners should incorporate minority populations (e.g., race, color, and gender) and lower-income communities in the site selection process and provide clear information related to the process of site selection (see O'Hare et al. 1983; Kreith and George 2002). Further research is needed to analyze public preferences incorporating various individuals and groups and/or preferences of minority populations.

## APPENDIX A. Hedonic Studies Used in Meta-Analysis and Data


Bouvier, Rachel A., J Halstead, K Conway, and A Manalo. “The Effect of Landfills on Rural Residential Property Values: Some Empirical Evidence.” Regional Analysis and Policy. 30 (2): 23-37. 2000.

Kiel, Katherine A., and Katherine T. McClain. 1995b. "House Prices during Siting Decision Stages: The Case of an Incinerator from Rumor through Operation." *Journal of Environmental Economics and Management*. 28 (2): 241-55.

Lim, J. S., and P. Missios. 2007. "Does Size Really Matter? Landfill Scale Impacts on Property Values." *Applied Economics Letters*. 14 (10): 719-723.

Nelson, Arthur C, John Genereux, and M Michelle Genereux. 1992a. “Price Effects of Landfills on House Values.” *Land Economics.* 68 (4): 359-65.

Nelson, Arthur C, Member, ASCE, John Genereux, and M Michelle Genereux. 1992b. “Price Effects of Landfills on Residential Land Values.” *Journal of Urban Planning and Development.* 118 (4): 128-137.

Nelson, Arthur C, John Genereux, and M Michelle Genereux. 1998. "Price Effects of Landfills on Different House Value Strata." *Journal of Urban Planning and Development.* 123 (3): 59-67.

Ready, Richard C. 2005. *Do Landfills Always Depress Nearby Property Values? Issues at Hand*. NERCRD Regional Rural Development Paper. 27. University Park, PA: Northeast Regional Center for Rural Development.

Reichert, Alan K, Michael Small, and Sunil Mohanty. 1991. "The Impact of Landfills on Residential Property Values.” *Journal of Real Estate Research.* 7 (3): 297-314.

Thayer, Mark, Heidi Albers, and Morteza Rahmatian. 1992. "The Benefits of Reducing Exposure to Waste Disposal Sites: A Hedonic Housing Value Approach." *Journal of Real Estate Research*. 7(3): 265-282.

Table A.1. Data for Meta-Analysis

| Study | MWTP | Pop | Y | MSW | Active | N | SE | Function |
|---|---|---|---|---|---|---|---|---|
| Nelson et al.(1992a) | 6.2 | 430 | 64020 | 1 | 1 | 708 | 1.47 | 1 |
| Nelson et al. (1992b) | 28.44 | 188.9 | 64020 | 1 | 1 | 255 | 4.27 | 0 |
| | 2.46 | 368.2 | 63498 | 1 | 0 | 140 | 1.72 | 0 |
| | 5.88 | 258.9 | 60849 | 1 | 0 | 126 | 1.88 | 0 |
| Nelson et al.(1997) | 2.64 | 1213 | 73540 | 1 | 1 | 436 | 1.11 | 1 |
| | 4.32 | 1213 | 73540 | 1 | 1 | 143 | 1.19 | 1 |
| | 8.43 | 1213 | 73540 | 1 | 1 | 65 | 3.21 | 1 |
| Ready (2005) | 10.86 | 3646.2 | 56103 | 1 | 1 | 11069 | 1.42 | 0 |
| | 7.21 | 4941.7 | 65061 | 1 | 1 | 11069 | 1.78 | 0 |
| Lim and Missios (2003) | 3.65 | 1055 | 62483 | 1 | 1 | 331 | 1.83 | 0 |
| | 2.21 | 4159 | 65998 | 1 | 1 | 1139 | 0.96 | 0 |
| Bouvier et al. (2000) | 6.27 | 446.8 | 61786 | 1 | 0 | 59 | 3.54 | 0 |
| | 2.8 | 1498.5 | 62757 | 1 | 1 | 47 | 4.86 | 0 |
| | 0.65 | 77.4 | 53976 | 1 | 0 | 101 | 2.71 | 0 |
| | 0.99 | 2320 | 47341 | 1 | 0 | 59 | 2.07 | 0 |
| Reichert et al. (1992) | 6.4 | 756.23 | 66142 | 1 | 1 | 1631.05 | 2.061 | 1 |
| | 4 | 6283.3 | 37242 | 1 | 1 | 1631.05 | 2.061 | 1 |
| Kiel and McClain (1995) | 1.7 | 746.7 | 71785 | 0 | 1 | 662 | 0.95 | 0 |
| | 3.2 | 804.17 | 71785 | 0 | 1 | 711 | 0.97 | 0 |
| | 2.7 | 855 | 71785 | 0 | 1 | 855 | 0.89 | 0 |
| Thayer et al. (1992) | 1.6 | 1623.8 | 35438 | 0.85 | 0.75 | 2323 | 2.195 | 0 |
| | 0.3 | 1623.8 | 35438 | 0.85 | 0.75 | 2323 | 2.195 | 0 |

## APPENDIX B. Theoretical Basis for Measuring Social Costs

While benefits provided by a new landfill site are shared by all citizens, harm from a landfill (e.g., reduction in housing values) concentrate on nearby residents (Kunreuther and Kliendorfer 1986; Mitchell and Carson 1986). In order to measure the community's social costs for each target site, the City can measure willingness to pay (WTP) to restrict the proposed landfill nearby his or her residence because environmental policy usually focuses on improvements in environmental quality rather than deliberate degradation of the environment (Pearce et al. 2006).

In order to obtain the resident's WTP for avoiding the landfill, the City considers the hypothetical situation: (i) pre-policy (with a landfill); and (ii) post-policy (without a landfill). Following Mitchell and Carson (1989), it is assumed that planners have a policy option to restrict the landfill on the target site, and the residents have a property right only to the initial situation $q^0$ (the level of environmental quality with the landfill). Thus, the residents do not have the property right to the benefits by the post-policy that improves environmental quality.

The value that a resident at the target site places on environmental quality (q) with the landfill is (for a theoretical background, see Roberts et al. 1991):

$$\mathrm{u}^0 = \mathrm{u}^0\ (\mathrm{x}, \mathrm{q}^0) \qquad \text{(B-1)}$$

where $u^0$ is the level of utility from which a change in welfare is measured, x is a vector of quantities of private goods, and $q^0$ is the level of environmental quality with a landfill.

The resident's level of utility without the landfill is given by:

$$\mathrm{u}^1 = \mathrm{u}^1\ (\mathrm{x}, \mathrm{q}^1) \qquad \text{(B-2)}$$

where $\mathrm{u}^1$ is the level of utility without the landfill, x is a vector of quantities of private goods, and $q^1$ is the level of environmental quality without the landfill.

Because the landfill site produces negative external effects, $\mathrm{u}^0 \equiv \mathrm{v}\,(\mathrm{p}, \mathrm{q}^0, \mathrm{y})$ is less than $\mathrm{u}^1 \equiv \mathrm{v}\,(\mathrm{p},\ \mathrm{q}^1,\ \mathrm{y})$.

In order to value this improvement in environmental quality ($q^0$ to $q^1$) (i.e., avoiding external effects from the landfill), the associated dual problem minimizes total consumer expenditures needed to maintain a given level of utility. The problem with the landfill is (B-3), and the problem without the landfill is (B-4):

Minimize $\sum_{\mathrm{j}=1}^{\mathrm{n}} \mathrm{P_j}\,\mathrm{X_j}$ subject to $\mathrm{U}^0 = \mathrm{U}^0\,(\mathrm{X}, \mathrm{Q}^0)$ (B-3)

Minimize $\sum_{\mathrm{j}=1}^{\mathrm{n}} \mathrm{P_j}\,\mathrm{X_j}$ subject to $\mathrm{U}^1 = \mathrm{U}^1\,(\mathrm{X}, \mathrm{Q}^1)$ (B-4)

where $P_j$ is the price vector of private good j, $X_j$ is a vector of quantities of private good j, and $\sum_{j=1}^{n} P_j\,X_j$ is the total consumer expenditures given utility with the landfill ($U^0$) or without landfill ($U^1$).

The solution to these problems defines the expenditure functions ($e^0$ and $e^1$) shown in equations (B-5) and (B-6):

$\mathrm{e}^0 = \mathrm{e}\left(\mathrm{P_j}, \mathrm{Q}^0, \mathrm{U}^0\right) = \mathrm{M}^0$ (B-5)

$\mathrm{e}^1 = \mathrm{e}\left(\mathrm{P_j}, \mathrm{Q}^1, \mathrm{U}^0\right) = \mathrm{M}^1$ (B-6)

where $\mathrm{M}^0$ and $\mathrm{M}^1$ are the consumer's income levels before and after the policy decision, holding utility U at $\mathrm{U}^0$. Thus, the decrease in income required to maintain the resident's level of utility ($\mathrm{U}^0$) from an increase in environmental quality ($\mathrm{Q}^0$ to $\mathrm{Q}^1$) can be defined as:

$\mathrm{WTP_i} = \mathrm{e}\,(\mathrm{p}, \mathrm{q}^0,\ \mathrm{u}^0) - \mathrm{e}\left(\mathrm{p_j}, \mathrm{q}^1, \mathrm{u}^0\right) = \mathrm{M}^0 - \mathrm{M}^1$ (B-7)

The differences of the income level represent the resident's maximum $WTP_i$ in order to move away or restrict the proposed landfill from the target sites, which is considered as an external cost to a resident with the landfill.

The aggregate value of the community's annual WTP at time t for each target can be expressed as the sum of the individual's $WTP_i$ (Roberts et al. 1991; Pearce et al. 2006):

Aggregate WTP = $\sum_{\mathrm{i}}^{\mathrm{n}} \mathrm{WTP_i}$, i = 1, 2, …, n (B-8) where $WTP_i$ is the ith resident WTP, and n is the number of households for the target site. Aggregate WTP represent the social costs for the target sites at the specific time t.

The present value (PV) of the stream of annual social costs is (Pearce et al. 2006)

$$\mathrm{PV} = \sum_{\mathrm{i}=0}^{\mathrm{T}} \frac{\text{Aggregate WTP(t)}}{(1+\mathrm{i})^{\mathrm{t}}} \qquad \text{(B-9)}$$

where aggregate WTP (t) represent the annual social costs at time t for the target site, i is the discount rate (a real interest rate adjusted for inflation), and T is the number of years. Discounting is a controversial issue: a constant discount rate, a zero discount rate, and a time declining rate (Freeman 2003; Pearce et al. 2006). For this study, a constant discount rate (3%) is utilized. For sensitivity analysis, a 7% interest rate is also considered. The federal agencies mandated the use of 7% discount rate, but the U.S. EPA recommended the 2-3% rate for economic analyses of its regulation (Freeman 2003). The U.S. EPA's recommendation is similar to current financial markets. For example, the ten-year historic federal average prime rate (5.8%) and the Honolulu CPI (ten-year average percentage change, 2.8%) indicates a 3 % real interest rate.

Equation (B-9) is the sum of T terms of a geometric series with the common ratio equal to 1 / (1+i). Thus, present value of an annuity (the current value of a stream of equal payments over a specified period of time) is (Boardman et al. 2006):

$$\mathrm{PV} = \mathrm{A}\left[\frac{1-(1+\mathrm{i})^{-\mathrm{t}}}{\mathrm{i}}\right] = \mathrm{A}\mathrm{a}_{\mathrm{i}}^{\mathrm{T}} \qquad \text{(B-10)}$$

where A is the annual social costs at time t, and $a_i^T$ is an annuity factor where $\mathrm{a}_{\mathrm{i}}^{\mathrm{T}}$= [1-$(1+\mathrm{i})^{-\mathrm{T}}$] / i. The variable i is the discount rate (a real interest rate adjusted for inflation), and T is the number of years.

A common way to compare projects with unequal time length is the equivalent annual value (EAV) method. This method transforms the present value to an annual basis, which can be compared with other projects. The formula of the EAV is as follows (Boardman et al. 2006):

$$\text{EAV} = {}^{\text{PV}}\!/_{a_i^T} \tag{B-11}$$

# APPENDIX C. Theory and Practice: Meta-Analysis for Benefits Transfer

Meta-analysis for benefits transfer (MA-BT) attempts to model an underlying valuation function which accounts for the characteristics of policy or target sites. This study adopted the framework of MA-BT suggested by Rosenberger and Phipps (2007). The valuation function is an envelope function for individual site specific valuation functions estimated from different studies, where marginal WTP for distance is a function of the characteristics of the site (e.g., income and population density). Based on an underlying utility theoretic model, Bergstrom and Taylor (2006) presented three approaches to meta-analysis: (i) the strong structural utility theoretic (SSUT) approach; (ii) the non-structure theoretic utility approach (NSUT); and (iii) the weak structural utility theoretic approach (WSUT).

The individual utility maximization problem is to maximize a strictly concave utility function subject to the compact (closed and bounded) budget set. An individual's utility for each target site has the following utility form (Freeman 2003):

$$\text{maximize}_{x,q}\ u(x, q) \quad \text{subject to } y = px + WTP\ (q) \qquad \text{(C-1)}$$

where u is individual's utility, x is a vector of private goods quantities, and q is the level of environmental quality. The variable y is income, p is the price vector of private goods, and WTP is the individual's willingness to pay (WTP) for avoiding negative external effects from the landfill, which is the function of the environmental quality (q).

This individual's utility maximization yields a set of conditional demand functions for marketed goods i.e., x= x (p, y - WTP (q); q). Freeman (2003) referred to these functions as the conditional demand functions since the functions are conditioned upon the imposed environmental quality (q). Inserting the conditional demand functions into the utility function provides the conditional indirect utility function. The form of the underlying conditional indirect utility function is (Bergstrom and Taylor 2006):

v = $\mathrm{v_i}$ ($\mathrm{p_i}$, $\mathrm{y_i}$; q) (C-2)

where $P_i$ is prices of market goods faced by individual i, $y_i$ is the ith household income, and $q_i$ is the level of environmental quality available to the ith individual household.

The individual's indirect function with the landfill ($v^0$) is

$\mathrm{v^0 = v_i^0(p_i^0,\ y_i - WTP\ (q_i),\ q_i^0)}$ (C-3)

The individual's indirect function without the landfill ($v^1$) is

$\mathrm{v^1 = v_i^1(P_i^1,\ y_i - WTP,\ q_i)}$ (C-4)

The difference between (C-4) and (C-3) represents an increase in welfare obtained by avoiding negative external effects from the landfill, which can be defined by:

Δu = $\mathrm{v^1}$- $\mathrm{v^0}$= $\mathrm{v^1(p_i^1,\ y_i - WTP,\ q_i^1)}$ - $\mathrm{v^0\ (p_i^0,\ y_i - WTP,\ \ q_i^0)}$ > 0 (C-5)

Solving (C-5) provides WTP results in the general bid function:

WTP= f ($\mathrm{p_i^1}$- $\mathrm{p_i^0}$, $\mathrm{q_i^1 - q_i^0}$) (C-6)

The general WTP can be affected by the change in prices of market goods, the change in environmental quality, and other factors such as non-income household characteristics.

Bergstrom and Taylor (2006) referred to the strong structural utility theoretic (SSUT) approach which involves specifying a structural form of the indirect utility function such as (C-2). The particular form of (C-6) that is derived from (C-2) to (C-5) can be used for BT. The non-structural utility theoretic (NSUT) approach is at the other end of the spectrum on the utility theoretic approach. The NSUT approach does not explicitly specify the connection between variables and an underlying utility function (see Woodward and Wui 2001). The weak structural utility theoretic (WSTU) approach is in between the SSUT and NSUT approaches. The WSTU approach approximately specifies the connection between explanatory variables and an underlying utility function. In the WSUT approach, study design variables can be included in order to explain different WTP or marginal WTP estimates that occur from different research designs. They suggested utilizing the SSUT and WSUT approaches for benefits transfer. The NSUT approach is not suitable for BT.

Following Bergstrom and Taylor (2006), this study utilizes the WSUT approach for BT (see Smith at al. 2002; Smith and Huang 1995; Walton et al. 2006). While core economic variables (e.g., Y and POP) based on economic theory are key factors for benefits transfer reflecting differences between target sites, study design variables (e.g., N, SE, and FUNCTION) can improve the meta-analysis function for BT by explaining different MWTP.

# REFERENCES


Agency for Toxic Substances & Disease Registry (ATSDR). 2001. *Landfill Gas Primer- An Overview for Environmental Health Professionals*. Atlanta, GA. http://www.atsdr.cdc.gov/hac/landfill/html/intro.html [accessed July 7, 2010]

Bateman, Ian, Andrew A. Lovett, and Julii S. Brainard. 2005. *Applied Environmental Economics: A GIS Approach to Cost-Benefit Analysis*. Cambridge, UK: Cambridge University Press.

Baban, Serwan, and Joseph Flannagan. 1998. "Developing and Implementing GIS-Assisted Constraints Criteria for Planning Landfill Sites in the UK." *Planning Practice & Research*. 13 (2): 139-152.

Bagchi, Amalendu. 1994. *Design, Construction, and Monitoring of Landfills*. New York, N.Y.: J. Wiley and Sons.

Banzhaf, H. Spencer. 2008. *Environmental Justice: Opportunities through Markets*. Bozeman, MT: PERC.

Barrow, Christopher J. 1991. *Land Degradation.* Cambridge, England: Cambridge University Press.

Bergh, J. C., Kenneth J. Button, Peter Nijkamp, and G. C. Pepping. 1997. *Meta-Analysis in Environmental Economics*. Boston: Kluwer Academic Publishers.

Bergland O., K. Magnussen, and S. Navrud. 2002. Benefit Transfer: Testing for Accuracy and Reliability. In *Comparative Environmental Economic Assessment,* ed. Florax, R.J. G. M., Peter Nijkamp, and K.G. Willis. Cheltenham, UK: E. Elgar Publishing.

Bergstrom, John C., and Laura O. Taylor. 2006. "Using Meta-Analysis for Benefits Transfer: Theory and Practice." *Ecological Economics*. 60 (2): 351-360.

Bingemer, H. G., and P. J. Crutzen. 1987. “The Production of Methane from Solid Wastes.” *Journal of Geophysical Research*. 92. 2182-2187.

Boardman, Anthony E., David H. Greenberg, Aidan R. Vining, and David L. Weimer. 2006. *Cost-Benefit Analysis: Concepts and Practice*. Upper Saddle River, N.J.: Pearson/Prentice Hall.

Bouvier, Rachel A., John M. Halstead, Karen S. Conway, and Albert B. Manalo. 2000. “The Effect of Landfills on Rural Residential Property Values: Some Empirical Evidence.” *Regional Analysis and Policy*. 30 (2): 23-37. 2000.

Bowen, William. 2002. "An Analytical Review of Environmental Justice Research: What Do We Really Know?" *Environmental Management.* 29 (1): 3-15.

Boyle, Kevin J. 2003. Contingent Valuation in Practice. In *A Primer on Nonmarket Valuation,* ed. Champ, Patricia A., Kevin J. Boyle, and Thomas C. Brown. Boston: Kluwer Academic Publishers.

Boyle, Melissa A., and Katherine A. Kiel. 2001. “A Survey of House Price Hedonic Studies of the Impact of Environmental Externalities.” *Journal of Real Estate Literature.* 9 (2): 117-144.

Brander, Luke, Raymond Florax, and Jan Vermaat. 2006. "The Empirics of Wetland Valuation: A Comprehensive Summary and a Meta-Analysis of the Literature." *Environmental and Resource Economics.* 33 (2): 223-250.

Brisson, Inger, and David W. Pearce. 1995. *Benefits Transfer for Disamenity from Waste Disposal*. London in England: Centre for Social and Economic Research on the Global Environment, University College London and University of East Anglia.

Brookshire, David S. 1992. *Issues Regarding Benefits Transfer*. Paper Presented at the Association of Environmental and Resource Economists Workshop. Utah.

Brouwer, Roy, and Spaninks F. A. 1999. “The Validity of Environmental Benefits Transfer: Further Empirical Testing.” *Environmental and Resource Economics*. 14: 95-117.

Carson, Richard T., Nicholas E. Flores, Kerry M. Martin, and Jennifer L. Wright. 1996. "Contingent Valuation and Revealed Preference Methodologies: Comparing the Estimates for Quasi-Public Goods." *Land Economics.* 72 (1): 80-99.

Carson, Richard T., and Michael W. Hanemann. 2005. Contingent Valuation. In *Handbook of Environmental Economics 2,* ed. Mäler, Karl-Göran, and Jeffrey R. Vincent. Amsterdam, Germany: Eisevier.

Champ, Patricia A., Kevin J. Boyle, and Thomas C. Brown. 2003. *A Primer on Nonmarket Valuation*. Boston: Kluwer Academic Publishers.

Church, F. M., and F. E. Shepherd. 1989. “The Ozone Hole and the Greenhouse Effect.” *Gas Engineering and Management*. 29: 282-284.

City & County of Honolulu. 2002. *Final Supplemental Environmental Impact Statement for the Waimanalo Gulch Sanitary Landfill Expansion*. Honolulu, HI.

______.2003. *Report of the Mayor’s Advisory Committee on Landfill Siting Selection: (Final Report).* Honolulu, HI.

______. 2004. *Solid Waste Integrated Management Plan*. Honolulu, HI.

______.2006. *Environmental Impact Statement Preparation Notice for the Waimanalo Gulch Sanitary Landfill Expansion*. Honolulu, HI.

______. 2008a. *Integrated Solid Waste Management Plan*. Honolulu, HI.

______.2008b. *Final Environmental Impact Statement: Waimanalo Gulch Sanitary Lateral Expansion*. Honolulu, Hi.

Daneshvar, Roozbeh. 2005. “Customizing ArcMap Interface to Generate a User-friendly Landfill Site Selection GIS Tool.” *Environmental Information Archives*. 1: 428-437.

Davis, Bruce E. 2001. *GIS: A Visual Approach.* Albany, N.Y.: Thomson Learning.

Desvousges, William H., Reed F. Johnson, and Spencer H. Banzhaf. 1998. *Environmental Policy Analysis with Limited Information: Principles and Applications of the Transfer Method*. MA: Edward Elgar.

Diamond, Peter A., and Jerry A. Hausman. 1994. “Contingent Valuation: Is Some Number Better Than No Number?” *Journal of Economic Perspectives*. 8(4): 45-64.

Downing, Mark, and Teofilo Ozuna Jr. 1996. "Testing the Reliability of the Benefit Function Transfer Approach." *Journal of Environmental Economics and Management.* 30 (3): 316-322.

Eggertsson, Thrainn. 1990. *Economic Behavior and Institutions,* Cambridge: Cambridge University Press.

El-Fadel, M., A. N. Findikakis, and J. O. Leckie. 1997. "Environmental Impacts of Solid Waste Landfilling.” *Journal of Environmental Management*. 50 (1): 1-26.

Eshet, Tzipi, M. Baron, and M. Shechter. 2005. "A Critical Review of Economic Valuation Studies of Externalities from Incineration and Landfilling." *Waste Management & Research: the Journal of the International Solid Wastes and Public Cleansing Association, ISWA.* 23 (6): 487-504.

_____. 2007. "Exploring Benefit Transfer: Disamenities of Waste Transfer Stations.” *Environmental & Resource Economics.* 37 (3): 521-547.

Farber, S. "Undesirable Facilities and Property Values: A Summary of Empirical Studies." *Ecological Economics: the Journal of the International Society for Ecological Economics.* 24 (1):1-14.1998.

Freeman, A. Myrick. 1979. “Hedonic Prices, Property Values, and Measuring Environmental Benefits a Survey of the Issues.*” Scandinavian Journal of Economics*. 81 (2): 154-173.

______. 2002. *The Measurement of Environmental and Resource Values Theory and Methods.* Washington, D.C.: Resources for the Future.

Gamble, Hays Bentley. 1982. *Effects of Solid Waste Disposal Sites on Community Development and Residential Property Values*. Institute for

Research on Land & Water Resources, the Pennsylvania State University.

Garrod, Guy and K. Willis. 1998. “Estimated Lost Amenity due to Landfill Disposal.” *Resources, Conservation and Recycling.* 22(1): 83-95.

Gayer, Ted, and Kip W. Viscusi. 2002. "Housing Price Responses to Newspaper Publicity of Hazardous Waste Sites." *Resource and Energy Economics* 24 (1): 33-51.

Greene, William H. 2003. *Econometric Analysis*. Upper Saddle River, N.J.: Prentice Hall.

Guam EPA. 2004. *Preliminary Landfill Site Suitability Report*. Guam: Department Public Works.

Hamilton, James T. 1995. "Testing for Environmental Racism: Prejudice, Profits, Political Power?" *Journal of Policy Analysis and Management.* 14 (1): 107-132.

Hamilton, Lawrenc C. 2004. *Statistics with Stata.* Belmont, CA: Thomson/Brooks Cole.

Hasan, Mohammad R., K. Tetsuo, and Syed A. Islam. 2009. “Landfill Demand and Allocation for Municipal solid Waste Disposal in Dhaka City-An Assessment in a GIS Environment.” *Journal of Civil Engineering*. 37 (2): 133-149.

Hedges, Larry V., and Ingram Olkin. 1985. *Statistical Methods for Meta-analysis*. Orlando: Academic Press.

Hester, Ronald E. 1995. *Waster Treatment and Disposal*. Cambridge, England: Royal Society of Chemistry.

Heywood, D. Ian, Sarah Cornelius, and Steve Carver. 2002. *An Introduction to Geographical Information Systems.* New York: Addison Wesley Longman.

Higgs, Gary. 2006. “Integrating Multi-Criteria Techniques with Geographical Information Systems in Waste Facility Location to Enhance Public Participation.” *Waste Management & Research*: *the Journal of the*

*International Solid Wastes and Public Cleansing Association*, ISWA. 24 (2):105-17.

Highfill, Jannett, and Michael McAsey. 2001. "Landfilling Versus Backstop Recycling When Income Is Growing." *Environmental and Resource Economics*. 19(1): 37-52.

Hirshfeld, S., P. A. Vesilind, and E. I. Pas. 1992. "Assessing the True Cost of Landfills." *Waste Management and Research.* 10 (6): 471.

Hite, Diane, Wen Chern, Fred Hitzhusen, and Alan Randall. 2001. "Property-Value Impacts of an Environmental Disamenity: The Case of Landfills." *The Journal of Real Estate Finance and Economics.* 22 (2): 185-202.

Johnston, Jack, and John Dinardo 1997. *Econometric Methods*. Economics Series. Singapore: McGraw-Hill.

Kao, Jehng-Jung, Wei-Yea Chen, Hung-Yue Lin, and Show-Jyi Guo. 1996. "Network Expert Geographic Information System for Landfill Siting." *Journal of Computing in Civil Engineering.* 10 (4): 307-318.

Kasturi, Prahlad, and Thomas A. Loudat. 2004. "Socio-Economic and Environmental Impacts of Landfill Sites on Oahu, Hawaii." *Journal of Environmental Systems*. 31 (4): 289-313.

Kiel, Katherine A. 2006. *Environmental Contamination and House Values*. Worcester, MA: Dept. of Economics College of the Holy Cross. http://academics.holycross.edu/economics-accounting/papers [accessed July 8, 2010].

Kiel, Katherine A., and Jeffrey E. Zabel. 2001. "Estimating the Economic Benefits of Cleaning Up Superfund Sites: The Case of Woburn, Massachusetts." *Journal of Real Estate Finance and Economics.* 22 (2/3): 163-201.

Kiel, Katherine A., and Katherine T. McClain. 1995a. "Measuring the Impact of the Discovery and Cleaning of Identified Hazardous Waste Sties on House Values." *Land Economics*. 71: 428-435.

______. 1995b. "House Prices during Siting Decision Stages: The Case of an Incinerator from Rumor through Operation." *Journal of Environmental Economics and Management*. 28 (2): 241-55.

Kiel, Katherine, and Michael Williams. 2007. "The Impact of Superfund Sites on Local Property Values: Are All Sites the Same?" *Journal of Urban Economics.* 61 (1): 170-192.

Kirchhoff, Stefanie, Bonnie Colby, and Jeffrey LaFrance. 1997. "Evaluating the Performance of Benefit Transfer: An Empirical Inquiry." *Journal of Environmental Economics and Management.* 33 (1): 75-93.

Kjeldsen, Peter, Morton A. Barlaz, Alix P. Rooker, Anders Baun, Anna Ledin, and Thomas Christensen. 2002. "Present and Long-Term Composition of MSW Landfill Leachate: A Review". *Critical Reviews in Environmental Science and Technology.* 32 (4): 297-336.

Kohlhase, Janet. E. 1991. “The Impact of Toxic Waste Sites on Housing Values.” *Journal of Urban Economics.* 30 (1): 1-26.

Kontos, Themistoklis D., Dimitrios P. Komilis, and Constantinos P. Halvadakis. 2003. "Siting MSW Landfills on Lesvos Island with a GIS-based Methodology." *Waste Management & Research.* 21: 262-277.

______. 2005. "Siting MSW Landfills with a Spatial Multiple Criteria Analysis Methodology." *Waste Management.* 25.8: 818-832.

Kreith, Frank, and George Tchobanoglous. 2002. *Handbook of solid waste management*. London: McGraw-Hill.

Kristrom, Bengt, and Pere Riera. 1996. “Is the Income Elasticity of Environmental Improvements Less Than One?” *Environmental and Resource Economics.* 7: 45-55.

Krupnick, Alan, Kenneth Harrison, Eric Nickell, and Michael Toman. 1996. "The Value of Health Benefits from Ambient Air Quality Improvements in Central and Eastern Europe: An Exercise in Benefits Transfer.” *Environmental and Resource Economics.* 7 (4): 307-332.

Kunreuther, Howard, and Paul R. Kleindorfer. 1986. "A Sealed-Bid Auction Mechanism for Siting Noxious Facilities." *The American Economic Review.* 76 (2): 295-299.

Kunreuther, Howard, and Douglas Easterling. 1996. “The Role of Compensation in Siting Hazardous Facilities.” *Journal of Policy Analysis and Management*. 15 (4): 601-622.

Lim, Jong Seok, and Paul Missios. 2007. "Does Size Really Matter? Landfill Scale Impacts on Property Values.” *Applied Economics Letters*. 14 (10): 719-723.

Lindhjem, Henrik, and Stale Navrud. 2008. "How Reliable Are Meta-Analyses for International Benefit Transfers?" *Ecological Economics.* 66 (2-3): 425-435.

Lipsey, Mark W., and David B. Wilson. 2001. *Practical Meta-Analysis. Applied Social Research Methods Series*. 49. Thousand Oaks, California: Sage Publications.

Lober, Douglas J. 1995. "Resolving the Siting Impasse: Modeling Social and Environmental Locational Criteria with a Geographic Information System."*Journal of the American Planning Association.* 61 (4): 482-495.

Loomis, John, Brian Roach, Frank Ward, and Richard Ready. 1995. "Testing Transferability of Recreation Demand Models across Regions: A Study of Corps of Engineer Reservoirs.” *Water Resources Research.* 31 (3): 721-730.

Loomis, John B., and Douglas S. White. 1996. “Economic Benefits of Rare and Endangered Species: Summary and Meta-Analysis.” *Ecological Economics*. 18 (3): 197-206.

Lovett, Andrew, Julii Brainard, and Ian Bateman. 1997. "Improving Benefit Transfer Demand Functions: A GIS Approach." *Journal of Environmental Management.* 51 (4): 373-390.

Luthbom, K., and A. Lagerkvist. 2003. *Tools for Landfill Siting*. Ninth International Waste Management and Landfill Symposium. Italy: CISA.

Maantay, Juliana, and John Ziegler. 2006. *GIS for the Urban Environment*. Redlands, California: ESRI Press.

Malarin, Hector and William J. Vaughan. 1997. *An Approach to the Economic Analysis of Solid Waste Disposal Alternatives*. Washington D.C.: U.S. EPA.

Mӓler, Karl-Gӧran, and Jeffrey R. Vincent. 2003. *Handbook of Environmental Economics*. Amsterdam, Germany: Elsevier.

McCluskey, Jill J., and Gordon C. Rausser. 2003. "Hazardous Waste Sites and Housing Appreciation Rates." *Journal of Environmental Economics and Management.* 45 (2): 166-176.

McBean, Edward A., Frank A. Rovers, and Grahame J. Farquhar. 1995. *Solid Waste Landfill Engineering and Design*. Englewood Cliffs, N.J.: Prentice Hall PTR.

MethodEx. 2007. *Methods and Data on Environmental and Health Externalities: Harmonizing and Sharing of Operational Estimates*. Oxon, UK.

Michaels, R. Gregory, and V. Kerry Smith.1990. "Market Segmentation and Valuing Amenities with Hedonic Models: The Case of Hazardous Waste Sites." *Journal of Urban Economics.* 28 (2): 223-242.

Mitchell, Robert Cameron, and Richard T. Carson. 1986. "Property Rights, Protest, and the Siting of Hazardous Waste Facilities." *The American Economic Review.* 76 (2): 285-290.

______. 1989. *Using Surveys to Value Public Goods: The Contingent Valuation Method*. Washington D.C.: Resources for the Future.

Moeltner, Klaus, Kevin J. Boyle, and Robert W. Paterson. 2007. "Meta-Analysis and Benefit Transfer for Resource Valuation-Addressing Classical Challenges with Bayesian Modeling." *Journal of Environmental Economics and Managem*ent. 53 (2): 250-269.

Moncur, James E. T., and Richard L. Pollock. 1988. "Scarcity Rents for Water: A Valuation and Pricing Model". *Land Economics.* 64 (1): 62-72.

Mueck, Steve, and Errol Nye. 2006. *Flora and Fauna Assessment of a Potential Landfill Site*. Hernes Oak, Victoria: Natural & Cultural Heritage Consultant.

Navrud, Stale. 2004. Value Transfer and Environmental Policy. In *The International Yearbook of Environmental and Resource Economics 2004/2005: New Horizons in Environmental Economics Series,* ed. Tietenberg, Thomas H., and Henk Folmer. Northampton, MA: Edward Elgar Publishing,

Navrud, Stale and Richard C. Ready. 2007. *Environmental Value Transfer: Issues and Methods*. Dordrecht, The Netherlands: Springer.

Nelson, Arthur C., John H. Genereux, and Michelle Genereux. 1992a. "Price Effects of Landfills on House Values." *Land Economics.* 68 (4): 359-65.

______. 1992b. "Price Effects of Landfills on Residential Land Values." *Journal of Urban Planning and Development.* 118 (4): 128-137.

______. 1998. "Price Effects of Landfills on Different House Value Strata." *Journal of Urban Planning and Development.* 123 (3): 59-67.

Nelson, Jon P. 2004. "Meta-Analysis of Airport Noise and Hedonic Property Values: Problems and Prospects". *Journal of Transport Economics and Policy*. 38 (1): 1-27.

O'Hare, Michael, Lawrence S. Bacow, and Debra Sanderson. 1983. *Facility Siting and Public Opposition*. New York: Van Nostrand Reinhold.

O'Leary, Philip, and Patrick Walsh. 2002a. "Landfilling as the Cornerstone of an Integrated Waste System." *Waste Age.* 33 (1): 38-46.

______. 2002b. "Evaluating a Potential Sanitary Landfill Site." *Waste Age.* 33 (5): 74-83.

Opaluch, James J., Stephen K. Swallow, Thomas Weaver, Christopher W. Wessells, and Dennis Wichelns. 1993. "A Landfill Site Evaluation Model That Includes Public Preferences Regarding Natural Resources and Nearby Communities." *Waste Management & Research*. 11 (3): 185-201.

Ortolano, Leonard. 1997. *Environmental Regulation and Impact Assessment*. New York: Wiley.

Palmquist, Raymond B. 2005. Property Value Models. In *Handbook of Environmental Economics*, ed. Karl-Goran Maler and Jeffer R. Vincent. Amsterdam: North-Holland.

Pearce, David. 1980. The Social Evidence of Environmental Costs and Benefits. In *Progress in Resource Management and Environmental Planning*, ed. O'Riordan, T. and K. Turner. London, England: Wiley & Sons.

_______ 2002. *The Role of Property Rights in Determining Economic Values for Environmental Costs and Benefits*. Bristol, England: Environment Agency.

Pearce, David W., Giles Atkinson, and Susana Mourato. 2006. *Cost-Benefit Analysis and the Environment: Recent Developments*. Paris, France: OECD.

Pearce, F. 1989. "Methane: the Hidden Greenhouse Gas." *New Scientist*. 6. 37-41.

Pohland, Frederick G., and Stephen R. Harper. 1987. *Critical Review and Summary of Leachate and Gas Production from Landfills*. Cincinnati, OH: U.S. Environmental Protection Agency.

Pommerehne, W. W. 1988. Measuring Environmental Benefits: A Comparison of Hedonic Efficiency in Public Economics. In *Welfare and Efficiency in Public Economics*, ed. D. Bs, M. Rose, and C. Seidl. Berlin, Germany: Springer Verlag.

Porter, Richard C. 2002. *The Economics of Waste*. Washington, D.C.: Resources for the Future.

Randolph, John. 2004. *Environmental Land Use Planning and Management*. Washington, D.C.: Island Press.

Ready, Mark J., and Richard C. Ready. 1995. "Optimal Pricing of Depletable, Replaceable Resources: The Case of Landfill Tipping Fees." *Journal of Environmental Economics and Management.* 28 (3): 307-323.

Ready, Richard C. 2005. “Do Landfills Always Depress Nearby Property Values? / Issues at Hand.” *NERCRD Regional Rural Development Paper*. 27. University Park, PA: Northeast Regional Center for Rural Development.

Reichert, Alan K., Michael Small, and Sunil Mohanty. 1991. "The Impact of Landfills on Residential Property Values.” *Journal of Real Estate Research.* 7 (3): 297-314

Repetto, Robert C. 1992. *Green Fees: How a Tax Shift Can Work for the Environment and the Economy*. Washington, D.C.: World Resources Institute.

Ringquist, E. J. 2005. “Assessing Evidence of Environmental Inequities: A Meta-Analysis.” *Journal of Policy Analysis and Management.* 24 (2), 223-247.

Roberts, Roland K., Peggy V. Douglas, and William M. Park. 1991. “Estimating External Costs of Municipal Landfill Siting Through Contingent Valuation Analysis: A Case Study.” *Southern Journal of Agricultural Economics.* 23 (2): 155-166.

Rosen, Sherwin. 1974. “Hedonic Prices and Implicit Markets: Product Differentiation in Pure Competition.” *Journal of Political Economy*. 82 (1): 34-55.

Rosenberger, R. S., and J. B. Loomis. 2000a. "Panel Stratification in Meta-Analysis of Economic Studies: An Investigation of Its Effects in the Recreation Valuation Literature." *Journal of Agricultural and Applied Economics*. 32: 459-470.

______. 2000b. *Benefit transfer of Outdoor Recreation Use Values a Technical Document Supporting the Forest Service Strategic Plan*. U.S.: Department of Agriculture, Forest Service, Rocky Mountain Research Station.

______. 2003. Benefit Transfer. In *A Primer on Nonmarket Valuation,* ed. Patricia A. Champ, Kevin J. Boyle, and Thomas C. Brown. Boston: Kluwer Academic Publishers.

Rosenberger, R. S., and T. D. Stanley. 2006. "Measurement, Generalization, and Publication: Sources of error in benefit transfers and their management." *Ecological Economics*. 60 (2): 372-378.

Rosenberger, R. S., and T. Phipps. 2007. Correspondence and Convergence in Benefit Transfer Accuracy: Meta-Analytic Review of the Literature. In *Environmental Value Transfer: Issues and Methods*, ed. Stale Navrud and Richard Ready. Dordrecht, The Netherlands: Springer.

Santo J. M. L. 2007. Transferring Landscape Values. In *Environmental Value Transfer: Issues and Methods*, ed. Stale Navrud and Richard Ready. Dordrecht, The Netherlands: Springer.

Sener, Basak, M. L. Suzen, V. Doyuran. 2006. “Landfill Site Selection by Using Geographic Information Systems.” *Environmental Geology.* 49: 376-388.

Sheppard, J. C., H. Westberg, J. F. Hopper, K. Ganesan, and P. Zimmerman. 1982. “Inventory of Global Methane Sources and Their Production Rates.” *Journal of Geophysical Research*. 87; 1305-1312.

Stocking, Michael, and Niamh Murnaghan. 2001. *Handbook for the Field Assessment of Land Degradation*. Sterling, VA: Earthscan Publications.

Shrestha R. K., and J. B. Loomis. 2003. “Meta-Analytic Benefit Transfer of Outdoor Recreation Economic Values: Testing Out-of-Sample Convergent Validity.” *Environmental and Resource Economics*. 25: 79-100.

Shrestha, R., R.S. Rosenberger, and J. B. Loomis. 2007. Benefit Transfer Using Meta-Analysis in Recreation Economic Valuation. In *Environmental Value Transfer: Issues and Methods*, ed. Stale Navrud and Richard Ready. Dordrecht, The Netherlands: Springer.

Siddiqui, Muhammad Z., Jess W. Everett, and Baxter E. Vieux. 1996. “Landfill Siting Using Geographic Information Systems: A Demonstration.” *Journal of Environmental Engineering.* 122 (6): 515-523.

Smith, V. Kerry, and William H. Desvousges. 1986a. "Asymmetries in the Valuation of Risk and the Siting of Hazardous Waste Disposal Facilities." *American Economic Review*. 76 (2): 291-294.

______. 1986b. "The Value of Avoiding a LULU: Hazardous Waste disposal Sites." *The Review of Economics and Statistics*. 68 (2): 293-299.

Smith, V. Kerry, and Ju-chin Huang. 1995. "Can Markets Value Air Quality? A Meta-Analysis of Hedonic Property Value Models." *Journal of Political Economy***.** 33(1): 209-227.

Smith, V. Kerry, and Subhrendu K. Pattanayak. 2002. "Is Meta-Analysis a Noah's Ark for Non-Market Valuation?" *Environmental & Resource Economics.* 22 (1): 271-296.

Smolen, Gerald E., Gary Moore, and Lawrence V. Gonway. 1992. "Economic Effects of Hazardous Chemical and Proposed Radioactive Waste Landfills on Surrounding Real Estate Values." *Journal of Real Estate Research.* 7: 293-295.

Stanley, T. D. 2001. "Wheat from Chaff: Meta-Analysis as Quantitative Literature Review." *Journal of Economic Perspectives*. 15: 131-150.

Stanley, T. D., and S. B. Jarrell. 2005. "Meta-Regression Analysis: A Quantitative Method of Literature Surveys". *Journal of Economic Surveys*. 19 (3): 299-308.

Stocking, Michael, and Niamh Murnaghan. 2001. *Handbook for the Field Assessment of Land Degradation*. Sterling, VA: Earthscan Publications.

Swallow, Stephen K., James J. Opaluch, and Thomas F. Weaver. 1992. "Siting Noxious Facilities: Approach that Integrates Technical, Economic, and Political Considerations." *Land Economics*. 68 (3): 283-301.

Swallow, Stephen K., Thomas Weaver, James J. Opaluch, and Thomas Michelman. 1994. "Heterogeneous Preferences and Aggregation in Environmental Policy Analysis: A Landfill Siting Case." *American Journal of Agricultural Economics.* 76: 431-443.

Taylor. 2003. The Hedonic Method. In *A Primer on Nonmarket Valuation,* ed. Patricia A. Champ, Kevin J. Boyle, and Thomas C. Brown. Boston: Kluwer Academic Publishers

Tchobanoglous, George, Hilary Theisen, and Samuel A. Vigil. 1993. *Integrated Solid Waste Management: Engineering Principles and Management Issues*. New York: McGraw Hill.

______. 2002. *Handbook of Solid Waste Management.* New York: McGraw-Hill.

Thayer, Mark, Heidi Albers, and Morteza Rahmatian. 1992. "The Benefits of Reducing Exposure to Waste Disposal Sites: A Hedonic Housing Value Approach." *Journal of Real Estate Research*. 7(3): 265-282.

United States. 1995. *Hazardous and Nonhazardous Waste Demographics of People Living near Waste Facilities: Report to Congressional Requesters*. Washington, D.C.: The Office.

U.S. EPA. 1993. *MSW Landfill Criteria Technical Manual*. Springfield, VA.

______. 1995. *Decision-Maker's Guide to Solid Waste Management. Volume II*. Washington, D.C. Office of Solid Waste, Municipal and Industrial Solid Waste Division, RCRA Information Center.

______. 2007. *Municipal Solid Waste in United States: Facts and Figures*. Springfield, VA.

Vasiloglou, V. C. 2004. “New Tool for Landfill Location.” *Waste Management & Research. 22:427-439.*

Walsh, Patric an Phillip O’Leary. 2002. “Landfill Community Relations Programs.” Waste Age 33 (11): 58-63.

Walton, Harry, Richard Boyd, Tim Taylor, and Anil Markandya. 2006. *Explaining Variation in Amenity Costs of Landfill: Meta-Analysis and Benefit Transfer*. Bath, UK: Methodex.

Westlake, Kenneth. 1995. *Landfill Waste Pollution and Control*. Chichester, West Sussex, England: Albion Publishing Limited.

White, Halbert. 1980. “A Heteroskedasticity-Consistent Covariance Matrix Estimator and a Direct Test for Heteroskedasticity.” *Econometrica*. 48: 817-838.

Wooldridge, Jeffrey. 2000. *Introductory Econometrics: A Modern Approach*. Mason, Ohio: Thomson.

______. 2002. *Econometric Analysis of Cross Section and Panel Data*. Cambridge, Mass: MIT Press.

Ye M. H., Yezer A. M. J. 1997. “Where Will We Put the Garbage? Economic Efficiency versus Collective Choice.” *Regional Science and Urban Economics*. 27.1: 47-64.