INCORPORATING ENVIRONMENTAL COSTS INTO AN ECONOMIC ANALYSIS OF WATER SUPPLY PLANNING: A CASE STUDY OF ISRAEL

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ABSTRACT

The world is facing a growing challenge in maintaining water quality and meeting increasing demands for water resources. This trend is evident in the Middle East where water scarcity has reached critical levels. To cope with shortage, many Middle Eastern countries are exploring unconventional water sources. However, most discussions and project analyses focus on the geopolitical dimension of the water crisis and supply planning, ignoring the additional social costs of water projects, like externalities. This study explores ways to include environmental impacts in the economic assessment of water supply options to determine how social costs, defined as private plus external costs, change the relative attractiveness of water supply alternatives. Using the marginal opportunity cost framework, the direct, external, and user costs of three water supply projects in Israel are valued: (1) groundwater extraction and depletion, (2) wastewater reclamation and reuse in agriculture, and (3) desalination. The study suggests that an analysis using private costs alone is misleading, since full social costing changes the relative attractiveness of the project alternatives. Therefore, Israeli policy makers may not always make socially efficient decisions about water supply. The research concludes by discussing the analysis within the broader policy context, highlighting the other policy options available to decision makers, additional research needs, and the difficulty of achieving sustainability in a political unstable region.

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LIST OF ABBREVIATIONS

BFT	Benefits function transfer
CA	Coastal Aquifer
CO_2	Carbon dioxide
COI	Cost of illness
CVM	Contingent valuation method
dS/m	Decisiemens per meter
DC	Direct cost
DoE	U.S. Department of Energy
DSM	Demand-side management
EC	External cost
EC_e	Electrical conductivity of the soil saturation extract
EC_{iw}	Electrical conductivity of the irrigation water
EC_{sw}	Electrical conductivity of the soil water
ECU	European currency unit
Kg N/ha	Kilograms of nitrogen per hectare
kWh	Kilowatt-hour
m^3	Cubic meter
MDC	Marginal direct cost
MEC	Marginal external cost
mg/l	Milligrams per liter
Mm^3	Million cubic meters
mmhos/cm	Millimhos per centimeter
MOC	Marginal opportunity cost
MSF	Multistage flash distillation
MUC	Marginal user cost
N-D	Nitrification-denitrification
NIS	New Israeli Shekel
NO_x	Nitrous oxides
PM	Particulate matter
RO	Reverse osmosis

SAR	Sodium adsorption ratio
SAT	Soil and aquifer treatment
SO_2	Sulfur dioxide
TCM	Travel cost method
UC	User cost
U.K.	United Kingdom
U.S.	United States
U.S.D.	United States dollar
WTP	Willingness to pay

CHAPTER 1: INTRODUCTION

1.0 Introduction

Given the importance of water to human and ecosystem survival, water quantity and quality are an important environmental concern. Evidence already exists that the world is facing a growing challenge in maintaining water quality and meeting the rapidly growing demand for water resources (Rosegrant 1997). Many regions of the world that deal with critical water shortages and contamination are facing famine, economic breakdown, and potential warfare (Starr 1990). Within the Middle East Region, severe water scarcity is a problem as most countries' water availability is below 1000 cubic meters/person/year, the threshold considered necessary for industrial, population, and agricultural development (Shiklomanov 2000). Israel and Jordan are below the 500 cubic meters/person/year mark, defining them as water stressed (Shuval 1992).

The struggle of Middle East countries to meet present and future demand for water resources has led to the exploitation of unconventional water sources. The importation of water via the sea and pipelines, desalination, wastewater reclamation and reuse, as well as regional water diversions have been discussed for years. Yet, most of the debate has centered on the technical, financial, and political aspects of increased water supply. With geopolitics playing a central role in most of the proposed projects, the environmental implications of water supply alternatives have been overlooked. Therefore, decision makers have not considered the full social costs of supply, which includes the private costs an agent incurs in conducting an activity and the external costs that fall on other people who cannot exact compensation for them (Black 1997). As the political and water situation in the Middle East worsens, many countries are moving towards unilateral water development within national borders. Decision makers perceive projects like desalination and wastewater reclamation and reuse as the answer to water scarcity. With unconventional water sources becoming the prominent supply solution, social costing is necessary to ensure efficient resource use and socially optimal decisions.

Social costing allows policy makers to make socially efficient resource use decisions for two reasons (Field and Olewiler 1994). First, water planning and pricing based on social costs ensures the optimal amount of development occurs at the optimal price. Without considering full social costs, the quantity of water consumed is too high and the price per unit of water supplied is too low. Second, social costing allows policy makers to formulate the socially optimal choice among project alternatives. Given the enormous cost associated with new water supply projects, selecting the project with the lowest social cost is imperative.

1.1 Research Objectives

This research explores ways to include environmental impacts in the assessment of water supply options, using Israel as a case study. A central research question guides this study:

How does social costing change the cost of water supply development?

- For a water supply project in isolation?
- For water supply projects relative to each other?

In an effort to answer these research questions, this study conducts an economic valuation analysis using the marginal opportunity cost framework (Pearce and Markandya 1989). The analysis details three water development projects that form a major part of Israel's water policy: groundwater extraction and depletion, wastewater reclamation and reuse in agriculture, and desalination. The goal of the analysis is to fill a knowledge gap that prevents decision makers from making socially optimal decisions about water planning and development. By examining the effects of social costing on the projects, I hypothesize that their costs will increase significantly and their relative attractiveness will change. Since policy makers should base national water planning on social, not private costs, such a result could have important policy implications.

1.2 Scope and Boundaries of the Study

The State of Israel is the study area. Thus, social costs are limited to Israel's national borders and the analysis ignores any global impacts from the water supply alternatives. Israel was selected as the case study since per capita water availability is among the lowest in the Middle East, the exploitation of unconventional water sources is a national policy, the economy is moving toward a mature western model permitting investment in expensive projects, and historical evaluations of water projects have typically considered private costs in isolation from other social costs (Beamont 2000).

The key study variable is the social cost of water supply development. The analysis omits the estimation of benefits. Since the Israeli water distribution system mixes various sources of supply, the Israeli government does not differentiate between sources of water. Therefore, as long as water quality remains constant, the benefits to water users are treated as equal across all project alternatives. In addition, the analytical sections do not address the political aspects of water supply development. Chapter seven summarizes the relevant political issues with the policy implications of the analysis.

1.3 Report Organization

The following chapter summarizes the literature on Middle East water supply, including the geopolitics of water and the economic data for various projects in the region. Subsequently, chapter two elaborates on the three water projects under consideration in this study. Chapter three presents the study approach and analytical methods. Chapter four, five, and six describe the impacts of each project, provide the economic valuation of groundwater extraction and depletion, wastewater reclamation and reuse in agriculture, and desalination respectively, and report the social costs of each project. Chapter seven concludes the report with a discussion of the policy implications and conclusions.

CHAPTER 2: REVIEW OF WATER IN THE MIDDLE EAST

2.0 World Water Supply

Water is the most important natural resource because it is the basis of life on earth. Clean, available water plays an essential role in the quality of human life, economic, and social development as well as human health (Shiklomanov 2000). However, water availability is becoming an important global problem as the demand for freshwater increases and the quantity of good quality water decreases. Many countries are already exploiting conventional water sources beyond their annual recharge and new sources of water are becoming increasingly expensive to access[1]. In addition, pollution from industrial, agricultural, and household discharges is reducing water quality and affecting human and ecosystem health (Rosegrant 1997). As a result, many countries with water shortages are facing famine, economic breakdown, or potential warfare (Starr 1990).

2.1 Middle East Water Supply

The Middle East is facing severe water shortage. Currently, most countries' water supply is less than the 1000 cubic meters (m^3)/person/year threshold considered necessary for industrial, residential, and agricultural development (Shiklomanov 2000). Furthermore, most major river systems cross international borders, making water shortages subject to political conflict. Because of the complexity of water scarcity, there is an extensive literature dealing with Middle East water issues. Most studies address these issues from two perspectives. The first surveys the geopolitics of water and the second focuses on the economic evaluation of supply projects. Geopolitical issues arise from water shortages in areas where river or groundwater systems cross international borders. Therefore, the literature discusses conflicts among nations in the Tigris-Euphrates, the Nile, and the Jordan basins in addition to the Israeli-Palestinian negotiations on the joint use of one aquifer. Economic evaluations of water supply options quantitatively assess unconventional water sources available to Middle East countries. Each of these topics is discussed below.

[1] A conventional water source is groundwater or surface water extracted at or below the renewable recharge and used in a single country. An unconventional water source is any other water source.

Three major Middle East water systems are subject to transboundary political conflict: (1) Tigris-Euphrates, (2) Jordan River, and (3) Nile River. The Tigris-Euphrates system, which originates in Turkey and crosses into Syria and Iraq, is a source of conflict because of Syrian and Iraqi demands for increased water allocations and Turkey's unilateral continuation of the GAP project (Wolf 1996). The Nile River conflict stems from Egypt's allocation demands and the potential for unilateral upstream development by Ethiopia (Wolf 1996; Sadik and Barghouti 1994). The conflict in the Jordan basin arises from the allocation demands of Israel, Jordan, the Palestinian Territories, Syria, and Lebanon and is complicated by the absence of peace between Israel and her neighbors (Wolf 1994). Israel controls the basin since its capture of the headwaters in 1967, and unilateral water usage by Israel has created conflicts between Israel and Jordan and Israel and the Palestinians. Moreover, because the headwaters of the Jordan River were Syrian land, territorial disagreements exist between Israel and Syria (Biswas et al. 1997)[2].

In addition to surface water conflicts, the Israelis and Palestinians disagree over groundwater extraction because Israel pumps one third of its water from an aquifer that recharges in the West Bank (Baskin 1993). Although, there is no international law in force to govern the use and development of international groundwater water basins (Rosegrant 1997), water experts have proposed allocation schemes based on the principle of equitable apportionment (see Shuval (2000, 1992) and Assaf et al. (1993))[3]. Other approaches to water allocation include formulas for distribution based on state obligations, natural water flows, and the use of open markets (Zarour and Isaac 1993). Since there is no consensus on the appropriate allocation mechanism, discussions continue between Palestinian and Israeli water experts (Feitelson and Haddad 1994-6)[4].

[2] For additional details on the history and context of the Jordan basin conflict, see Amery and Wolf (2000), Wolf (1995), Wolf and Lonergan (1995), Biswas (1994), Isaac and Shuval (1994), Kliot (1994), Lonergan and Brooks (1994), and Lowi (1993).

[3] Many countries have accepted the following set of principles for water disputes among riparian countries: (1) prior consultation, (2) avoidance of significant injury, (3) equitable apportionment of water, (4) nondiscrimination and nonexclusion, and (5) provision of settlement of disputes. These principles are included in the Helsinki rules formulated by the Law Association in 1966 and are not binding (Rosegrant 1997). Although these principles were originally drafted for surface water, experts equally apply them to groundwater (Shuval 2000).

[4] Additional references on the Israeli-Palestinian dispute over groundwater sources can be found in Rouyer (2000), Soffer (1998), Rouyer (1997), Baskin (1994), Elmusa (1994), and Kally (1991/2).

The second focus of Middle East water studies is the economic evaluation of water supply options[5]. This perspective describes and quantifies water scarcity and proposes a solution grounded in a particular project. Table 2.1 highlights the prominent Middle East water supply projects, their estimated private costs of supply, and the source of the economic evaluation.

[5] "Water supply options" is used interchangeably with water supply alternatives, water supply development, water development options, and water supply projects.

Table 2.1: Private Costs of Middle East Water Supply Projects (1999 U.S.D.)

Project	Project Description	Cost/m^3
Groundwater Depletion	The extraction of groundwater above the yearly renewable recharge. Used to meet current water shortages.	Unknown
Wastewater Reclamation and Reuse[6]	Treatment facilities reclaim urban sewage for reuse in agriculture, municipal, and industrial sectors.	\$0.06-0.36 Hoffman and Harussi (1999)
Litani Diversion(1)	Diversion from the Litani River (in Lebanon) to Israel and Jordan (Hussein and Al-Jayyousi 1999). Additional information: Murakami and Musiake (1994).	\$0.12-0.15 Haddad and Lindner (2001)
Nile Diversion	Diversion from the Nile River to Gaza and possibly Israel (Wachtel 1993). Additional information: Bleier (1997), Dinar and Wolf (1994), and Fishelson (1994).	\$0.23 Kally and Fishelson (1993)
Med-Dead/ Red-Dead Canal	Pipeline from the Mediterranean or Red Sea to the Dead Sea for hydropower and desalination plants (Chamish 1994). Additional information: Israel MOEI (1995), Murakami (1995), Glueckstern, and Fishelson (1992)	\$0.71-0.83 Israel MOEI (1995)
Desalination	Desalination of seawater or brackish water[7]. Facilities exist in the Arabian Peninsula and Israel. Additional information: Glueckstern (1999), Glueckstern and Priel (1999), and Livnat (1994).	\$0.73-0.81 Priel (2001)
Imports from Turkey	The Israeli and Turkish governments have discussed water imports via tanker.	\$0.85(2) Cohen (2000)
Turkey's Peace Pipeline	Diversion from Turkey's Ceyhan and Seyhan Rivers to parts of the Middle East. Additional information: del Rio Luelmo (1996), Gruen (1993), and Utkan (1991).	\$1.30-1.60(3) Haddad and Lindner (2001)
Turkey's Mini Peace Pipeline	Diversion from Turkey's Ceyhan and Seyhan Rivers to Amman and the West Bank.	Mini Pipeline: \$0.30-0.40(4) Wachtel (1993)

Remarks: (1) Assumes hydroelectric generation on the route produces U.S. \$0.05/m^3 of electricity; (2) Cost depends on the shipping fees and the price Turkey charges per cubic meter of water; (3) The cost estimates are uncertain and should be used with caution; (4) Assumes energy self-sufficiency from hydropower along the route.

[6] "Wastewater reclamation and reuse" is used interchangeably with effluent reuse, reuse of treated sewage, reuse of treated wastewater, and effluent irrigation (when allocated to agriculture).

[7] Freshwater typically possesses less than 1,000mg/l of total dissolved solids, seawater approximately 33,000mg/l total dissolved solids, and brackish water approximately 1,000–3,000mg/l total dissolved solids.

The costs of unconventional water sources are U.S. \$0.06-1.60/m^3 (Table 2.1). These figures only represent private costs. In addition, each project requires various levels of international cooperation, making some of the alternatives unfeasible in the current political climate. By way of comparison, Arlosoroff (N.d.) cites demand-side management program costs at U.S. \$0.05-0.40/m^3 in Israel.

The costs of unconventional water sources gain perspective when compared to the price Israeli consumers pay for water. Table 2.2 summarizes the Israeli fee structure by sector using average prices (water is normally charged using increasing block rates). Since some sectors are subsidized, the price per cubic meter varies.

Table 2.2: Average Israeli Water Prices by Sector (1999 U.S.D.)

Sector	Price/$m^{3(1)}$
Agricultural Use	
Freshwater	\$0.21
Runoff/Partly Salinated Water	\$0.15
Effluent	\$0.12
Industrial Use	\$0.33
Residential Use(2)	\$0.87

Source: (Plaut 2000)

Remarks: (1) 1U.S.D. = 4 New Israeli Shekel (NIS); (2) This number represents the average price charged by municipalities to residential consumers. Mekorot, Israel's water supply company, charges a municipality U.S. \$0.34/m^3 for all water provided to households.

The average price for a cubic meter of freshwater in Israel is U.S. \$0.21-0.87 (Table 2.2). The prices charged for freshwater are not based on where water originates since the distribution system mixes various sources of supply. Thus, as long as water quality is the same, any benefit received by a sector will remain constant even if the government changes the freshwater source. The only exception is the use of treated wastewater in agriculture, since potable water and effluent differ in water quality and price. However, the analysis makes an adjustment for this difference in Section 5.1.

2.1.1. Conclusions: Middle East Water Supply

The review of Middle East water supply highlights two intertwined problems. The first problem relates to the broader question of water allocations among nations, and speaks directly to the political climate of the region. The second problem relates to water scarcity and the project(s) that are best suited to solve water shortages. This problem addresses project-specific issues like private costs and technical considerations. The major difference between these problems is that the first advances the debate on conflict dispute resolution, a question that policy makers are working to solve via negotiation, while the second advances the debate about water scarcity, a problem that will exist even with a peace treaty. Consequently, even with peace, Middle East nations must explore large-scale unconventional water schemes for domestic water supply.

The above discussion highlights a gap in the literature regarding water supply development. Clearly, the exploitation of unconventional water sources is necessary in the Middle East. However, with geopolitics playing a central role in water development, the debate has centered on the technical, financial, and political aspects of the projects. Policy discussions, especially in Israel, have been preoccupied with these dimensions of viability. As a result, the environmental implications of water supply development have been ignored.

2.2. Water Supply in Israel

This report selects three of the water development options presented in Table 2.1 for a fuller analysis. These options are: (1) groundwater extraction and depletion, (2) wastewater reclamation and reuse in agriculture, and (3) desalination. Groundwater extraction was selected because aquifer depletion is the only current method of meeting demand until new desalination and wastewater treatment plants become operational in the next four years, making it the *de facto* water policy. Wastewater reclamation and reuse was chosen since it is a national priority and the Israeli government has committed to reuse all effluents within the next five years[8]. Desalination was selected because of recent commitments to four large-scale desalination plants (Tal 2001). Together, these projects make up an important part of Israel's water policy moving into the twenty-first century.

2.2.1. Groundwater Extraction and Depletion

Although Israel has an intricate and closely monitored water system, the persistent growth in population, industry, and agricultural development has led to the depletion of its major water sources. Israel relies on two aquifers and one lake for almost all of its water supply and these water sources are discussed below.

The Mountain Aquifer underlies the foothills in the center of the country and is mainly composed of karstic limestone (Figure 1). The basin comprises three subaquifers: the Western, North Eastern, and Eastern Aquifer. The Western Basin, also known as Yarkon-Taninim, flows in north and westward directions, with overflows discharging in the Taninim Springs. The Northeastern Basin flows to the northeast with discharges in the Beit Shean Springs. The Eastern Basin flows towards the Jordan Rift Valley with saline discharges in the northern Dead Sea Region. The Western Basin has high quality water, although chloride concentrations have increased in the last 30 years, resulting in concentrations ranging from 50-250mg/l. The Northeastern Basin has deteriorated from surface contamination linked to agricultural practices and saline water intrusion from depletion. The Eastern Basin has high water quality and low chloride and nitrate concentrations. All three basins are regenerated by precipitation with average annual renewable recharges of 360 million cubic meters (Mm^3), 145Mm^3, and 170Mm^3 respectively. Renewable recharge represents less than 10% of the total aquifer capacity (Jordan MWI *et al*. 1998).

The Coastal Aquifer (CA) underlies the coastal plain, adjacent to the Mediterranean Sea, and is composed of sandstone (Figure 1). The aquifer is bounded to the east by the

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[8] The Israeli government considers agriculture a national priority rooted in the history of the country's development. The Water Commissioner's current five-year plan indicates that 90% of all wastewater is

foothills of the mountain belt, in the north by the Carmel Mountains, in the south by the Sinai Desert, and in the west by the Mediterranean Sea (Jordan MWI *et al*. 1998). The major flow of the reservoir is towards the Mediterranean Sea where it eventually interfaces with seawater (Nativ and Isaar 1988). The CA is a valuable storage basin since the sandstone layers hold water efficiently. However, water quality has been severely affected by development on the coastal plain, overpumping, and the circular flow of water from extraction to irrigation to drainage recharge, leading to increases in salinization (Isaar 1998). Average chloride concentrations range from 50-250mg/l but reach 6000mg/l in some parts of the coast. Average nitrate concentrations are between 10 and 70mg/l (Jordan MWI *et al*. 1998). The aquifer has a mean annual recharge of 250Mm^3 in addition to 50Mm^3 of agricultural drainage, representing less than 5% of total reservoir capacity (Kessler 2000).

Lake Kinneret, also called the Sea of Galilee or Lake Tiberias, is the only surface water lake in the State of Israel (Figure 1). Located in the Galilee Region, the upper Jordan River and numerous smaller streams feed the lake (Jordan MWI *et al*. 1998). Water levels are regulated between 209m and 214m below sea level. The sea is bounded at the lower end by the threat of saline intrusion from springs trapped in the lower reaches of the lake and is bounded at the upper end by the location of the City of Tiberias and other settlements on the banks of the lake (Berkowitz 2000). The water quality of the Kinneret is moderate with average chloride concentrations approximately 200mg/l (Jordan MWI *et al*. 1998). The lake has a surface area of 167km^2, an average depth of 26m, and a renewable water supply of approximately 465Mm^3 (Jordan MWI *et al*. 1998).

2.2.2. Wastewater Reclamation and Reuse in Agriculture

Israel's water scarcity greatly affects its farm sector through the limitation of agricultural possibilities. Currently, the agricultural sector receives 60% of the freshwater supply. However, population growth and increasing urban demand for freshwater will require reallocation of good quality water to domestic uses (Weinstein 1996). Within the next 40 years, the country will be devoting almost all of its freshwater supply to domestic

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allocated to agriculture (Hoffman and Harussi 1999).

consumption (Arlosoroff 1995b). Therefore, to maintain agriculture, unconventional water sources, including treated wastewater, must replace freshwater allocations. By 2040, treated sewage will constitute 70% of agricultural water supply (Haruvy *et al*. 1997b) [9].

Israeli agriculture has used treated wastewater for decades with treatment levels significantly improving with time (Weinstein 1996). Sewage plants use three levels of treatment: (1) primary treatment such as screening of coarse solids and grit removal, (2) secondary or biological treatment using low rate processes like stabilization ponds or high rate processes like activated sludge, and (3) tertiary treatment using nitrification-denitrification processes (to reduce macronutrient levels) and soil and aquifer treatment (SAT) (Haruvy 1997). Regulations promulgated by the Ministry of Health in 1992 require secondary treatment to a minimum baseline of 20mg/l biological oxygen demand and 30mg/l total suspended solids in every settlement with a population of more than 10,000 people (SOI 2000). The regulations also apply if municipalities dump effluent into rivers, streams, or the ocean.

In Israel, piped sewage is treated in biological treatment plants, either in oxidization ponds, aerated lagoons, or in activated sludge systems. Treatment facilities may include nitrogen and phosphorous removal. After treatment, the effluent is placed in seasonal reservoirs; the storage bodies that regulate the constant flow of treated wastewater and the seasonal demand for irrigation. The storage reservoir is either a surface water body or an underground confined aquifer (i.e. SAT). Water quality in surface reservoirs is inferior to that from aquifer storage and the use of treated wastewater from surface reservoirs is limited to irrigation of industrial crops, fodder, and other nonedible crops, like cotton, forests, and pastures. Effluent from tertiary treatment with SAT is released for unrestricted irrigation (Shevah and Shelef 1995).

[9] Treated wastewater is mainly a product of domestic uses. The industrial contribution to the wastewater stream is nominal and because of pollution problems, industrial users are required to pretreat their discharges before releasing sewage into the municipal system (Arlosoroff 1995a). Therefore, industrial sewage released into the domestic wastewater stream is of similar quality to urban sewage (Gabbay 1998).

2.2.3. Desalination

Desalting seawater is a technically proven solution to chronic water shortages in many countries of the Middle East. For Israel, located along the coast, it promises an unlimited supply of water. Mekorot, Israel's national water supply company, has built and operated small- and medium-sized desalination facilities in the southern part of the country since the 1960's. Eilat, a small tourist town located by the Red Sea, was the first city to use desalination and its facilities comprise 90% of Israel's desalinating capacity (Glueckstern and Priel 1999).

Numerous desalination technologies have been developed since the late 1950's when the desalting of seawater was invented. Today, two technologies dominate in use: multistage flash distillation (MSF) and reverse osmosis (RO). MSF is a distillation method where vapors are evaporated from saline water. The process then condenses the vapor to form freshwater. The RO process, on the other hand, pushes saline water through a membrane that allows passage of water molecules but prevents passage of dissolved materials. The result is two liquids, freshwater and brine, where brine is defined as a liquid more saline than seawater (Keenan 1992). Although the RO membranes are sensitive to initial water quality, because the process requires less energy per cubic meter of freshwater produced, it will most likely be the technology used in Israeli desalination plants.

Mekorot has been involved in desalination since the 1960's when it opened the "Sabra" plant in Eilat. The company has pursued implementation of existing technologies as well as analytical studies and field-testing of new technologies for the last 30 years. Mekorot started testing the RO technology for brackish water in the 1970's; by the summer of 1998, Mekorot was operating 34 brackish water RO units in 26 different sites within Israel. In parallel with the implementation of brackish water RO in the 1970's, Mekorot started field-testing seawater RO plants. In June 1997, Mekorot completed the design of the first seawater RO plant in Israel in the Sabra facility (Glueckstern 1999). In April 2000, Israel's Ministerial Committee for Economic Affairs approved recommendations for the construction of the country's first major seawater desalination plant on the shores of Ashkelon. The plant will provide 50Mm^3 of water per year at an approximate cost of

U.S. \$0.70/m^3 (Liu 2000). Moreover, three more plants, scheduled for construction by 2004, will provide an additional 150Mm^3 of desalination capacity and are slated for construction along the coast (Tal 2001).

2.3 Summary

Water scarcity is forcing policy makers to exploit unconventional water sources to meet growing demand. However, because water supply is subject to geopolitical conflicts, social costing of project alternatives has not occurred. Thus, decision makers need a framework for incorporating environmental impact into project evaluation to rank unconventional supply projects based on their social costs. The next chapter introduces the study approach and analytical methods.

CHAPTER 3: STUDY APPROACH AND METHODS

3.0 Introduction

This chapter describes the study approach and analytical methods used in this study. Section 3.1 presents the conceptual framework and each of its components. The valuation methods associated with each component of the theoretical framework are summarized in Section 3.2. Section 3.3 provides the evaluation stance, including the assumptions used in the analysis and the structure of each analytical chapter.

3.1 Conceptual Approach

Incorporating environmental impacts into project evaluation requires a conceptual framework that adequately accounts for all social costs relevant to the projects under investigation. This study uses the marginal opportunity cost (MOC) approach because it provides a framework for explaining and understanding social costs, and it captures all the relevant costs of Israeli water supply development in a unified manner. Marginal cost is defined as the cost associated with a small or unit change in the rate of usage, while opportunity cost is defined as the next best alternative use for a given resource (Warford 1997). Although the original application of MOC was to natural resource depletion, it is also applicable to public investment decisions (Pearce and Markandya 1989). Furthermore, it is especially relevant to water supply planning since marginal costs that include environmental, economic, and disposal costs should form the basis of water pricing to ensure efficient resource use (Warford 1997).

MOC comprises the sum of three components measured in economic terms and expressed as (Pearce and Markandya 1989):

$$MOC = MDC + MEC + MUC \tag{3.1}$$

Where:

MOC = marginal opportunity cost
MDC = marginal direct cost
MEC = marginal external cost
MUC = marginal user cost

In Equation (3.1), marginal direct cost (MDC) incorporates the private costs of water development, while marginal external cost (MEC) and marginal user cost (MUC) capture the additional social costs typically ignored in the financial analyses of the water projects summarized in Table 2.1. The MDC, MEC, and MUC are all measured using economic costs, which represent the true opportunity cost or the cost net of any market imperfections or transfer payments. The following paragraphs discuss each of these concepts in more detail.

3.1.1. Marginal Direct Cost

MDC includes investment and operating costs incurred by the responsible agency in the production of the good/service in question (Warford 1997). For example, sewage reclamation requires labor to operate a treatment plant and materials to run the treatment process. These types of costs make up a part of the MDC of wastewater treatment. The private costs of the water supply projects listed in Table 2.1 should equal the MDC if there are no price distortions. This study assumes there are no pricing distortions and therefore, the terms private cost and direct cost are used interchangeably in the remainder of this report.

3.1.2. Marginal External Cost

MEC captures the externalities associated with a project and the behavioral responses to a policy intervention. Externalities are positive or negative attributes or effects of a good/service or its production not reflected in the price of the product/service; instead, they are shifted onto others (Perkins 1994). An example of an externality is the human health cost associated with particulate matter emitted by a coal-burning power plant. The loss of income from reduced workdays is not included in the cost of electricity, but externalized to communities located downwind from a plant.

MEC can also include the behavioral responses to a policy intervention. For example, if the government replaces agriculture’s freshwater allocations with treated wastewater, a farmer’s response to the policy might include crop switching to avoid yield reductions from the excess salinity in the effluent. Freeman (1993) presents a model that captures this notion of an externality where a change in production of an economic agent stems from a government intervention[10]. Freeman's model incorporates three sets of functional associations. First is the physical relationship between some measure of environmental or resource quality and the human interventions that affect it. The intervention modeled explicitly is government actions to prevent or ameliorate unregulated market activity or to prevent or enhance the value of a market or nonmarket service. Second is the relationship between human uses of the environment or resource and human dependence on that environmental asset or resource. Typically, human dependence on the environmental or resource asset is related to how much of the asset they use and the other inputs into the production process. The third relationship gives the economic value of the uses of the environment and can be measured in monetary terms. By combining these distinct relationships, Freeman's model shows the magnitude of impact a government intervention has on an economic agent. This behavioral model is important for this analysis because the Israeli government intervenes in the provision of water to agriculture when it forces farmers to accept treated wastewater in place of freshwater (Section 5.3.1). Freeman's model is used for conceptualizing the impacts on Israeli farmers from this forced substitution.

3.1.3. Marginal User Cost

MUC arises from intertemporal considerations associated with the depletion of a nonrenewable resource, or the exploitation of a renewable resource above natural regeneration[11]. In both instances, the use of the resource today precludes the use of that portion of the resource tomorrow. The MUC represents the cost of foregone future benefits. In some cases, resource managers or owners may take MUC into account. This inclusion occurs when property rights for the resource in question are clearly defined, and social and financial discount rates are congruent (Warford 1997). However, this analysis is concerned with situations where this is not the case.

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[10] Pearce and Nash (1981) define externalities as "variables controlled by one agent that enter into the production function of another agent."
[11] Marginal user cost is synonymous with royalty, resource rent, and depletion premium (Pearce and Turner 1990).

The user cost concept has traditionally been used to calculate the optimal depletion rate of a nonrenewable resource (Pearce and Turner 1990). Since natural resource economics treats resources in the ground as capital assets, the user cost represents the royalty on the marginal unit of a resource, or the expected capital gains accruing to the owner of the resource as the resource price rises through time. The optimal price of a nonrenewable resource is, therefore, equal to the sum of the extraction costs and the MUC (Pearce and Turner 1990). User cost is an important natural resource concept since it helps define the optimal intertemporal use of a natural resource (Howe 1979).

3.2 Study Methods

This section explores the analytical methods available to value the marginal direct, external, and user costs of a water project. Where appropriate, the following three subsections also provide a rationale for the methods used to quantify the environmental impacts of the three selected water projects.

3.2.1. Marginal Direct Cost

If there are no pricing distortions, a project's direct cost is calculated using market prices and engineering cost estimates. Since this analysis assumes no pricing distortions, no adjustments are made to market prices and the financial and economic direct costs are considered equal.

3.2.2. Marginal External Cost

Various economic valuation methods are needed to calculate the MEC of a policy intervention since some behavioral responses and externalities have market prices and others do not. This report uses the following valuation methods for quantifying the externalities of water supply development: (1) market prices, (2) changes in productivity, (3) dose-response functions, (4) control cost, (5) travel cost method (TCM), and (6) contingent valuation method (CVM). Market prices, changes in productivity, dose-response functions, and control costs are direct valuation approaches that use actual market prices or observable behaviors. TCM is a direct valuation approach that uses surrogate markets and indirectly infers a value from observed behavior. CVM is a survey-oriented approach and uses hypothetical behavior to estimate values (Tietenberg 2000; Hufschmidt et al. 1983). Each method is described below (IIED 1994; Hufschmidt et al. 1983; Dixon et al. 1983).

1. **Market prices:** Use the prevailing prices for goods and service traded in domestic or international markets and include changes in the value of output and loss of earnings. Market prices are frequently used in this report because price information is relatively easy to obtain and market prices accurately reflect willingness to pay (WTP) for costs and benefits of goods and services that are traded. However, market prices do not reflect nonuse values and nonmaterial damages. Thus, they may underestimate an externality. When market prices are adjusted for distortions, they are called shadow prices.
2. **Changes in productivity:** Physical changes in production are valued using market prices for inputs or outputs. Changes in productivity occur when a project or policy causes unintended damages to another productive system.
3. **Dose-response function:** Measures the value of a nonmarket resource by modeling the physical contribution of the resource to economic output. Dose-response functions estimate the entire demand function, but they require explicit modeling of the dose-response relationship, which is complex and uncertain.
4. **Control cost:** Measures the value of an environmental asset by the costs incurred in avoiding a negative impact. Control costs are easy to quantify because they are based on market prices and use actual expenditures. However, the results may underestimate the true effects since nonuse values and nonmaterial damages are excluded. Control cost is also called preventative expenditures.
5. **Travel cost method:** Estimates the demand for recreational sites by measuring the direct costs of visiting those sites. This method uses market prices and actual expenditures. However, the results may underestimate the true value of an externality since TCM may not capture the maximum WTP, the choice of value for travel time changes the results, and nonuse values are ignored.
6. **Contingent valuation method:** Establishes a monetary value for an environmental asset by asking people their WTP for that asset. CVM is advantageous because it can include use and nonuse values. On the other hand, this method has numerous biases

and the divergence between willingness to pay and willingness to accept can skew the results.

Appendix A lists the six economic valuation techniques used in this report and details the strengths and weaknesses of each approach.

3.2.3. Marginal User Cost

In this study, MUC represents the cost of foregone future benefits from the overexploitation of groundwater. MUC is specifically relevant to groundwater depletion since present day depletion carries a future opportunity cost, and that opportunity cost must be accounted for in a social costing analysis. The user cost concept has been discussed and applied empirically by numerous authors (OECD 1994; Munasinghe 1990; El Serafy et al. 1989; Repetto et al. 1989). Two commonly used approaches are the net price method (Repetto et al. 1989) and the marginal user cost method (OECD 1994; Pearce and Markandya 1989). The net price method is appropriate when an analysis requires the deduction of user costs at a project or national level. The MUC method is applicable when an analysis is calculating the economic costs of a project output and the user cost has been ignored (FAO 2001). Another approach is the true income approach (El Serafy et al. 1989). This method distinguishes between the total receipts from extraction and depletion of a nonrenewable resource and the true income associated with that nonrenewable resource[12]. This analysis applies the marginal user cost method, since this approach is the most useful for calculating the user costs of water supply projects. The marginal user cost method is estimated as follows (OECD 1994):

$$MUC = (P_b - C)/(1+r)^T \tag{3.2}$$

Where:

MUC = user cost
P_b = price of replacement or backstop technology
C = marginal production/direct costs of existing technology
r = discount rate
T = number of years until the backstop technology replaces the existing technology

[12] The annual earning from sales of a nonrenewable resource includes an income portion, which can be spent on consumption, and a capital element, which should be set aside each year. The capital element of annual earnings should be invested to create a perpetual stream of income that would provide the same level of true income both during the life of a resource as well as after the resource has been exhausted.

MUC, as illustrated in Equation (3.2), is estimated as the present value cost of replacing an environmental asset at some future point and assumes that the direct cost of the existing technology remains constant. The MUC will depend on how strong future demand is relative to today's demand, what substitutes are likely to be available in the future, the cost of the backstop technology, and the discount factor (Pearce and Markandya 1989).

3.3 Evaluation Stance

This analysis calculates the social cost of each supply alternative based on the cost of one cubic meter of freshwater made available by the implementation of a project. The analysis does not produce a value for water and omits discussing the allocation of water across sectors. For this reason, the opportunity cost of water is not relevant to this analysis. In addition, unless otherwise stated, the analysis uses the following assumptions: (1) distribution costs are the same across all projects; and (2) no additional infrastructure is required to accommodate a project. All calculations use values expressed in 1999 U.S.D. Each analytical chapter includes: (1) an introduction that briefly summarizes the chapter; (2) a description of the environmental impacts of the supply option by type of cost; (3) the economic analysis; and (4) a summary and discussion of the results. Where applicable, a sensitivity analysis of the results is provided. Although the MOC framework specifies the use of marginal costs, this analysis uses average costs as a proxy for marginal costs unless otherwise stated. For this reason, the analysis will refer to MDC, MEC, and MUC as direct cost, external cost, and user cost from this point forward.

CHAPTER 4: GROUNDWATER EXTRACTION AND DEPLETION

4.0 Introduction

This chapter estimates the marginal opportunity cost (MOC) of groundwater extraction and depletion following the framework described in Equation (3.1). The first section describes the environmental impacts of depletion and summarizes the valuation methods used for quantifying the direct, external, and user costs. The next three sections estimate each component of MOC. Section 4.5 summarizes the results of the analysis and discusses the implications of these results.

4.1 Impacts of Depletion

The environmental impacts of overpumping apply mainly to the Mountain and Coastal Aquifers, although some of them equally apply to Lake Kinneret. Table 4.1 lists the environmental impacts of depletion in order of importance and in accordance with the MOC framework. Rows 1 to 3 lists the impacts that affect water quantity and row 4 the impact that affects water quality.

Table 4.1: Environmental Impacts of Groundwater Depletion According to the MOC Framework

Environmental Impact	Type of Cost
Depletion: By depleting an aquifer by one unit of water today, that unit of water is no longer available for sale in the future when the price rises (from resource scarcity), representing forgone income to the resource owner.	USER COST
Seawater/freshwater interface: Inland movement of the seawater/freshwater interface occurs as water levels drop (Harpaz 2000). • Irreversible process: reduces the operational capacity of the reservoir. • Only applicable to the Coastal Aquifer. • Dictates the number of years until the aquifer is unusable.	EXTERNAL COST
Saline springs: Changes in pressure from dropping water levels lead to the release of saline springs confined within deep aquifers. • Difficult to predict timing and magnitude of the impact. • Large changes in pressure can lead to irreversible penetrations of saltwater (Isaar 1993).	EXERNAL COST
Drying up of springs: Springs dry up when water levels drop below the points of discharge (Harpaz 2000). • Many nature reserves and ecosystems, some endangered, have been damaged in Israel.	EXTERNAL COST
Anthropocentric pollution: Pollution from human activity above an aquifer causes water quality deterioration. • Leads to well closures and reduces available potable water in an aquifer (Ben Tzi 2001). • Mainly experienced in the Coastal Aquifer.	EXTERNAL COST

Although seawater intrusion is an external cost, it dictates the number of years until the aquifer is unusable. Therefore, it is the focal point of a user cost analysis. Although the release of saline springs can be equally, if not more, damaging, there is too much uncertainty surrounding the prediction of impacts to include them in the analysis. The drying up of springs is an externality imposed on the environment and thus, considered in that context. However, because quantitative estimates of ecosystem degradation are lacking, they are only included qualitatively. Anthropocentric pollution is considered the most severe form of depletion (Ben Tzi 2001); but as it affects water quantity indirectly, it is not explored in this study. The discussion in Section 4.5 reviews the implications of omitting the aforementioned impacts. Table 4.2 describes the valuation methods used to quantify the direct, external, and user costs of groundwater depletion as described in Section 3.2.

Table 4.2: Methods Used for Valuing the Direct, External, and User Costs of Groundwater Depletion

Type of Cost	Valuation Method
Direct Cost	Market prices used to calculate the extraction costs for a typical well in the coastal plain.
External Cost	Ecosystem degradation described qualitatively.
User Cost	Market prices used to calculate the user cost of foregone future benefits using Equation (3.2).

4.2 Direct Cost

Extraction costs are the direct costs (DC) associated with groundwater use and represent the cost of lifting one cubic meter of water from the aquifer source, through a well, and into the national distribution system. The age of the well affects the DC since the capital cost component of construction represents a large proportion of the extraction costs (Arlosoroff 2001). The long-run marginal cost of extraction from the Coastal Aquifer into the public system is U.S. \$0.40/m^3 and the marginal cost of extraction for private wells is U.S. \$0.12/m^3. Public wells supply 65% of domestic water supply and private wells, which are usually local, shallow wells, supply 35% of domestic water supply (Fishelson 1994). Thus, the weighted average of the two marginal costs, U.S. \$0.30/m^3, represents the DC in this analysis.

4.3 External Cost

The main externality of groundwater depletion is ecosystem degradation from the drying up of springs. This impact is well documented since Israel is high in biodiversity and internationally known for its richness in natural vegetation (Frankenberg 1999).

However, because ecosystem degradation is difficult to quantify, a case study of the En Afeq Nature Reserve describes the impacts qualitatively.

The En Afeq Nature Reserve, located in the Western Galilee coastal plain, is one example of a unique and diverse ecosystem. The Nature Reserve contains the last remnant of a former 2,000-hectare swamp, making En Afeq the largest remaining coastal freshwater wetland of Israel. The Israeli government declared En Afeq a nature reserve in 1978 and later, it was proclaimed an international Ramsar site because of its rare and special ecosystem (Ortal 1999). The Nature Reserve receives its water from the Na'aman Springs, which discharges from the Western Galilee Aquifer. In the past, the springs discharged approximately 50-60Mm^3/year. However, because of drought and overpumping of the aquifer, the discharge has dropped to 10% of that amount. In addition, because of freshwater diversions from the underground basin, the average salinity of discharges increased fourfold during the last 50 years (Burgerhart 1999). Water shortages were exacerbated in 1998/9 when a drought caused the water table to drop to an unprecedented level, leaving the Nature Reserve dry for almost three months (Shurky 2000).

Overpumping of the Western Galilee Aquifer has led to ecosystem degradation and has threatened the long-term sustainability of the En Afeq wetland ecosystem (Shurky 2000). Some well-documented changes include the extinction of numerous fish species, a dramatic decrease in migratory birds, and swift changes in vegetation, including the proliferation of invader species more favorable in salty water and arid environments (Arieli 2000)[13]. Further, ecosystem degradation from overpumping occurs in other parts of Israel. Rehabilitation work has begun adjacent to Lake Kinneret where water levels have dropped by a few meters and large areas of land are exposed. However, the

[13] Researchers from Wageningen Agricultural University conducted a vegetation survey to determine the types of vegetation in the reserve, their spatial distribution, the influence of hydrology and grazing on the floristic composition of the vegetation, which species can be used as indicator species, and whether the current management practices are adequate (Burgerhart 1999). In addition, the Nature Authority commissioned other studies in reserve management and drought impacts. However, where available, the results do not provide for an assessment of ecosystem degradation beyond a qualitative description of changes and influences.

fruitfulness of rehabilitation is uncertain and stress on the ecosystem continues, since winter 2000/01 was drier than expected.

4.4 User Cost

Aquifer depletion carries a user cost (UC) because overpumping today creates future foregone benefits. Therefore, this analysis calculates the user cost of depletion in the Coastal Aquifer using the Equation (3.2). This approach requires information on the DC of the current source of supply (C), the price of the backstop technology (P_b), years until the current supply is exhausted (T), and the discount rate (r). The DC (C) is equal to U.S. $0.30/m^3 and is assumed to stay constant over time and the discount rate (r) is equal to the social discount rate of 3%[14]. The following points discuss the other variables.

1. **Price of the Backstop Supply Technique (P_b):** The choice of backstop technology affects the user cost since the price of the backstop is an important variable in the calculation. For this analysis, desalination is the backstop technology since the Israeli government is pursuing desalination as a strategy for future water supply. The social cost (or MOC), of desalination is U.S. $0.83-1.13/m^3 and the analysis uses U.S. $1.00/m^3 as an approximation. Chapter seven provides a detailed explanation of desalination's MOC. This analysis uses the social costs of desalination instead of the private costs because this report is concerned with the social costs of water development and seeks to quantify the costs of each water project from a public planning perspective. Consequently, it would be inappropriate to use private costs.
2. **Number of Years Until the Exhaustion of Groundwater Supplies (T):** Groundwater supplies are exhausted when the seawater/freshwater interface in the Coastal Aquifer moves beyond the predetermined threshold point of 1.5km inland from the seashore. At this point, Israeli hydrologists expect the flow within the aquifer to change (from reduced water pressure) and for seawater to intrude

[14] Extraction costs remain constant over time since the capital cost component of well construction, as opposed to energy, drives extraction costs.

unrestricted and rapidly inland (Harpaz 2001)[15]. The calculation uses the following assumptions:

a. Based on historical monitoring from 1980-85, excess pumping of 70-100Mm^3 resulted in an inland movement of the interface by 30-90m, equivalent to the estimate of an Israeli hydrologist (Harpaz 2001; Nativ and Isaar 1988). In the last few years, extraction from the Coastal Aquifer has been 70-200Mm^3 above renewable recharge (Melloul and Zeitoun 1999). This trend continued through 1999/00 (Israel MOE 2000). Therefore, four scenarios are modeled:
   i. Conservative scenario: The aquifer is overpumped by 70-90Mm^3/year resulting in a movement of the interface by 30m/year.
   ii. Base Case (1): The aquifer is overpumped by 90-110Mm^3/year resulting in a movement of the interface by 60m/year.
   iii. Base Case (2): The aquifer is overpumped by 110-130Mm^3/year resulting in a movement of the interface by 90m/year.
   iv. Accelerated Case: The aquifer is overpumped by 170-200Mm^3/year resulting in a movement of the interface by 180m/year.
b. Based on monitoring results, the maximum seawater intrusion into the aquifer has reached a distance of 0.2-2.0km with the highest level of intrusion found in the Dan Metropolitan Area and Netanya Regions (Melloul and Zeitoun 1999). Thus, three possibilities are modeled within each scenario described in point a:
   i. The interface is 0.2km inland from the coast in 1999.
   ii. The interface is 0.5km inland from the coast in 1999.
   iii. The interface is 1.0km inland from the coast in 1999.

Table 4.3 presents the results of the user cost calculation based on Equation (3.2) and the above considerations. The analysis only includes long-term overpumping of the Coastal Aquifer since seasonal depletion does not affect the interface if winter rains are sufficient for full recharge (Harpaz 2001). In addition, the analysis does not include changes in

[15] The freshwater flow within the aquifer moves from inland towards to sea and maintains aquifer pressure, holding the freshwater/seawater interface in place (Harpaz 2001). Since an aquifer requires many generations for rehabilitation, massive seawater intrusion renders such a basin unusable (Gvirtzman 2000).

rainfall patterns because of climate change. The results outline four scenarios (conservative, base case (1) and (2), accelerated) to account for uncertainty in the parameters. Each scenario lists the number of years (T) until the aquifer is unusable.

Table 4.3: User Cost of Groundwater Depletion (1999 U.S.D./m^3)

Scenario	Freshwater/Seawater Interface Starting Point		
	0.2km	**0.5km**	**1.0km**
Conservative Scenario	\$0.19 T = 43 years	\$0.26 T = 33 years	\$0.43 T = 17 years
Base Case (1) Scenario	\$0.37 T = 22 years	\$0.43 T = 17 years	\$0.55 T = 8 years
Base Case (2) Scenario	\$0.46 T = 14 year	\$0.50 T = 11 years	\$0.59 T = 6 years
Accelerated Scenario	\$0.57 T = 7 years	\$0.59 T= 6 years	\$0.64 T = 3 years

The UC of groundwater depletion is U.S. \$0.19-0.64/m^3 (Table 4.3). However, the MOC of desalination may be undervalued (Section 7.3) and consequently, Table 4.3 may underestimate the user cost. Moreover, the UC directly reflects the cost of the backstop technology and if desalination costs decrease with time (from efficiency gains and research and development), Table 4.3 may overestimate the user cost. In sum, although there are some uncertainties in the figures, they may cancel each other out.

4.5 Summary and Discussion of Results

This chapter estimates the MOC of groundwater extraction and depletion following the framework described in Equation (3.1). Using market prices, a qualitative case study, and the user cost method defined in Equation (3.2), the analysis calculates the direct, external, and user costs of groundwater depletion. Table 4.4 presents the results of the economic valuation.

Table 4.4: Marginal Opportunity Cost of Groundwater Extraction and Depletion (1999 U.S.D.)

Impact	Cost/m^3
Direct Cost	$0.30
External Cost	
Ecosystem Degradation	Negative Impact
User Cost	$0.19-0.64
Total Cost/m^3	**$0.49-0.94**

The social cost of groundwater extraction and depletion ranges from U.S. $0.49-0.94/$m^3$ (Table 4.4). However, some uncertainties exist:

1. The analysis does not quantify ecosystem degradation and studies show that depletion negatively affects nature reserves and ecosystems that rely on spring discharges.
2. The calculation ignores the release of saline springs confined within the Coastal and Mountain Aquifers and anthropocentric sources of pollution from above ground. Anthropocentric sources of pollution alone can reduce potable water supply in the aquifers by up to 90Mm^3/year (Ooku and Abir 2000).
3. The user cost calculation omits the impacts of climate change on rainfall patterns and subsequent aquifer recharge. If predictions about drought periods and strong rains are true, renewable recharge may drop substantially and depletion will accelerate, leading to a higher user cost than represented in Table 4.3.

Because of points 1-3, the figures listed in Table 4.4 represent a minimum estimate of the social cost of groundwater extraction and depletion.

CHAPTER 5: WASTEWATER RECLAMATION AND REUSE IN AGRICULTURE

5.0 Introduction

This chapter estimates the marginal opportunity cost (MOC) of water supplied from wastewater reclamation and reuse following the framework described in Equation (3.1). The first section describes the environmental impacts of effluent reuse in agriculture and summarizes the valuation methods used for quantifying the direct, external, and user costs. The next three sections estimate each component of MOC. Section 5.5 summarizes and discusses the results of the analysis, and presents a sensitivity analysis.

5.1 Impacts of Reusing Treated Wastewater in Agriculture

Reusing treated wastewater in agriculture produces positive and negative impacts, which farmers' actions influence. Table 5.1 lists the environmental impacts of effluent reuse in agriculture according to the MOC framework.

Table 5.1: Environmental Impacts of Reusing Treated Wastewater in Agriculture According to the MOC Framework

Environmental Impact	Type of Cost
Crop Mix: When freshwater is substituted with effluent, farmers may change their crop mix. Crop mix changes are induced by government restrictions on effluent irrigation or crop salt-tolerance levels.	EXTERNAL COST: Behavioral(1)
Fertilizer Inputs: When freshwater is substituted with effluent, farmers may change the quantity of fertilizer applied. • Macronutrient concentrations in the effluent could benefit farmers, depending on the kind of crop grown. • Damages can occur from excess nitrogen.	EXTERNAL COST: Behavioral(1)
Salts: Effluent with elevated levels of sodium, chloride, and boron can reduce plant and soil productivity by: • Altering the electrical conductivity of the soil (osmotic effect). • Changing the sodium adsorption ratio of the soil. • Inducing specific ion toxicity. Salts that leach from the root profile into groundwater basins increase the salinity of drinking water supplies.	EXTERNAL COST
Nitrates/Nitrogen: • When leached into drinking water sources, nitrates can cause human health impacts (Methemoglobinemia, stomach cancer, hypertension in children, and fetal malformations). • Nitrogen contributes to the eutrophication of water sources (Hanley 1989).	EXTERNAL COST
Heavy metals, inorganic compounds, and human health impacts (from pathogens): • Heavy metals and inorganic compounds build up in the soil and groundwater sources and may cause long-term health problems. • Human health impacts from pathogens can occur from physical contact or consumption of products irrigated with effluent (Wallach 1994).	EXTERNAL COST

Remarks: (1) When the Israeli government forces farmers to use treated wastewater instead of freshwater, it is not trying to influence farm production; it is changing water allocation to agriculture. However, when substitution occurs, the farm's production function changes, as per Freeman (1993), making the behavioral response an externality.

The analysis of effluent reuse considers all of the impacts listed above except the effects of heavy metals, inorganic compounds, human health impacts, and the eutrophication of water sources from nitrogen. Although experts consider heavy metals hazardous to human health, Israeli regulations require the separation of industrial effluent from municipal effluent unless it is of similar quality. Since most heavy metals originate from industry, the content of heavy metals in the wastewater stream is low. In addition, activated sludge systems remove most heavy metals from the effluent and divert them to the sludge, which is disposed of separately. Although inorganic compounds, including disinfection byproducts and plasticizers, are known as a problem, no consensus exists on the possible long-term risks (Friedler and Juanico 1996). Human health impacts are omitted since Israeli epidemiological studies concluded that secondary treatment is adequate to prevent the occurrence of disease from pathogens (Avnimelech 1993). Finally, this analysis does not address the eutrophication of water sources since Israel is moving towards 100% reuse of treated wastewater. Therefore, effluent discharges directly into rivers, streams, or the coastal zone will be minimal. Table 5.2 discusses the valuation methods used to quantify the direct, external, and user costs of reusing treated wastewater in agriculture as described in Section 3.2.

Table 5.2: Methods Used for Valuing the Direct, External, and User Costs of Effluent Irrigation

Type of Cost	Valuation Method
Direct Cost	**Additional treatment, distribution, and irrigation costs:** • Market prices used to calculate the treatment costs above those legislated by law for river disposal. • Market prices used to calculate additional distribution and irrigation system costs to prepare effluent for irrigation.
External Cost	**Crop mix:** Market prices used to calculate lost income from crop mix changes because of effluent restrictions. **Fertilizer use:** Market prices used to calculate farm savings from the reduction in fertilizer purchases because nitrogen is in the effluent stream. **Salinity on crops and soil:** • A crop salinity function used to calculate the relative yield decrease of a salt sensitive and a moderately salt sensitive crop when effluent is used in place of freshwater. Market prices used to translate yield decreases into a loss of farm income. • Market prices used to calculate the changes in productivity when soil properties are altered. • The effects of ion toxicity are described qualitatively. **Salts on groundwater:** Market prices used to calculate the cost of desalinating groundwater when effluent irrigation occurs above an aquifer. **Nitrates on groundwater:** Three valuations are undertaken: • Control costs to eliminate nitrogen or nitrates. • Changes in productivity calculated for meeting nitrogen restrictions. • Benefits transfer of contingent valuation (CVM) studies measuring the willingness to pay (WTP) to prevent groundwater contamination.
User Cost	Not applicable.

5.2 Direct Cost

The direct costs (DC) of effluent irrigation represent the treatment costs beyond a secondary level, the additional distribution costs required to separate effluent from freshwater, and the irrigation system costs to adapt farm equipment to lower quality water. First, Israeli water quality regulations require all effluents discharged into the environment to have less than 20mg/l biological oxygen demand and 30mg/l total suspended solids. Secondary treatment, at a cost of U.S. \$0.21/m^3, is adequate to meet these water quality regulations. Therefore, this expense is treated as a sunk cost. However, additional treatment may be needed for unrestricted irrigation, such as tertiary treatment with soil and aquifer treatment (SAT). Second, additional distribution costs are incurred because a different distribution system is required to separate treated sewage from drinking water and additional infrastructure is needed to regulate the year round flow of wastewater and the summer demand for irrigation water. Third, irrigation system costs represent costs to farmers for adapting irrigation equipment and operations to accommodate changes in water quality. The direct cost of effluent reuse is equal to the sum of the cost for treatment beyond secondary treatment, extra distribution costs, and the costs of adapting irrigation systems for changes in water quality.

Table 5.3 presents the DC when treated wastewater is used in place of freshwater. The storage and conveyance costs to move effluent from a treatment plant to seasonal reservoirs and then to farm fields, evaporation losses, and water quality changes from storage represent the additional distribution costs. Filtration and chlorination to prevent blockages in irrigation pipes, additional irrigation maintenance and depreciation costs, additional water for the leaching of excess salts, and soil salinity tests for protection against salt buildup represent the irrigation system costs. Additional treatment costs are the extra cost for tertiary treatment (with SAT) associated with unrestricted irrigation.

Table 5.3: Additional Distribution, Irrigation System, and Treatment Costs from Effluent Irrigation (1999 U.S.D.)

Item	Cost/m^3
Distribution Costs	
Conveyance to storage	\$0.022
Storage (seasonal reservoirs)	\$0.070
Conveyance to fields	\$0.070
10% evaporation loss	\$0.012
Change of water quality	Not available
Follow up and quality control	\$0.012
Total Distribution Costs	**\$0.186**
Irrigation System Costs	
Filtration and chlorination chemicals	\$0.025
Accelerated depreciation	\$0.005
Maintenance	\$0.002
10% of irrigation water	\$0.012
Soil salinity tests	\$0.006
Total Irrigation System Costs	**\$0.05**
Additional Treatment Cost (tertiary)	**\$0.15**

Source: (Haruvy *et al*. 2001)

Table 5.3 summarizes the treatment, distribution, and irrigation system costs associated with effluent reuse. For secondary treated sewage, the relevant costs are distribution and irrigation system costs and the DC equals U.S. \$0.24/m^3. For tertiary treated sewage with SAT, the relevant costs are the conveyance costs (U.S. 0.09/m^3), irrigation system costs, and treatment costs. Since tertiary treatment with SAT stores water in an aquifer, storage costs in seasonal reservoirs are not applicable. Therefore, the DC of effluent reuse using tertiary treated sewage with SAT is U.S. \$0.29/m^3.

5.3 External Cost

This section examines the external costs (EC) of reusing treated wastewater in agriculture as outlined in Table 5.1. First, the analysis explores a farmer's behavioral response to a substitution of freshwater for effluent. Second, the effects of salts on plant and soil productivity and the subsequent loss in farm income are calculated. Third, the contribution of salts to groundwater sources and the need for desalination as a remediation measure are examined. Last, nitrate pollution of groundwater sources is quantified using control costs, changes in productivity, and CVM studies on groundwater protection from other areas of the world.

5.3.1. Behavioral Response

When farmers receive effluent in place of freshwater, numerous behavioral responses may occur, as conceptualized by Freeman (1993). First, to avoid damages from excess salinity, farmers can change the crop mix from salt sensitive to salt tolerant crops. However, since few Israeli farmers crop switch because of salts, the analysis omits this behavioral response (Tarchitsky 2001). Second, since 70% of wastewater in 2005 will be treated to a secondary level or less, farmers may change their crop mix to meet restrictions on effluent irrigation. Third, a farmer can reduce the quantity of fertilizer applied since the effluent stream may contain macronutrients.

Changes in Crop Mix

Secondary treated sewage is restricted to the irrigation of industrial crops, fodder, and nonedible food crops, while tertiary treated sewage with SAT is released for unrestricted irrigation. Thus, farmers cannot grow vegetables eaten raw if they are allocated secondary treated effluent in place of freshwater. In 2005, the Israeli government will allocate 10% of treated wastewater to vegetable irrigation (Hoffman and Harussi 1999). Assuming the effluent is from secondary treatment, farmers must switch from vegetable crops to a field crop, like cotton. Given the financial return of U.S. $1.014/m^3 for vegetables and U.S. $0.322/m^3 for cotton, the loss of farm income per cubic meter of effluent is U.S. 0.692/m^3 (Haruvy and Vered N.d.). Assuming the loss in farm income occurs in 2005 and the social discount rate is 3%, the present value cost per cubic meter of secondary treated effluent is U.S. $0.58.

Changes in Fertilizer Use

Treated wastewater serves a dual purpose for a farmer; it provides a water source and a nutrient source. Unless nutrients are removed during wastewater treatment, the nutrient enriched effluent stream provides a cost savings to the farmer by way of reduced fertilizer requirements. However, wastewater irrigation may damage the crop if there are excess nutrients. Excess nitrogen causes reproductive growth to suffer in crops whose production is based on fruit or seeds, like cotton and citrus. Moreover, since the nitrogen and the effluent stream are inseparable, a farmer must apply nutrients synonymously with irrigation schedules instead of optimum fertilization times, negating some of the nitrogen benefits and contributing to nitrogen damage (Haruvy *et al*. 1999; Avnimelech 1997). Appendix B summarizes the macronutrient availability in secondary treated wastewater and the Israeli Ministry of Agriculture Extension Service's recommendations on fertilizer requirements.

Haruvy *et al*. (1999) studied the benefits and costs of nutrients in effluent irrigation, and found that secondary treated sewage with 40mg/l nitrogen provides a cost savings in fertilizer use of U.S. \$0.012-0.022/m^3 (1999 U.S.D.). The authors calculated these savings using a range of six crops: cotton, corn, avocado, mango, orange, and grapefruit. Accounting for damage from excess nutrients, the cost savings actually range from U.S. \$0.00-0.016/m^3 (1999 U.S.D.), and are negative for some crops. Other studies by Shuval (1997) and Oron and DeMalach (1987) showed that fertilizer savings are approximately U.S. \$0.06/m^3 (1999 U.S.D.). However, this figure was calculated using data from the 1980's and neither study accounted for damages from excess nutrients. Therefore, this analysis adopts Haruvy *et al*. (1999)'s findings and assumes that fertilizer savings from irrigating with secondary treated effluent is U.S. \$0.00-0.016/m^3.

5.3.2. Salinity and Plant/Soil Impacts

Salt accumulation in agricultural crops and soil is a global problem. Since treated wastewater contains approximately 100mg/l of additional salts, impacts occur more rapidly and with greater severity in effluent irrigation. Salt accumulation induces an osmotic effect, changes soil properties, and causes specific ion toxicity. The osmotic effect and changes to soil properties are considered in more detail below. Appendix C provides more detail on the main impacts of salt accumulation.

Osmotic Effect

Salt accumulation reduces the osmotic potential of the soil, harming a plant's ability to absorb water. The osmotic effect is measured from crop salt tolerance. Crop salt tolerance is the plant's ability to endure the effects of excess salt in the soil and is expressed as the relative yield decrease for a given level of soluble salts in the root medium compared with yields under nonsaline conditions (Maas 1990). Maas and Hoffman (1977) developed the following relationship to measure the osmotic effect on plant growth[16]:

$$1\text{-}Y_2/Y_1 = B(EC_x\text{-}A) \tag{5.1}$$

Where:

$1 - Y_2/Y_1$ = relative yield decrease from nonsaline to saline conditions
B = percentage yield decrease from a one unit increase in electrical conductivity above threshold limit
EC_x = electrical conductivity of the soil saturation extract (EC_e) or electrical conductivity of the irrigation water (EC_{iw}) (millimhos per centimeter (mmhos/cm) or decisiemens per meter (dS/m))[17]
A = salinity threshold (mmhos/cm or dS/m)

Using the results from Equation (5.1), the present value loss of farm income from the osmotic effect can be calculated using the following equation:

[16] The study that developed Equation (5.1) evaluated crop responses to salinity under uniform, linear conditions that are rarely achieved in normal field conditions. However, experimental studies have shown that Equation (5.1) can provide an approximate guide (Shalhevet 1994; Dasberg *et al*. 1991; Bielorai *et al*. 1978).

[17] The relationship between EC_e and EC_{iw} is as follows: electrical conductivity of the soil water (EC_{sw}) = $3*EC_{iw}$ and $EC_e = EC_{sw}*0.5$ (Frenkel 1984). Either one is acceptable to use in Equation (5.1). There is no consensus in the literature regarding plant uptake and response to salinity in the root zone. However, in high frequency irrigation, characteristic of many regions in Israel, the zone of maximum water uptake is the upper part of the root zone where the soil is influenced mostly by the salinity of irrigation water (Maas and Hoffman 1976).

$$L = \sum_{t=1}^{n} \{[(P^t - C^t) * (1 - Y_2^t / Y_1^t)] / Q^t\} / (1 + r)^t \quad (5.2)$$

Where:

L = present value loss of farm income (U.S.D./m^3)
P^t = price of crop in time t (U.S.D./hectare)
C^t = farming costs in time t (U.S.D./hectare)
$1\text{-}Y_2/Y_1$ = relative yield decrease from nonsaline to saline conditions in time t
Q = effluent used per hectare in time t (m^3)
r = discount rate (%)
t = year

The parameter estimates for Equation (5.2) are based on Maas and Hoffman (1977) and salinity data from Israel. Maas and Hoffman (1977) specify the crop salt tolerance levels (A) at 1.8 dS/m for grapefruit (a salt sensitive midvalue crop) and 2.5 dS/m for tomatoes (a moderately salt sensitive high-value crop), and the decreases from salt concentrations above the crop threshold (B) at 16% and 9.9% respectively, for grapefruit and tomatoes. The average electrical conductivity of Israeli effluent (EC_{iw}) is 1.5-2.2dS/m (Weber *et al*. 1996). However, since treatment processes do not remove salts, the EC_{iw} of the effluent stream will change depending on the source of the wastewater. Consequently, higher and lower values are possible. The average financial returns per hectare for grapefruit and tomatoes are U.S. $1,340 and U.S. $5,680 and water use per hectare is 7,370m^3 and 5,600m^3 (Haruvy and Vered N.d.).

Table 5.4 provides cost estimates for the osmotic effect on a moderately salt sensitive and a salt sensitive crop, using Equations (5.1) and (5.2). The calculations assume that a percentage yield decline, or percentage decrease in growth, can be applied to income, since this relationship provides the best available approximation of income loss.

Table 5.4: A Quantitative Assessment of the Osmotic Effect on Crop Productivity (1999 U.S.D.)

Crop	Relative Yield Decrease ($1-Y_2/Y_1$)	Loss of Income/m^3 Effluent
Grapefruit	0-24%	$0.00-0.043
Tomato	0-8%	$0.00-0.079

The loss to farm income is up to U.S. $0.08/m^3 when a high value, moderately salt sensitive crop is affected by excess salinity and is up to U.S. $0.043/m^3 when a midvalue, salt sensitive crop is affected (Table 5.4). These total losses are potentially large since 40% of citrus crops and 10% of vegetable crops will be using effluent irrigation by 2005 (Hoffman and Harussi 1999). Further, Equation (5.1) assumes leaching of salts through the soil from heavy winter rains, but this is not always the case in Israel, especially during drought years.

<u>Sodium Adsorption Ratio</u>

The sodium adsorption ratio (SAR) defines the influence of sodium on soil properties by measuring the relative concentration of sodium, calcium, and magnesium. High SAR values can lead to lower permeability and affect soil tilth (Rhoades and Loveday 1990). Although sodium does not reduce the intake of water by a plant, it changes soil structure and impairs the infiltration of water, affecting plant growth (Hoffman *et al*. 1990). Additional impacts include increased irrigation and rainwater runoff, poor aeration, and reduced leaching of salts from the root zone because of poor soil permeability.

Research provides a general scale to measure permeability hazards using SAR and the electrical conductivity of infiltrating water. Figure 2 gives threshold values where permeability hazards are likely or unlikely (Rhoades *et al*. 1992). However, this classification provides no guidance on the costs of reduced permeability and changes to soil properties. Preliminary work by Haruvy *et al*. (2001) quantified the impacts of elevated SAR levels using changes in productivity. Table 5.5 summarizes the impacts and causes of SAR changes, the preliminary costs of those impacts, and the drivers of changes in productivity.

Table 5.5: Preliminary Costs Estimates Associated with Changes in the Sodium Adsorption Ratio (1999 U.S.D.)

Impact	Cause	Cost/m^3	Drivers - Changes in Productivity
Germination problems	Permeability of the top soil	\$0.03	Labor costs and reduced revenues
Yield loss (10-15%)	Increased runoff	\$0.045	Loss of income from reduced yield
Additional leaching (10-20%)	Decreased hydraulic conductivity (poor drainage)	\$0.052	Cost of additional water

Source: (Haruvy *et al*. 2001)

The figures presented in Table 5.5 indicate that the external cost of changes in soil properties from effluent irrigation is U.S. \$0.13/m^3.

Salinity and Plant/Soil Impacts: Conclusion

There are two main impacts of salts on plant and soil productivity. First, the osmotic effect decreases crop yields and the loss of farm income equals U.S. \$0.00-0.08/m^3, depending on the type of crop grown. Second, the sodium content in the effluent stream affects soil properties. The preliminary cost estimate for changes in SAR is U.S. \$0.13/m^3. Therefore, the total cost to farmers from salt impacts on plant and soil productivity is U.S. \$0.13-0.21/m^3. Since wastewater treatment plants cannot remove salts, these costs equally apply to secondary and tertiary treated sewage.

Specific ion toxicity is also an impact of salinity in irrigation water, but this impact is not quantified. A toxicity problem occurs when salt ions accumulate in crops and lead to a reduced crop yield (Ayers and Westcot 1976). Although some herbaceous plants and woody crops are susceptible to specific ion toxicities, the calculation of yield reductions is troublesome since little research exists beyond the quantification of thresholds. Appendix C describes the effects of ion toxicity in more detail.

5.3.3. Salinity and Groundwater

Treated wastewater typically has 100mg/l more salts than freshwater. When farmers apply treated wastewater to crops, some of salts leach into groundwater sources, causing increased salt concentrations in drinking water. Currently, the average annual salinity increases in the Coastal Aquifer is 2-2.5mg/l (Ben Tzi 2001). If effluent irrigation continues, the concentration of chlorides in the Coastal Aquifer may reach the Ministry of Health's legal drinking limit of 250mg/l. Assessing the impacts of groundwater salinization from effluent irrigation requires a comparison of the costs of drinking water supply when effluent irrigation does and does not occur.

A study by Sharon *et al*. (1999) used a portion of the Coastal Aquifer in the Sharon Region of Israel to highlight the costs imposed by effluent irrigation on drinking water supply in a town of 120,000 inhabitants[18]. The authors calculated the costs of municipal water supply using a hydrological-financial model that simulated the movement of water from the farm field to the aquifer, the increase in salt concentrations in the aquifer, and the costs of desalination to meet the 250mg/l legal limit. Since desalination significantly lowers the chloride content of water, the model assumed desalinated water and groundwater are mixed until the combined water quality meets the legal limit.

Table 5.6 summarizes the desalination costs to a representative town from groundwater salinization caused by effluent irrigation. Column one lists the time period in five-year increments. Column two outlines the salinity content of groundwater when initial salinity concentrations are 150mg/l. Column three provides the percentage of groundwater desalinated. Column four details the desalination costs, representing the additional costs for drinking water supplies because of effluent irrigation. Column five shows the percentage increase in aggregate water costs to the town.

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[18] The study uses the Coastal Aquifer as an illustrative example. The same effects are expected in the Mountain Aquifer, with differences attributed to site-specific hydrological characteristics.

Table 5.6: Additional Costs of Water Supply Associated with Groundwater Salinization from Effluent Irrigation (1999 U.S.D.)

Year	Salinity of Groundwater (mg/l chlorides)	Percentage of Groundwater Desalinated	Desalination Costs ($/m^3)	Percentage Increase in Water Cost ($/m^3)
2005	155	0%	$0.00	0%
2010	157	0%	$0.00	0%
2015	168	0%	$0.00	0%
2020	193	0%	$0.00	0%
2025	216	0%	$0.00	0%
2030	237	0%	$0.00	0%
2035	257	7%	$0.01	7%
2040	275	20%	$0.04	23%
2045	292	30%	$0.06	33%
2050	307	36%	$0.07	39%

Source: (Sharon *et al*. 1999)

Table 5.6 indicates that the town incurs additional water costs starting in 2035. The following equation models the increase in present value water costs for the entire nation by generalizing the results from Sharon *et al*. (1999):

$$C = \sum_{t=1}^{n} [(Q_d^t * P^t) / Q_e^t] / (1+r)^t \tag{5.3}$$

Where:

C = present value cost per cubic meter of effluent (U.S.D.)
Q_d^t = quantity of drinking water used by Israeli residential sector in time t (m^3)
P^t = desalination costs per cubic meter of drinking water in time t (U.S.D.)
Q_e^t = quantity of effluent used in Israel in time t (m^3)
r = discount rate (%)
t = year

Equation (5.3) is used to calculate the increase in drinking water costs to Israel from effluent irrigation above aquifer sources, using the year 2040 as an example. In 2040, 12.8 million Israelis will use 1150Mm^3 of drinking water (MWG 1996) and the additional water costs in 2040 for desalination are U.S. $0.04/m^3 (Table 5.6). In addition, Israel will reuse 1070Mm^3 of effluent in agriculture in 2040. Thus, the present value cost per cubic meter of effluent because of desalination costs from groundwater salinization is U.S.

$0.013 using a social discount rate of 3%. The present value cost of drinking water supplies will increase if initial salinity levels in the aquifer are higher than 150mg/l.

5.3.4. Nitrates in Groundwater

Unless nitrification-denitrification (N-D) occurs at the treatment plant, treated wastewater contains nitrogen. The presence of nitrogen is a benefit to farmers since they can reduce their fertilizer use. However, some nitrogen can leach into groundwater sources as nitrates. Nitrate pollution is an important external cost in effluent irrigation because of the human health effects associated with elevated levels of nitrates in drinking water (Wallach 1994). Nitrates leaching into groundwater is important in Israel because more than half the wells in the Coastal Aquifer have nitrate concentrations higher than the European drinking water standard of 45mg/l and 20% of the wells are higher than the Israeli standards of 90mg/l (Haruvy 1997). With secondary treated sewage containing approximately 40mg/l of nitrogen, nitrate levels will continue to increase.

This section examines three valuation methods listed in Appendix A and Section 3.2 for quantifying nitrate leaching from effluent that contains nitrogen:

1. Control costs: The cost of nitrogen removal by N-D at the wastewater treatment plant and the cost of nitrate removal by electrodialysis at the pumping well.
2. Changes in productivity: The loss in farm income from reducing nitrogen applications by one unit, expressed as kilograms of nitrogen per hectare (kg N/ha).
3. Contingent valuation method: CVM studies measure the WTP for groundwater with reduced nitrates, for groundwater with no nitrates, or for the preservation of a groundwater source from pollution.

Of the three methods listed above, control costs and changes in productivity are the preferred method for valuing the impacts of nitrate pollution. Control costs and changes in productivity are based on market values, making these methods more reliable than CVM. On the other hand, CVM includes nonuse values for groundwater protection (Appendix A). Consequently, the results list a range of estimates that includes all three methods.

Control Costs for Nitrogen/Nitrate Removal

Israel uses both N-D and electrodialysis, with N-D occurring at some wastewater treatment plants and electrodialysis occurring at individual wells. When wastewater treatment plants use N-D and supply the effluent to agriculture, a farmer no longer receives the benefit of reduced fertilizer inputs. A farmer continues to apply fertilizer and contributes to nitrate pollution. However, when treatment facilities at the well use electrodialysis, drinking water quality improves. With electrodialysis, a farmer still receives the benefits of nitrogen and the public gets cleaner drinking water, making this approach more attractive. However, as Table 5.7 illustrates, the cost of nitrification-denitrification is 60% lower than the cost of electrodialysis. The implications of this result are discussed in Section 5.4.

Table 5.7: Control Costs of Nitrogen/Nitrate Removal by Treatment Process (1999 U.S.D.)

Treatment Process	Cost/m^3
Nitrification-denitrification[(1)]	\$0.09
Electrodialysis[(2)]	\$0.25

Source: (Haruvy 1997 and Expert Opinion)

Remarks: (1) N-D targets the removal of nitrates and ammonia (80-90%). However, the removal of organic nitrogen is limited in this process (Wallach 1994); (2) Electrodialysis removes 100% of the organic nitrogen, but only 30-50% of the nitrates and ammonia (Wallach 1994).

The control costs for nitrogen or nitrate removal are U.S. \$0.09-0.25/m^3 (Table 5.7). The electrodialysis costs are an approximation since no site-specific data are available. Electrodialysis costs depend on the nitrate reduction required and the size of the plant[19]. N-D costs are based on estimates from tertiary treatment plants that currently use this process.

Changes in Productivity for Nitrogen Restrictions

Since no market estimates exist for the cost of nitrate pollution, changes in productivity from reducing nitrogen applications by one unit (kg N/ha) can provide an estimate for

[19] Although other remedial technologies exist for removing nitrates from drinking water, such as reverse osmosis and ion exchange, treatment facilities in Israel use electrodialysis.

nitrate pollution. This approach assumes lower nitrogen applications will result in less nitrate leaching.

Several studies used changes in productivity to calculate the lost farm income from nitrogen restrictions (Haruvy *et al*. 1997a; Andreasson-Gren 1991). Haruvy *et al*. (1997a) used a linear programming model to calculate changes in agricultural profits in the southern area of Israel from nitrogen restrictions. Andreasson-Gren (1991) calculated the decrease in net farm income caused by a reduction in the application of nitrogen for a coastal bay in Sweden. Although the results from Andreasson-Gren (1991) provided detailed costs for eliminating nitrogen inputs, the author reported the results in a manner that allows for comparison. Moreover, since Haruvy et al. (1997a) used Israeli data, this analysis uses Haruvy *et al*. (1997a)'s results. They defined the cost of nitrogen restrictions as the lost income per unit of nitrogen (kg N/ha) reduced expressed in cubic meters of applied effluent[20]. The authors calculated the cost of reducing nitrogen inputs from 25kn N/ha to 15kg N/ha at U.S. \$0.11-0.14/m^3 (1999 U.S.D.).

Contingent Valuation Method and Benefits Transfer

A benefits transfer is "the application of monetary values obtained from a particular nonmarket goods analysis to an alternative or secondary policy setting" (Brookshire and Neill 1992). Benefits transfer is useful for valuing nitrate reductions since there are no specific data available for the study area, and a full-scale valuation study is outside the scope of this project. Appendix D summarizes nine contingent valuation studies from United States and Europe, to provide a cross section on the values of groundwater protection from nitrates and other pollutants. Table 5.8 provides a brief summary of the study results in Appendix D.

[20] The study assumed one cubic meter of wastewater has 51mg/l of nitrogen.

Table 5.8: Contingent Valuation Studies on Groundwater Protection from Nitrates and other Pollutants (1999 U.S.D./household/year)

Study Source	Study Site	Mean WTP
Poe (1998)	Wisconsin	$212
Stenger and Willinger (1998)	France	$110-$128
Crutchfield et al. (1997)	Indiana, Nebraska, and Washington	$607-$876
Powell et al. (1994)	Massachusetts, Pennsylvania, and NY	$70
Jordan and Elnagheeb (1993)	Georgia	$148
Sun et al. (1992)	Georgia	$861
Shultz and Lindsay (1990)	New Hampshire	$164
Hanley (1989)	England	$30
Edwards (1988)	Massachusetts	$2,285

The WTP estimates range from as low as U.S. $30/household/year to as high as U.S. $2,285/household/year (Table 5.8). Some of the variability is attributable to differences in the explanatory variables, like income, which is statistically significant in almost all studies. The rest of the variability is attributable to survey-specific variables including: definition of groundwater contamination, information in the survey instrument, respondent's knowledge of the problem, the payment vehicle, and the variables regressed. See Boyle *et al*. (1994) for the results of a meta-analysis on groundwater valuation studies, which included many of the studies listed in Table 5.8[21].

Benefits function transfer (BFT) is a more sophisticated approach to transferring WTP estimates. BFT transfers the entire demand function from the study site to the policy site, and many experts describe it as preferable to benefits transfer (Downing and Ozuna 1996; OECD 1994). However, the studies in Appendix D do not allow for a proper transfer of the demand function since: (1) some authors did not report the regression results properly, (2) Israeli data for <u>all</u> the variables regressed are not available, and (3) the authors are measuring different types of groundwater protection. Therefore, Table 5.9

presents the second-best approach by listing a subset of Appendix D. This table summarizes the studies most similar to Israel not only in the definition of groundwater, but also in the explanatory variables. In each study, the mean income per household was statistically significant and $\pm$15% the mean income of the average household in Israel. Furthermore, each study modeled a reduction in nitrate pollution to meet the standard of 45mg/l nitrates (equivalent to 10mg/l nitrogen). The results in Table 5.9 are reported in WTP per cubic meter of water and assume that 1.6 million Israeli households consume 1000Mm^3 of groundwater each year.

Table 5.9: Subset of WTP Estimates for Groundwater Protection from Nitrate Contamination (1999 U.S.D.)

Study	Mean WTP/m^3/Year	Value Measured in CVM Study
Jordan and Elnagheeb (1993)	\$0.24	Improvements in drinking water quality to meet nitrogen standard of 10mg/l
Poe (1998)	\$0.34	Protection of well water to <10mg/l when the probability of water being >10mg/l is 50%[22].
Mean WTP	**\$0.24-0.34**	

Although BFT is the preferred valuation technique, Table 5.9 illustrates that when variables are more strictly controlled, a convergence of WTP figures is possible. In addition, the figures in Table 5.9 are similar to electrodialysis costs, providing additional consistency to the results.

Nitrates in Groundwater: Conclusion

The analysis uses three valuation techniques to quantify groundwater contamination from nitrates. The control costs range from U.S. \$0.09-0.25/m^3. The loss of farm income from changes in productivity due to nitrogen restrictions is U.S. \$0.11-0.14/m^3. The WTP for a reduction in nitrate concentrations to 45mg/l ranges from U.S. \$0.24-34/m^3. Therefore,

[21] Boyle et al. (1994) concluded that despite the limitations of each study, the variations in WTP are not random and estimates reflect systematic differences in groundwater values. In addition, value differences could be more clearly identified by future improvements in groundwater valuation studies.

[22] Since 50% of Coastal Aquifer wells have nitrate levels above 45mg/l, this probability is appropriate.

this analysis uses an estimate of U.S. \$0.09-0.34/m^3. This estimate only applies to secondary treated sewage since tertiary treatment facilities remove nitrogen.

5.4 User Cost

Wastewater reclamation and reuse in agriculture has no user cost since using treated wastewater today does not preclude the use of that portion of the treated wastewater tomorrow. Therefore, there are no costs of future foregone benefits and a discussion of user cost is not applicable for this water supply option.

5.5 Summary and Discussion of Results

This chapter estimates the MOC of wastewater reclamation and reuse following the framework described in Equation (3.1). Using the valuation techniques described in Section 3.2, the analysis calculates the direct, external, and user costs of effluent reuse. Table 5.10 presents the results of the economic valuation. The cost estimates are broken out by treatment process since the impacts of effluent irrigation using secondary treated sewage differ from tertiary treated sewage.

Table 5.10: Marginal Opportunity Cost of Wastewater Reclamation and Reuse in Agriculture (1999 U.S.D./m^3)

IMPACT	Cost - Effluent Irrigation with Secondary Treated Sewage	Cost - Effluent Irrigation with Tertiary Treated Sewage
Direct Cost		
Additional Treatment	*Not applicable*	*$0.15*
Additional Distribution, and Irrigation System Costs	*$0.24*	*$0.14*
Total Direct Cost	**$0.24**	**$0.29**
External Cost		
Crop Mix Changes	*$0.58*	*Not applicable*
Fertilizer Use	*$0.00-(0.016)*	*Not applicable*
Salinity on Crop Productivity	*$0.00-0.08*	*$0.00-0.08*
Ion Toxicity	*Negative Impact*	*Negative Impact*
Sodium Adsorption Ratio	*$0.13*	*$0.13*
Salinity on Groundwater	*$0.013*	*$0.013*
Nitrates on Groundwater	*$0.09-0.34*	*Not applicable*
Total External Cost	**$0.80-1.14**	**$0.14-0.22**
User Cost	None	None
Total Cost/m^3	**$1.04-1.38**	**$0.43-0.51**

The social cost of effluent irrigation ranges from U.S. \$1.04-1.38/m^3 for secondary treated sewage and U.S. \$0.43-0.51/m^3 for tertiary treated sewage (Table 5.10). Tertiary treated sewage has lower costs since there are no storage costs, irrigation restrictions, or impacts from nitrogen concentrations. Thus, it is cheaper from a social perspective for the Israeli government to use tertiary treatment for wastewater allocated to agriculture, even though the private costs of tertiary treatment are U.S. $0.05/m^3 higher than secondary treatment. Chapter seven discusses this point in more detail.

Table 5.10 represents a minimum estimate of the social cost of wastewater reclamation and reuse in agriculture for the following reasons:

1. The cost of land for additional treatment facilities (i.e. tertiary treatment) is omitted from the DC of wastewater treatment.
2. The analysis does not calculate voluntary crop switching to avoid the osmotic effect.
3. If the soil does not leach salts in the winter months, the osmotic effect in the next growing season is more severe and farm income is reduced further.
4. The analysis excludes the effect of specific ion toxicity.
5. The desalination costs from groundwater salinization will increase if the salinity content in the effluent stream continues to rise, or if the initial groundwater salinity levels are higher.
6. The figures associated with tertiary treatment, which includes N-D, underestimate the true impact of agricultural practices since farmers continue to use fertilizer. However, unless the nitrogen is already in the irrigation water, fertilizer applications are an externality of agricultural practices and not effluent irrigation. If the scope of this analysis was broadened to include all agricultural practices, electrodialysis becomes a more attractive option than N-D because it allows treatment plants to forgo N-D and allows farmers to apply fertilizer, while still providing the public with nitrate free drinking water.

Given the uncertainties described above, Table 5.11 presents a sensitivity analysis that measures the effect of a change in direct or external costs on the MOC of secondary and tertiary treated effluent. The base case represents the values used in the analysis. Table 5.11 models all the impacts of effluent reuse except ion toxicity, fertilizer benefits, and nitrate pollution. The analysis omits ion toxicity because there are no quantitative estimates for this impact. Fertilizer benefits and nitrate pollution are ignored because they already have a range of estimates and therefore, a sensitivity analysis on these variables is not necessary.

Table 5.11: Sensitivity Analysis of Wastewater Reclamation and Reuse in Agriculture (1999 U.S.D./m^3)

Variable Analyzed	Cost of Variable Analyzed	MOC Effluent Irrigation (Secondary Treatment)	MOC Effluent Irrigation (Tertiary Treatment)
Direct Cost: Treatment Cost (TC)			
TC = Base Case	$0.15	N/a	$0.43-0.51
TC = +15%	$0.17	N/a	$0.45-0.53
TC = -15%	$0.13	N/a	$0.41-0.49
Direct Cost: Distribution Irrigation Cost (D/I)(1)			
D/I = Base Case	$0.24/0.14	$1.04-1.38	$0.43-0.51
D/I = +15%	$0.28/0.16	$1.08-1.42	$0.45-0.53
D/I = - 15%	$0.20/0.12	$1.00-1.34	$0.41-0.49
External Cost: Crop Switching			
Crop Switching	$0.58	$1.04-1.38	N/a
No Crop Switching	$0.00	$0.46-0.80	N/a
External Cost: Osmotic Effect			
EC_{iw} *= 1.5-2.2 dS/m (Base Case)*	$0.00-0.08	$1.04-1.38	$0.43-0.51
EC_{iw} *= 0.5 dS/m*	$0.00	$1.04-1.30	$0.43
EC_{iw} *= 4.0 dS/m*	$0.06-0.35	$1.10-1.65	$0.49-0.78
External Cost: Sodium Adsorption Ratio			
SAR = Base Case	$0.13	$1.04-1.38	$0.43-0.51
SAR = +15%	$0.15	$1.06-1.40	$0.45-0.53
SAR = -15%	$0.11	$1.02-1.36	$0.41-0.49
External Cost: Salts in Groundwater(2)			
Salinity: 150mg/l (Base Case)	$0.013	$1.04-1.38	$0.43-0.51
Salinity: 250mg/l	$0.03	$1.06-1.40	$0.45-0.53
Salinity: 450mg/l	$0.04	$1.07-1.41	$0.46-0.54

Remarks: (1) The distribution and irrigation systems costs differ for secondary and tertiary treated sewage. Therefore, column two reports the costs for secondary treated sewage first, followed by tertiary treated sewage; (2) Sharon et al. (1999) calculate the desalination costs when initial salinity levels in the aquifer are 250mg/l and 450mg/l.

The MOC of effluent irrigation with secondary and tertiary treated sewage is sensitive to changes in cost estimates of the osmotic effect and crop switching (Table 5.11). If the salinity content in the effluent stream is 50% higher than the base case (i.e. EC_{iw} = 4 dS/m), the MOC for reusing secondary treated sewage for irrigation increases by up to 20% and the MOC of reusing tertiary treated sewage for irrigation increases by up to 53%. If farmers did not crop switch because of effluent restrictions, the MOC of secondary treated sewage decreases by approximately 40-60%. This point is discussed in more detail in Chapter seven. The remaining variables do not have a large effect on the MOC of effluent reuse in agriculture.

CHAPTER 6: DESALINATION

6.0 Introduction

The third water project under consideration is desalination and this chapter estimates its marginal opportunity cost (MOC) following the framework described in Equation (3.1). The first section describes the environmental impacts of desalination and summarizes the valuation methods used for quantifying the direct, external, and user costs. The next three sections estimate each component of MOC. Section 6.5 summarizes and discusses the results of the analysis, and presents a sensitivity analysis.

6.1 Impacts of Desalination

Table 6.1 lists the most important environmental impacts of desalination classified according to the MOC framework.

Table 6.1: Environmental Impacts of Desalination According to the Marginal Opportunity Cost Framework

Environmental Impact	Type of Cost
Energy: Burning fossil fuels to generate power for desalination plants impacts: • Human health • Climate change • Agricultural crops, forests, biodiversity, noise levels, and causes material damages to monuments and historical sites These externalities are associated with all energy uses, but are particularly high in this analysis because of reverse osmosis' (RO) energy intensity.	EXTERNAL COST
Land-use: Land-use impacts relate to the loss of the open seashore for construction of desalination plants[23].	EXTERNAL COST
Brine discharge to the Mediterranean Sea: Rejected brine contains chemicals like antiscalants and washing solutions. Brine discharges may affect marine life.	EXTERNAL COST

[23] Desalination plants do not need to be located along the seashore. However, access to the coast reduces costs since seawater is readily accessible.

With five kilowatt-hours (kWh) of energy required for each cubic meter of desalinated water, energy is the most important externality of the desalting process. Furthermore, Israel will use coal-fired power plants to generate energy for desalination facilities. However, within the discussion of energy externalities, the analysis only examines human health and climate change impacts from a national perspective. Because the impacts on agriculture, forests, biodiversity, noise, and material damages are poorly understood or poorly documented, they are omitted. The analysis also examines land-use impacts given the value of the Israeli seashore. Brine discharge is discussed qualitatively since its effects on marine life are poorly understood. Table 6.2 discusses the valuation method used to quantify the direct, external, and user costs of desalination as described in Section 3.2.

Table 6.2: Methods Used for Valuing the Direct, External, and User Costs of Desalination

Type of Cost	Valuation Method
Direct Cost	Market prices used to calculate the treatment costs for a RO desalination facility.
External Cost	**Energy Externalities:** • Human health impacts calculated via benefits transfer from dose-response functions developed in other parts of the world. • National impacts of climate change are described qualitatively. **Brine discharge:** Described qualitatively. **Land-use:** Contingent valuation method (CVM), travel cost method, and market prices used to calculate the value of beach access for recreation and the preservation of the open seashore.
User Cost	Not applicable

6.2 Direct Cost

Desalination costs have dropped rapidly over the last decade with research and development creating processes that are more efficient. The costs of desalinating seawater are now U.S. \$0.70-0.80/m^3 and the costs of desalinating brackish water are

U.S. $0.20-0.35/m^3$ (Priel 2001; Semiat 2000)[24]. Table 6.3 illustrates a breakdown of the direct costs (DC) of desalting seawater using RO technology. The figures do not include the costs of transmission line construction to the plant.

Table 6.3: Direct Costs of a 50Mm^3/year Reverse Osmosis Desalination Plant (1999 U.S.D./m^3)

Category	Percentage of Cost	Optimistic Estimates	Conservative Estimates
Electric Power[(1)]	44%	$0.32	$0.36
Fixed Charges[(2)]	37%	$0.27	$0.30
Maintenance and Parts	7%	$0.05	$0.06
Membrane Replacement	5%	$0.04	$0.04
Supervision and Labor	4%	$0.03	$0.03
Chemicals	3%	$0.02	$0.02
Total		**$0.73**	**$0.81**

Source: (Priel 2001 and Semiat 2000)

Remarks: (1) The average price of electricity for industrial clients of the Israeli Electric Corporation in 1997 was approximately U.S. $0.06/kWh; (2) Based on a 20-year plant life and an interest rate of approximately 6%.

The DC of desalination are U.S. $0.73-0.81/m^3$ (Table 6.3). However, this figure may be undervalued for various reasons. First, Table 6.3 does not quantify the cost of brine disposal from a plant site because estimates are not available[25]. Second, the cost of land may not be included in the fixed charges and land has an opportunity cost. Wastewater treatment plants do not pay for the cost of land and therefore, it is possible that desalination plants are also not required to do so. Even if the cost of land is included in the estimates, the cost may not incorporate a premium for coastal land[26]. According to a recent study of coastal land values, the seashore increased property values by 30% (Israel MOE 1999a). This point is addressed in the sensitivity analysis in Section 6.5. Last, the energy price is based on the average price of electricity charged to industrial clients by

[24] The costs of desalting brackish water are omitted from this analysis because large-scale desalination in Israel during the next five years will focus mainly on seawater desalination.

[25] Brine disposal includes the cost of moving brine from a plant site to a disposal site. The effects of brine discharge are the negative externalities associated with dumping brines into the natural environment (i.e. disposal site).

[26] Desalination plants could be sited further inland. Decision makers would need to consider the extra cost of piping seawater further inland versus the costs of denying beach access to the public.

the Israeli Electric Corporation, a state monopoly. Thus, it may be undervalued if it includes subsidies. Alternatively, if a desalination plant can secure energy at a reduced price because of bulk purchases, the average energy price may be overvalued. Therefore, the two distortions may cancel each other out. This analysis assumes the minimum DC of desalination is U.S. \$0.73-0.81/m^3.

6.3 External Cost

The most important externalities associated with desalination are energy, land-use impacts, and the effects of brine discharge. This analysis addresses all three impacts, but does not quantify the external cost (EC) of brine discharge since estimates are not available. Energy and land-use issues are examined in detail because they have substantial impacts and a vast amount of research has gone into quantifying their damages.

6.3.1. Energy Externalities

Desalination uses 5kWh of electricity to desalinate one cubic meter of seawater, and Israel will likely use coal-fired power plants to generate this energy. As a result of the large electricity requirements, the impacts of energy are an important externality. For fossil fuel chains, most EC come from air pollutants emitted by power plants, as opposed to upstream or downstream activities like coal mining and waste disposal. The main impacts associated with fossil fuel production are on human health and climate change (DGXII 1995b). Human health impacts stem from the detrimental effects of emissions released during the operation of a power plant and are broken down into two costs: morbidity costs from illness because of chronic exposure, and mortality costs (Friedrich and Voss 1993). Climate change, despite the great range of uncertainty, is among the most serious side effects of fossil fuel power stations (Kollas 2000).

The analysis of energy externalities examines three dose-response studies for the quantification of human health impacts of air pollution from coal-fired power plants. In addition, the analysis summarizes a CVM study by Shechter (1992), which measured the willingness to pay (WTP) for clean air in the city of Haifa in 1986/7. This analysis cites two valuation approaches for energy externalities because of uncertainty surrounding the study estimates. The impacts of climate change are also introduced and discussed based on their relevance to Israel, but they are not quantified.

Human Health Impacts: Dose-response Function

The first major effort to quantify the externalities of energy began in 1988 and the methods of valuing energy externalities have become more sophisticated and accurate with time. The current approach is the dose-response function. The procedure includes the following steps (Freeman 1996):

1. Estimate emissions and other environmental stresses of the technology/fuel type.
2. Estimate changes in environmental quality as a function of emissions.
3. Estimate the physical effects of changes in environmental quality on the receptors.
4. Apply unit values to convert physical effects to monetary damages for each endpoint.
5. Aggregate damages across all receptors and endpoints.

Between 1991 and 1996, five major studies were completed using the dose-response approach. Each study provided estimates for some of the external environmental costs of adding capacity to an electricity generation system, based on the next or marginal plant (Freeman 1996). Of these five studies, the EU ExternE, Department of Energy (DoE), and New York are distinguished by their magnitude of effort, comprehensiveness of the analyses, and extensiveness of peer review (Krupnick and Burtraw 1996). For these reasons, they are the focal point of the analysis. Table 6.4 summarizes the cost estimates of each study and Appendix E describes the projects in detail.

Table 6.4: Cost Estimates of Human Health Impacts from Energy Externalities (1999 U.S.D.)

Study	Cost Estimate/kWh
ExternE	$0.018-0.033
DoE	$0.001
New York	$0.003-0.0042

Source: (Krupnick and Burtraw 1996; DGXII 1995b)

Table 6.4 reports cost estimates for human health impacts from U.S. $0.00-0.033/kWh. One explanation for the divergence is that the U.S. figures may be low because of strict U.S. regulations for power generation. Appendix E, Tables E.1 and E.3, illustrates this point with the particulate matter (PM) emissions per kWh in the U.S. being much lower than at European locations.

Of the three sets of figures, the ExternE studies appear the most consistent with Israeli conditions. First, PM emissions from Germany are the same as Israel (Appendix E, Table E.1) and PM causes most human health impacts. Second, the Spanish and Greek climates are Mediterranean, and therefore, the atmospheric conditions are similar to those of Israel. Although these explanations do not eliminate all the uncertainty, this analysis assumes that U.S. $0.02-0.03/kWh can proxy as a reasonable figure for the human health impacts of energy production from coal.

Human Health Impacts: Contingent Valuation Method

The only major valuation study conducted in Israel to measure health impacts from air pollution took place in the city of Haifa in 1986/7 (Shechter 1991). The study selected Haifa because it is an industrial city with high concentrations of heavy industry, including a power plant and oil refinery. In addition, the topography and meteorological conditions of the city created conditions conducive to pollution retention in parts of the metropolitan areas (Shechter 1992). The investigation was based on a survey of 3500 households and applied various valuation techniques to determine the value of air quality in Haifa. Table 6.5 summarizes the results of the contingent valuation and dose-response approaches. Since CVM measured the WTP to reduce the disutility associated with morbidity/mortality, and the dose-response function measured the cost of illness (COI), including payments for health visits, the CVM and COI valuations are additive (Shechter 1991).

Table 6.5: Valuation Results for Air Pollution in Haifa, Israel (1999 U.S.D.)

Valuation Technique	Procedure	Annual WTP per Household
Contingent Valuation Method[(1)]	WTP to prevent a 50% reduction in air quality. Payment vehicle: municipal property tax	\$286-397
Dose-response Function: Cost of Illness	Measured health care expenditures and the value of lost production, given a dose-response relationship between excess morbidity/mortality and pollution levels.	\$825

Source: (Shechter 1992)

Remarks: (1) The public was aware of air pollution-induced morbidity since articles were published in the local press dealing with air pollution during the 12-month period corresponding to the duration of the survey. Results summarize surveys that used open ended, bidding, and dichotomous choice elicitation techniques.

Table 6.5 lists the annual WTP per household to prevent a 50% reduction in air quality. Translating these results to the entire country and to a cost per kWh, the average WTP is estimated at U.S. \$0.014-0.02/kWh. This figure assumes 1.6 million Israeli households and 33.6 billion kWh of electricity generation a year (IEC 1998). Adding the COI measure to the WTP figures increases the cost by US \$0.04/kWh to US \$0.054-0.06/kWh. However, more than one desalination plant would be required to use enough electricity to induce a 50% reduction in air quality. Consequently, U.S. \$0.054-0.06/kWh is likely an over estimation for this analysis. This point is addressed in the sensitivity analysis in Section 6.5

<u>Climate Change</u>

Desalination facilities contribute to climate change by demanding electricity generated with fossil fuels. A 50Mm^3 desalination plant demands 250 million kWh of electricity. Most studies on energy externalities do not account for greenhouse gas emissions and the effects of climate change because damage estimates in the literature are highly uncertain.

However, if climate change damages prove to be large, an analysis that omits them will be highly misleading (Freeman 1996; Krupnick and Burtraw 1996). Thus, Table 6.6 lists some estimates of global warming impacts. The results may be inaccurate or incomplete and a range of error is expected (Frankhauser and Tol 1996).

Table 6.6: Recommended Estimates of Climate Change Damages (1999 U.S.D./kWh)

Study[1]	Low	Mid	High
ExternE: Greece[2]	$0.006	$0.03-$0.08	$0.23
ExternE: Spain[2]	$0.005	$0.03-$0.06	$0.18
Cline 1992	$0.0009	$0.003	$0.02
Frankhauser 1993	$0.0006	$0.002	$0.02
Tol 1994	$0.004	$0.017	$0.03
Hoymeyer and Gartner 1992	$0.28	$1.14	$7.47

Source: (DGXII 1995a, 1995b)

Remarks: (1) The research conducted on the damages of climate change assumes atmospheric carbon dioxide concentrations increase to twice the preindustrial level (Frankhauser and Tol 1996). In addition, the data do not represent possible surprises and catastrophes, which could greatly increase the impacts (Eyre 1997); (2) The ExternE studies are based on the results of the FUND model and use the following estimates to calculate global warming damages in all European countries: (a) Low (10% discount rate) 3.8 European Currency Units per ton of carbon dioxide (ECU/t CO_2) emitted, (b) Mid (3% discount rate) 18 ECU/t CO_2 emitted, (c) Mid (1% discount rate) 46 ECU/t CO_2 emitted, (d) High (0% discount rate) 139 ECU/t CO_2 emitted. However, because of uncertainty in the estimates, the ExternE study omitted them from the final analysis. 1 ECU = 1.25 U.S.D.

Climate change costs are between U.S. $0.00-$7.47/kWh (Table 6.6). This range is too large to provide any useful insight into the EC of climate change. Moreover, the estimates in Table 6.6 represent the <u>global</u> impacts of climate change. However, this report outlines the <u>national</u> costs to Israel for water supply development. Therefore, the figures are not consistent with this analysis. However, climate change will cause impacts to Israel through, for example, changes in weather patterns and extreme events. Unfortunately, the value of climate change impacts specific to Israel is not known.

6.3.2. Land-use Externality

Israel is a coastal nation with 70% of the country's residents living along its 188-kilometer coastal strip (Israel MOE 1999a). Since the coastal area is the main center of economic activity, changes in urban settlements, industry, energy, tourism, and transport activities are likely to have significant impacts. In recent times, urban and economic pressures for development, coupled with coastal attractions for tourism and recreation, have exacerbated conflicts along the Mediterranean shore.

According to Israeli planners, a new $50Mm^3$ desalination plant will be located along the coast for easy access to seawater, and will require $40{,}000m^2$ of land (Hoffman N.d.). As a result, the public will lose access to approximately 200m of coastline. Given coastline scarcity in Israel, and the public benefits of the seashore to the public, denying beach access creates a negative externality.

The Israeli Ministry of Environment conducted an economic valuation of the Mediterranean coast using the travel cost method, CVM, and market prices to measure the value of beach as a site for public recreation and leisure and the value of open seashore to the Israeli public (Israel MOE 1999a). Table 6.7 reports the results of the economic valuation. The values measured are listed in column one and described in column two. Column three lists the total cost per year to the Israeli public and column four reports the cost per cubic meter of desalinated water. The following equation calculates the cost per cubic meter of desalinated water for loss of beach access:

$$C = [(TC/b)*a]/Q \qquad (6.1)$$

Where:

C = cost per cubic meter of water desalinated (U.S.D./m^3)
TC = total cost per year for the loss of beach access (U.S.D.)
b = municipal regulated shoreline (km)
a = beach access lost for the construction of a desalination plant (km)
Q = quantity of water desalinated (m^3)

Israel has 24km of regulated beaches and this analysis assumes that the Israeli government will allocate land for a desalination plant within these 24km. Moreover, a $50Mm^3$ desalination plant cuts off 200 meters of coastline.

Table 6.7: Economic Valuation of the Israeli Coastline for Public Recreation (1999 U.S.D.)

Value Measured	Procedure and Assumptions	Total Cost/Year(1)	Cost/m^3(2)
Public recreation and leisure	• Vacationers and bathers surveyed between 1982 and 1994 by aerial photography at noon on Saturday in the month of August. • Price for entry to beaches, travel costs, parking costs, and municipal expenditures for maintaining beaches examined. • Consumer surplus estimated at 70% of the public outlay for beach recreation.	126 million	$0.021/$m^3$
Value of the open seashore	• Survey of 306 residents. • Respondents asked for their WTP to conserve the seashore. • 1.6 million households in Israel.	12.75 million	$0.002/$m^3$

Source: (Israel MOE 1999a)
Remarks: (1) 4NIS = 1U.S.D.; (2) The current valuation of shoreline loss does not include the visual damage imposed on society for desalination plants built adjacent to recreational beaches.

The EC of shoreline loss is U.S. $0.002-0.02/$m^3$ (Table 6.7). This figure represents the value of open seashore to the Israeli public and the value of the beach as a site for public recreation and leisure. In addition, Table 6.7 assumes that the Israeli government would have preserved the land used for desalination plants as recreational space within the 24km of regulated bathing beaches and that the externality value would increase as shoreline scarcity grows.

6.3.3 Brine Discharge

Rejected brine is a byproduct of the desalination process. Brine discharge is twice the concentration of seawater and contains chemicals like antiscalants, used in the pretreatment of the feed water, washing solutions, and rejected backwash slurries from the feed water. In large-scale desalination processes, brine discharge may detrimentally affect marine life. However, in smaller quantities, dilution and spreading can mitigate this effect and solve the problem. Furthermore, natural chemicals that do not harm the environment may replace synthetic chemicals in future (Semiat 2000). The issue is more serious when the desalination facilities are located inland. In sum, brine discharge, in large enough quantities (whether inland or by the coast), will likely cause externalities (Semiat 2000). However, the magnitude of impact is uncertain and cost estimates are not available.

6.4 User Cost

Desalination has no user cost since using desalinated water today does not preclude the use of that portion of the desalinated water tomorrow. Therefore, there are no costs of future foregone benefits and a discussion of user cost is not applicable for this water supply option.

6.5 Summary and Discussion of Results

This chapter estimates the MOC of desalination following the framework described in Equation (3.1). Using the valuation techniques described in Section 3.2, the analysis calculates the direct, external, and user costs of desalination. Table 6.8 presents the results of the economic valuation. The energy externalities for human health include the valuation results from the ExternE studies and the CVM and COI results from Shechter (1991).

Table 6.8: Marginal Opportunity Cost of Desalination (1999 U.S.D./m^3)

Impact	Cost
Direct Cost	\$0.73-0.81
External Cost	
Energy Externality: Human Health	*\$0.10-0.30*
Energy Externality: Climate Change	*Negative*
Land-Use Externality	*\$0.00-0.02*
Brine Discharge	*Negative*
Total External Cost	**\$0.10-0.32**
User Cost	None
Total Cost/m^3	**\$0.83-1.13**

The social cost of desalination ranges from U.S. \$0.83-1.13/m^3 (Table 6.8). However, this figure represents a minimum estimate for a variety of reasons:

1. The price of energy affects the cost per cubic meter of desalinated water. Since Israel imports its fossil fuels, increases in the world price of coal will increase the cost of desalination. In addition, the price of energy inputs may be distorted because it is based on the average costs charged by a state monopoly and may underestimate or overestimate the true economic cost of energy production.
2. This analysis does not include the costs of brine disposal, brine discharge, and transmission line access to the desalination plant.
3. The ExternE valuation yields a cost per kWh that is too low. First, the ExternE study examined the next or marginal plant. Because of cleaner technologies, these plants emit less pollution than existing coal, oil, or gas-oil power plants[27]. Second, Israel's sulfur dioxide emissions are higher than the ExternE locations. Third, the ExternE researchers quantified the impacts that they had the ability to quantify. Experts know many human health impacts exist, but not enough epidemiological research is available to monetize the effects. By omitting these impacts, the analysis values them at zero by default.
4. This analysis omits the impacts of energy production on agricultural crops, forests, biodiversity, noise levels, and material damages to monuments and historical sites.
5. Climate change predictions indicate the possibility of more severe droughts and extreme weather events. Moreover, catastrophes and possible surprise events are to be expected. The analysis ignores such impacts and they will increase the cost per cubic meter of desalinated water.
6. As shoreline property becomes scarcer, the opportunity cost of land increases. Therefore, the direct costs and land-use externality from public uses of the beach will increase with time. If desalination plants are located inland from the cost, the opportunity cost of denying shoreline access is not relevant. Further studies are needed to estimate the additional piping costs to move seawater inland versus the land-use externality from shoreline loss.

The MOC of desalination may also decrease with time, since research and development are continually creating processes that are more efficient. Recently, the Israeli government awarded a contract for the first $50Mm^3$ desalination facility. Freshwater from this desalination plant will be produced privately and sold to the Israeli government at a cost of U.S. $0.53/m^3$, substantially lower than any previous estimate (Hoffman 2001).

Given the uncertainties described above, Table 6.9 lists the results of a sensitivity analysis. The base case represents the values used in the analysis. Table 6.9 models environmental impacts of desalination except brine discharge and desalination's contribution to climate change. Since neither of these impacts have any quantitative estimates, it is not possible to include them in the sensitivity analysis.

Table 6.9: Sensitivity Analysis of Desalination (1999 U.S.D./m^3)

Variable Analyzed	Cost of Variable Analyzed	MOC of Desalination
Direct Cost		
DC = $0.73-0.81 (Base Case)	$0.73-0.81	$0.83-1.13
DC = -30%	$0.51-0.57	$0.61-0.89
DC = +30%	$0.95-1.05	$1.05-1.37
External Cost: Energy and Human Health		
$EC_{human\ health}$ *= $0.10-0.30 (Base Case)*	$0.10-0.30	$0.83-1.13
$EC_{human\ health}$ *= +25%*	$0.13-0.38	$0.86-1.21
$EC_{human\ health}$ *= +50%*	$0.15-0.45	$0.88-1.28
$EC_{human\ health}$ *= +75%*	$0.18-0.53	$0.91-1.36
$EC_{human\ health}$ *= -25%*	$0.07-0.23	$0.80-1.06
External Cost: Land-use Externality		
EC_{land} *= $0.00-0.02 (Base Case)*	$0.00-0.02	$0.83-1.13
EC_{land} *= +25%*	$0.00-0.03	$0.83-1.14
EC_{land} *= +50%*	$0.00-0.03	$0.83-1.14

[27] An average existing power plant has two times the nitrous oxides and sulfur dioxide emissions of the average new power plant per kWh (Krupnick and Burtraw 1996). In addition, Israel uses oil and gas-oil to generate 25% of energy demand. These power plants' emissions are higher than coal-fired units.

The MOC of desalination is especially sensitive to changes in the DC and the human health impacts from energy production (Table 6.9). If the DC of desalination increases by 30%, then the MOC increases by up to 27%. A 30% increase in the direct costs is possible since the DC of desalination omits the costs of brine disposal and transmission line access to the plant. In addition, it is unknown if the cost of land is included in the DC or if that cost includes a premium for coastal land, if applicable. If the cost of land, or its premium, is not included, direct costs could rise even further. For morbidity and mortality costs, a 75% increase in the externality estimate creates a 10-20% increase in the MOC of desalination. A 75% increase in the morbidity and mortality costs is plausible since existing Israeli power plants are more polluting than new power plants, the sulfur dioxide emissions from Israeli power plants are higher than European plants, and the ExternE study omitted some impacts because quantitative estimates were not available. Land-use externalities have little effect on the MOC of desalination.

CHAPTER 7: POLICY IMPLICATIONS AND DISCUSSION

7.0 Introduction

As Israel moves into the twenty first century, the country is facing severe water shortages. To meet the growing gap between demand and supply, Israeli decision makers are exploiting three water sources: (1) groundwater (through depletion), (2) wastewater reclamation and reuse in agriculture, and (3) desalination. Of these projects, policy makers consider groundwater depletion a stopgap measure for meeting short-term water shortages. For this reason, depletion has been occurring in Israeli aquifers for many years. Treated wastewater is seen as a primary source of supply for the agricultural sector and effluent is expected to increasingly replace freshwater allocations in the coming decades. Desalination, which takes place in Israel on a small-scale, is perceived as the long-term solution to water shortages. In deciding to pursue these water projects, Israeli decision makers make their decisions based on the private costs of supply. However, national water planning should be based on social, not private costs, and therefore, these three projects may not be the most socially efficient choices for the State of Israel.

7.1 Summary of Results

This research is concerned with incorporating environmental impacts into the assessment of water supply options. Such an assessment can aid our understanding of how social costing changes the costs of water supply development. Chapter three introduces the marginal opportunity cost (MOC) concept as an appropriate framework. Table 7.1 provides a summary of the MOC of each project examined in this report with the direct (DC), external (EC), and user (UC) costs broken out separately to explore their relative influence on MOC (columns 2-5). In addition, Table 7.1 presents the percentage increase in the cost per cubic meter when social costs replace direct costs (column 6).

Table 7.1: Marginal Opportunity Cost of Groundwater Depletion, Wastewater Reclamation and Reuse in Agriculture, and Desalination (1999 U.S.D.)

Project Alternatives	Direct Cost/m^3	External Cost/m^3	User Cost/m^3	MOC/m^3	% Increase MOC over DC
Groundwater Depletion	$0.30	Negative	$0.19-0.69	$0.49-0.94	60-215%
Effluent Reuse: Secondary Treatment	$0.24	$0.80-1.14	None	$1.04-1.38	330-475%
Effluent Reuse: Tertiary Treatment	$0.29	$0.14-0.22	None	$0.43-0.51	50-75%
Desalination	$0.73-0.81	$0.10-0.32	None	$0.83-1.13	15-40%

Table 7.1 provides some important insights into the costs of water development in Israel. The first set of conclusions relates to the costs of the water supply projects in isolation. The second set relates to the relative attractiveness of the water supply options. The next section discusses the main conclusions of the MOC analysis for each water project. Section 7.2 details the substantive policy implications.

7.1.1 Projects in Isolation

The social costs of the three projects are up to four and a half times the direct costs, indicating that these projects are more costly from a social perspective than from a private perspective (Table 7.1). If the analysis quantified all the impacts, the social costs would be even higher[28]. Since new and unconventional supply projects are expected to follow the same pattern, if decision makers continue to ignore the external effects of supply solutions, they will underestimate the opportunity cost of water development and burden third parties with the side effects. In addition, freshwater prices charged to water consumers range from U.S. $0.21-0.87/m^3 (Table 2.2). Consequently, if the social costs of water supply are accounted for, the price of freshwater across all sectors should be

[28] The analysis cannot quantify the following impacts: ecosystem degradation from the drying up of springs, saline spring releases from depletion, the impacts of ion toxicity on crop productivity, brine disposal and discharge from desalination plants, the national impacts of climate change, and energy

higher. The other important results of the MOC analysis by project alternative are listed below by project alternative.

Groundwater Depletion

The high UC of groundwater depletion more than doubles the cost per cubic meter of groundwater extraction. Decision makers rarely consider user cost and, thus, they underestimate the true costs of groundwater supply. The results of the UC calculation indicate that ignoring depletion will come at an enormous expense, especially when other project alternatives exist that cost less. In the Coastal Aquifer of Israel, for example, the worst-case scenario shows severe reductions to the operational capacity of the reservoir within 3-7 years if overpumping continues.

Wastewater Reclamation and Reuse in Agriculture

Salinity is a major problem when farmers use effluent for irrigation. The presence of salt in the wastewater stream reduces crop yields and degrades soils. Furthermore, water leaching from farm fields increases the groundwater salinity, raising future municipal drinking water costs. Farmers will bear the largest burden of high salt concentrations in the wastewater stream because the loss of income from salinity impacts on crops and soil (Section 5.5) are higher than the financial return for some crops (Haruvy *et al*. 1999). The only means of eliminating salts from treated sewage is through reductions at source or desalination plants. In addition to salts, the use of tertiary treatment with nitrification-denitrification (N-D) and soil and aquifer treatment (SAT) reduces the external costs of effluent irrigation by eliminating crop switching caused by effluent restrictions as well as groundwater contamination from nitrate pollution. The internalization of the externalities increases the DC of tertiary treated wastewater. This point is discussed in more detail in Section 7.2.1.

externalities related to agricultural crops, forests, biodiversity, noise pollution, material damages, and human health. Clearly, there is a need for additional work in quantifying externalities.

Desalination

The percentage increase in direct costs to social costs for desalination is low because the analysis omits many of the externalities. Furthermore, the analysis may undervalue the DC of desalination since it omits the costs of brine disposal and gives a point estimate for energy prices. Energy prices may affect the future DC of desalination because the Israeli electricity sector is deregulating and Israel intends to switch some of its coal power plants to natural gas (Almog 2000). Because natural gas prices can fluctuate and energy accounts for 44% of the DC of desalination, the cost of desalinated water could increase substantially. If the DC increased by 30%, the MOC of desalination would increase by up to 27%. Desalination is already an expensive technology and potential cost increases make it a risky investment.

This summary describes the results of the MOC analysis for each project in isolation from the other alternatives. These results are important for decision makers because they highlight the risks and uncertainties in the MOC estimates. With this understanding, the next section looks at the substantive policy implications.

7.2 Policy Implications

The Israeli government is pursuing groundwater depletion, wastewater reclamation and reuse in agriculture, and desalination as the three major water sources to meet present and future domestic water demands. However, Israeli decision makers have typically made water development decisions based on the private costs of supply. This study calculates the social costs of each project, to compare the options from a social perspective and evaluate whether Israel decision makers have made the optimal choice among existing water supply alternatives. This evaluation requires a comparison of the projects against each other and in relation to other policy options available to Israel for meeting its water needs. Evaluating the projects relative to each other provides insight into the degree to which Israel may wish to pursue each project. Currently, decision makers use groundwater depletion as a stopgap measure to meet short-term water shortages. However, if groundwater depletion has a higher cost than wastewater reclamation and reuse, the government should reconsider how it meets short-term water scarcity.

Evaluating the three projects in relation to the other policy alternatives is also important because there may be other viable supply sources or demand-side management (DSM) programs. If so, Israeli decision makers may chose to exploit those projects. However, because this comparison is outside the scope of the report, it is discussed in a cursory manner in Section 7.2.2. In sum, a comparison of the social costs of project alternatives is essential for decision makers informed water policy choices in future.

7.2.1. Relative Attractiveness of the Three Projects

Table 7.2 illustrates the project rankings based on DC and MOC. This table bases its results on Table 7.1 and considers the lower and upper MOC estimate separately because of the large range of estimates for some projects. The following scale is used for projects ranking: "1" indicates the most attractive project or the project with the lowest cost, and "4" indicates the least attractive project or the project with the highest cost.

Table 6.28: Project Ranking: Groundwater Extraction and Depletion, Wastewater Reclamation and Reuse in Agriculture, and Desalination

Project	Direct Cost	Lower MOC Estimate	Upper MOC Estimate
Groundwater Extraction and Depletion	3	2	2
Effluent Reuse: Secondary Treatment	1	4	4
Effluent Reuse: Tertiary Treatment	2	1	1
Desalination	4	3	3

When social costs are compared, the relative attractiveness of the projects changes (Table 6.28). The lower and upper end of the MOC range indicate that effluent reuse using secondary treated wastewater is the least attractive project and effluent reuse using tertiary treated wastewater with SAT is the most attractive project. Table 7.2 indicates the project with the lowest DC is the least attractive project from a social perspective.

Moreover, the project rankings show that desalination, typically thought of as the most expensive water project, is ranked third out of four from a social perspective. Furthermore, groundwater depletion, in the MOC estimates, is not the cheapest source of supply, yet most decision makers characterize this water option as the cheapest water source for meeting shortages.

The ranking in Table 7.2 raise some important implications for Israeli water policy. First, although the three projects under consideration are not mutually exclusive, and could all take place simultaneously, the extent to which Israel exploits each option is an important question. The social costing analysis shows that as long as Israel restricts secondary treated wastewater in irrigation, and farmers must crop switch away from high value crops like vegetables to low value crops like cotton, it is more efficient to spend additional funds to treat effluent to a tertiary level with SAT[29]. However, it is estimated that by 2005, only 28% of all wastewater will be treated to a tertiary level with SAT and 70% will be treated to a secondary level or less (Hoffman and Harussi 1999). The results also show that even if the external costs of effluent restrictions are omitted (Table 5.11), the MOC estimate for secondary treated sewage is still higher than the MOC of tertiary treatment. Thus, assuming that land is available to accommodate the need for spreading basins in SAT, the Israeli government should invest more heavily in tertiary treatment facilities. Second, although the quantity of wastewater treated is limited by household discharges, it is more efficient to invest in tertiary treatment plants with SAT than to move ahead with large-scale desalination. If Israel treated all wastewater to a tertiary level with SAT, and long-term water shortages still existed, then it would be reasonable for the government to pursue large-scale desalination. However, the government plans to have four desalination plants running by 2005, while it treats only 28% of all wastewater to a tertiary level with SAT. This analysis suggests that the Israeli government should aggressively pursue effluent irrigation with tertiary treatment before it commits to additional desalination plants. In sum, although Israel is the world leader in the reuse of

[29] The loss of farm income from effluent restrictions accounts for approximately 50% of the MOC of secondary treated wastewater (Section 5.3.1.)

treated sewage, the country should exploit tertiary treatment further before it considers other project alternatives.

The ranking in Table 7.2 also shows that desalination is more expensive than groundwater depletion, even when many of desalination's environmental impacts have not be monetized. Therefore, it appears to be cheaper for the Israeli government to deplete its aquifers today than to pursue large-scale desalination. However, if depletion continues, Israeli aquifers may become unusable and future generations will no longer have access to those water sources, in addition to incurring other associated external costs. Israeli decision makers need to consider the trade-off between the increased cost of desalination versus the cost to future generations of losing its aquifers as a source of water supply.

7.2.2. Broader Policy Implications

Within the broader policy arena, decision makers must choose among various project and policy alternatives. In this instance, the Israeli government chose to deplete groundwater, reuse effluent, and build desalination plants as the primary means of meeting domestic water demand. Were these decisions socially efficient? The answer requires a comparison of the three projects with other project alternatives, like other water supply projects, DSM projects, and other policy alternatives, like reallocating water between sectors. However, it is impossible to make these comparisons within the scope of this report, as it requires calculating the full social costs of the projects discussed in Table 2.1, all possible DSM options, and other relevant policy alternatives. Only when a project or policy has a DC higher than the MOC of the three projects discussed in this report can it be rejected without further analysis. For all other projects and policies, until further research is conducted, it is impossible to formulate any conclusions.

7.3 Conclusions

The results illustrated in Table 7.1 and the policy implications discussed above indicate Israeli policy makers are not always be selecting the water supply projects with the lowest social costs. First, groundwater depletion is a costly water option and less attractive than other viable alternatives, like irrigation with tertiary treated effluent (Table 7.1). However, the Israeli government has chosen to pursue depletion as a stopgap measure to combat water shortages. Years of overpumping have led to the current groundwater crisis in Israel, where aquifer depletion has reached alarming proportions. Second, the decision to pursue large-scale desalination to meet future water demands is more expensive than some cheaper alternatives. For example, aggressive DSM may postpone desalination by numerous years. Such a postponement would allow for more research into less costly desalination and renewable energy technologies, thereby reducing the direct and external costs of desalination. In summary, the cost of meeting water demand in the next decade is likely to rise as expensive desalination plants come online and groundwater sources become less viable.

Why have policy makers chosen to deplete groundwater sources and build desalination plants when these options are more expensive than other alternatives? One possible explanation relates to politics in the Middle East. Since Israel is at the center of continuing conflicts with many Middle East countries, any bilateral or multilateral project that requires transboundary movement of water is not viable, since it requires mutual agreement between countries in conflict. In addition, any water project that requires Israel to rely on an outside source for water may be perceived as a security risk since water availability is not under Israeli control and could be disrupted by the supplying state. Consequently, the benefit of desalination may outweigh the benefit of reliance on third parties for a critical resource like water. Similarly, groundwater depletion may presently be the best strategic choice for Israel, even though the country will have no usable aquifer in the long run. Thus, Middle East politics makes sustainability more difficult to achieve since the need for security and control of water outweighs the environmental damages of domestic water development. This trade-off highlights the incongruence between long-term sustainability and short-term survival. However, the following question remains: when peace emerges in the Middle East, will there be any natural resources left to sustain the region? The answer depends partly on whether environmental damaging projects remain a political necessity or whether Israel is able to move towards more sustainable policies.

Figure 1: Groundwater Basins and Direction of Groundwater Movement in the State of Israel (MFA 2001)

Figure 2: General Scale to Measure the Effects of Sodium Adsorption Ratio on Soil Properties (Rhoades *et al*. 1992)

Appendix A: Economic Valuation Techniques

Valuation Technique	Advantages	Disadvantages
Market Prices Uses prevailing prices for goods and services traded in domestic or international markets. Includes changes in the value of output and loss of earnings.	• Market prices reflect willingness to pay for costs and benefits of goods and services that are traded. • Price information relatively easy to obtain.	• Market imperfections and/or policy failures may distort market prices, which consequently fail to reflect the economic value of goods or services to society. • Nonuse values are ignored and nonmaterial damages are excluded.
Changes in Productivity Physical changes in production are valued using market prices for inputs or outputs. Changes in productivity occur when a project or policy causes unintended damages to another productive system.	• Market prices reflect willingness to pay for costs and benefits of goods and services that are traded. • Price information relatively easy to obtain.	• Market imperfections and/or policy failures may distort market prices, which consequently fail to reflect the economic value of goods or services to society. • Nonuse values are ignored and nonmaterial damages are excluded.
Dose-response Function Estimates the value of a nonmarket resource or ecological function from changes in economic activity, by modelling the physical contribution of the resource or function to economic output.	• Estimates the entire demand curve.	• Requires explicit modelling of the 'dose-response' relationship between the resource being valued and some economic output. • Relationship between pollution and damages difficult to estimate because of: site- and time-dependent effects, non-linear relationships, lags and discontinuities, correlation vs. causation, and uncertain knowledge of damages.

Appendix A: Economic Valuation Techniques (Continued)

Valuation Technique	Advantages	Disadvantages
Control Cost Measures the value of an environmental asset by the costs of avoiding a negative impact.	• Market prices reflect willingness to pay for costs and benefits of goods and services that are traded. • Price information relatively easy to obtain.	• Nonuse values are ignored and nonmaterial damages are excluded. • May overestimate welfare measures if other benefits are experienced.
Travel Cost Method Derives willingness to pay for environmental benefits at specific locations by using information on the amount of money and time that people spend to visit the location.	• Market prices reflect willingness to pay for costs and benefits of goods and services that are traded. • Price information relatively easy to obtain.	• Data intensive. • Restrictive assumptions about consumer behaviour (e.g. trip multi-functionality). • Results highly sensitive to statistical methods used to specify the demand relationship. • Nonuse values ignored.
Contingent Valuation Method Establishes a monetary value for an environmental asset by asking people how much they are willing to pay for it.	• Includes use and nonuse values.	• Biases: informational, starting point, vehicle, hypothetical, operational, mental account, warm glow effect, and embedding effect. • Willingness to pay and willingness to accept measures diverge. • The geographic area of the analysis can bias results.

Source: (Gilpin 2000; Garrod and Willis 1999; Hanley et al. 1997; IIED 1994; Pearce and Turner 1990; Dixon et al. 1986; Hufschmidt et al. 1983)

Appendix B: Macronutrient Concentrations in Secondary Treated Wastewater

Macronutrient	Macronutrient Concentration (per m^3)	Complications	Ministry of Agriculture Guidelines
Nitrogen	40mg/l: Almost 100% of crop requirements	• Nitrogen needed during vegetative growth in early spring, irrigation water needed in summer. • Quantity of nutrients in effluent available to crops depends on form of nitrogen, which differs by water source.	• Secondary treated wastewater can account for up to 80% of nutrient needs of the crop.
Phosphorous	10-15mg/l: 300% of crop requirements	• Problems exist with phosphorous buildup in the soil.	
Potassium	20mg/l: 50% of crop requirements		• Additional potassium required.

Source: (Tarchitsky 2001)

Appendix C: Salt Accumulation

Impacts of Salt Accumulation

Salt accumulation, as measured by the electrical conductivity of the soil saturation extract (EC_e), reduces the osmotic potential of the soil, harming a plant's ability to absorb water. High EC_e values are detrimental since a plant expends more energy on adjusting salt concentrations within its tissue to obtain the water it needs from the soil and less energy is available for growth. Excessive salinity can lead to stunted plants. In addition, high salinity values, depending on the concentrations of chloride, sodium, and boron, cause one or more of the salt ions to accumulate in the soil and/or plant and long-term buildup of these elements may lead to specific ion toxicity. Specific ion toxicity results in leaf burn, chlorosis, twig dieback, and nutrient deficiencies. Finally, the salinity content in the effluent can affect the sodium adsorption ratio of the soil, causing a reduction in soil porosity, hydraulic permeability, infiltration, and aeration. Different crops have different salt tolerance thresholds and dry and hot climate conditions exacerbate the aforementioned effects (U.S. EPA 1992; Eitan 1999; Feigin *et al*. 1991).

Effects of Ion Toxicity

Chloride and Sodium Toxicity

Citrus crops are the main species susceptible to ion toxicity from chloride and sodium. These crops have a threshold tolerance of 250mg/l for chloride and 100mg/l for sodium concentrations (Weber *et al*. 1996). From a cross section of 50 municipalities and cities, mean concentrations of chloride and sodium in the effluent stream are 330mg/l and 220mg/l respectively from 1990-1995 (Yaron et al. 1999). Given these concentrations, a reduction in crop yields from chloride and/or sodium ion toxicity is likely and can occur without external injuries (Maas 1990). Avocado yields are already affected by chloride toxicity in many parts of Israel (Tarchitsky 2001). However, with the Ministry of the Environment promulgating new regulations, and working with industry on alternative means of dumping brines, drops in the chloride and sodium levels are expected in the next decade. In the Dan metropolitan area alone, which contributes 30% of all effluent reused in agriculture, sodium concentrations have decreased from 294mg/l to 194mg/l and chloride concentrations have decreased from 340mg/l to under 260mg/l from 1993 to 1999 (Israel MOE 1999c). Therefore, although salinity concentration will never be zero, strategies can lessen the severity of impact.

Boron Toxicity

Boron is a problem in Israel because there is a narrow concentration between levels essential to crop growth and levels that are toxic. Sensitive crops, including most citrus species, have a boron tolerance threshold of only 0.5-0.75g/m^3, while vegetables are more boron tolerant with maximum thresholds of 1-4g/m^3. The boron values measured in 50 municipalities and cities between 1990 and 1995 indicate that average boron concentrations in the effluent stream were 0.63mg/l. However, four locations had concentration above 1.0mg/l and 12 locations had concentrations above 0.75mg/l (Yaron et al. 1999). Moreover, a recent Ministry of the Environment survey reported that 65% of seasonal effluent reservoirs have a boron content of 0.6-1.6mg/l (Inbar 2001). Given the danger of high boron concentrations, and the difficulty in leaching boron from soils, the Ministry of the Environment has enacted legislation that will effectively ban the use of boron in all detergents by 2008 and the expected discharges are forecasted to drop by 95 percent from 1996 to 2008 (Israel MOE 1999b)[30].

[30] Detergents account for 80-90% of boron in the effluent stream (Inbar 2001).

Appendix D: Summary of Willingness to Pay Studies for Groundwater Protection (1999 U.S.D.)

Study	Mean WTP per Household	Sample Size	Description of Protection and Contaminant[1]	Payment Vehicle	Type of Question	Mean Income per Household	Significant Variables
G.L. Poe (1998) Portage County Wisconsin	$212	275	Protection of private well water to =< 10mg/l N when the probability of N >10 mg/l equals 50%	Increased taxes and water costs	Dichotomous choice	$30,000	Income, age, education, probability of exposure
A. Stenger and M. Willinger (1998) Alsace, France	$110-$128	817	Preservation of water quality with no specific source of pollution[2]	Water bill	Open ended and dichotomous choice	$25,300	Localization, frequency, knowledge of risk, prevention, bid, income, dialect
S.R. Crutchfield *et al*. (1997) White River Indiana, Central Nebraska, Lower Susquehanna, and Mid-Columbia Basin, WA	$607-$876 (average of four regions)	819	Reduction to <10mg/l N or the complete elimination of nitrates	Filter costs for water tap	Dichotomous choice	$25,000	Bid, personal income, extra income, years lived at zip code, age
J.R. Powell *et al*. (1994) 12 Communities in Massachusetts, Pennsylvania, and New York	$70	1006	Water supply protection from unspecified pollution sources	Water bill	Checklist	$35,000	Income, contamination incident, perception of water safety, type of water supply, amount spent on bottled water, number of perceived contamination sources

Appendix D: Summary of Willingness to Pay Studies for Groundwater Protection (Continued) (1999 U.S.D.)

Study	Mean WTP per Household	Sample Size	Description of Protection and Contaminant[1]	Payment Vehicle	Type of Question	Mean Income per Household	Significant Variables
J.L. Jordan and A.H. Elnagheeb (1993) Georgia, USA	$148[3]	192	Improvements in drinking water to meet standard of 10mg/l N	Water bill	Checklist	$22-28,000	Income, sex, education, color, uncertainty about water quality
H. Sun *et al*. (1992) Georgia, USA	$861	603	Eliminate potential for contamination from pesticides and nitrates	Reduction in income available for other good and services	Dichotomous choice	$42,500	Income, health concern level, contamination probability, offer price
S.D. Shultz and B.E. Lindsay (1990) Dover, New Hampshire	$164	346	Hypothetical groundwater protection plan	Property taxes	Dichotomous choice	$36,500 (net income)	Land value, income, age, bid
N. Hanley (1989) Anglia, England	$30	106	Guaranteed water supplies below 50mg/l nitrates	Water rates		$19,000	Income
S.F. Edwards (1988) Cape Cod, Massachusetts	$2,285	585	Guaranteed preservation of groundwater supply from nitrate contamination	Bond vehicle	Dichotomous choice	$55,400	Income, bequest, personal use effect

Remarks: (1) For all U.S. studies, the legal nitrogen (N) limit is 10mg/l (approximately 45mg/l of nitrogen in the form of nitrates). Therefore, nitrate contamination occurs when nitrogen concentrations exceed 10mg/l; (2) Even though no specific source of pollution was identified, one of the major recurring sources of pollution is nitrates originating from agriculture. The use of fertilizers in the agricultural sector accounts for 50% of the nitrate contamination; (3) Mean WTP of a household using public wells after outliers have been rejected.

Appendix E: Summary of the European Union (ExternE), Department of Energy, and New York Studies on Energy Externalities

Study #1: European Union Energy Fuel Cycles Study: ExternE 1995

The Directorate-General XII of the European Commission conducted the ExternE study to develop methods for estimating full fuel cycle costs in the European context. The project addressed the complete "cradle-to grave" costs for site- and technology-specific fuel cycles on a marginal basis; the study calculated the external costs for a new incremental investment (DGXII 1995a). For most fuel cycles, two reference environments were considered: West Burton, U.K. and Lauffen, Germany, and nine fuel cycles are studied including coal, lignite, oil, and natural gas (Krupnick and Burtraw 1996)[31]. Implementation was carried out across all European countries. Table E.1 lists the emissions for the U.K., Germany, Spain, and Greece, and Table E.2 lists the valuation estimates. Table E.1 also includes Israel's coal-fired power plant emissions for comparison.

[31] The study used U.K. and German sites for valuing the fuel cycle costs for coal since the two countries are the biggest users of coal in the European Union. The technologies used are typical of the choices made for coal-fired power stations commissioned in 1990. Both stations are fitted with flue-gas desulfurization, reducing SO_2 emissions by 90%. The German plant, because of regulation, has NO_x abatement devices. In addition, the U.K. plant is required to use low NO_x burners. As a result, the emissions of NO_x from the two plants are different. Although the plants' impacts are measured regionally, the U.K. implementation extends to the U.K., whilst the German implementation extends to all of Western Europe (DGXII 1995a).

Table E.1: Emissions of Coal-Based Power Plants by ExternE Location Compared with Israel (grams/kWh)

Plant/Category	Plant Size (Megawatts)	Sulfur Dioxide (SO_2)	Nitrous Oxide (NO_x)	Particulate Matter (PM)	Carbon Dioxide (CO_2)
Israel: Coal power plant	1100-1650	4.2	3.1	0.2	830
U.K.: West Burton, Midlands of England	1800	1.1	2.2	0.16	880
Germany: Lauffen, North of Stuttgart	700	0.8	0.8	0.2	880
Spain: Valdecaballeros, South-western Spain	1050	1.18	1.7	0.3	1015
Greece: St. Dimitrios, Ptolemais(1)	367	1.19	0.99	0.25	1320

Source: (IEC 1998; DGXII 1995a, 1995b)
Remarks: (1) The Greek case study quantified the lignite fuel cycle.

Table E.2: Monetized Human Health Impacts: ExternE (U.S.D.)

Location	Morbidity (mECU/kWh)(1) ($1995)	Mortality (mECU/kWh)(1) ($1995)	Total Human Health(2) ($1999/kWh)	Reference Population
U.K.	0.5	3.2	$0.005	Local 3.3m
Germany	2.4	9.9	$0.018	Regional 477m
Spain	3.9	21.4	$0.033	Not available
Greece	2.8	17.1	$0.027	Not available

Source: (Kollas 2000; Eyre 1997; DGXII 1995a, 1995b)
Remarks: (1) 1.25 U.S.D.=1 ECU, 100 mECU = 1 ECU; (2) ExternE studies used dose-response functions for PM and ozone; SO_2 and NO_x were modeled indirectly via their contribution to the formation of sulfate and nitrate aerosols (DGXII 1995a).

Study #2: The U.S. Department (DoE) of Energy Fuel Cycles Study:

Oak Ridge National Laboratories/Resources for the Future 1995

This project investigated and developed methods for estimating full fuel cycle costs appropriate to new generation investments using 1990 technology. The study estimated damages for two reference environments: Oak Ridge, TN and northern New Mexico. The study considered six generation-technologies, including coal, oil, and gas (Krupnick and Burtraw 1996). Table E.3 and E.4 list the emissions and valuation figures.

Study #3: The New York State Environmental Externalities Cost Study: Hagler Bailley with the Tellus Institute 1995

This project was a joint industry and governmental effort led by the Empire State Electric Energy Research Corporation and the New York State Energy Research and Development Authority. The study built a computer model capable of estimating damages to New York and surrounding states from new and re-powered generation plants anywhere in New York (Krupnick and Burtraw 1996). Table E.3 and E.4 list the emissions and valuation figures. Table E.3 also includes Israel's coal-fired power plant emissions for comparison.

Table E.3: Emissions per Study Area: Israel, DoE, and New York (grams/kWh)

Region or Study	SO_2	NO_x	PM	CO_2	Reference Population
Israel (Coal emissions only)	4.2	3.1	0.2	860.0	Not applicable
Department of Energy	1.58	2.6	0.14	n/a	Local: 0.87m Total: 193m
NY State	1.74	1.9	0.14	n/a	Local: 0.64m Total: 93m

Source: (IEC 1998; Krupnick and Burtraw 1996)

Table E.4: Monetized Human Health Impacts:
DoE, and New York (1999 U.S.D./kWh)

Study	Morbidity (mills/kWh)	Mortality (mills/kWh)	Total Human Health
DoE 1995[(1)]	0.44	0.28	$0.001
NY State 1995	1.54	1.16	$0.0033-$0.0042

Source: (Krupnick and Burtraw 1996)
Remarks: (1) This study did not include impacts from SO_2 since it assumed the tradable permit system accounted for any impacts.

Why are the Studies Different?

Human health costs of the three studies to diverge because of site-specific externality effects and the use of distinct estimation methodologies (Eyre 1997; Parfomak 1997). First, when impacts are not global in character, there is reason to expect that external costs are site-specific. Site-specific externalities are relevant to human health impacts, since higher population densities increase costs (Eyre 1997). In addition, site-specific meteorological conditions can also affect external damages. For example, it is reasonable to expect higher values for a state like California since the atmospheric pollution from power generation affects large population centers. By contrast, low externality costs should exist in a state like Maine, where most of the emissions blow out to sea (Parfomak 1997). Other site-specific effects can include emissions per unit of time, which depend on abatement measures like flue-gas desulfurization and the plant and capacity utilization factors (Krupnick and Burtraw 1996). For these reasons, damage estimates expressed in terms other than a per person basis are highly sensitive to the reference population affected by a new plant (Krupnick and Burtraw 1996). The second reason why results differ among studies relates to different estimation methodologies. Some issues include (Eyre 1997; Freeman 1996; Krupnick and Burtraw 1996):

1. The technology used.
2. How uncertainties in the causes and nature of the impacts are expressed.
3. Spatial boundaries chosen in air quality models.
4. Assumptions in air quality models such as the number of endpoints, space, time meteorology, air chemistry, thresholds, stack parameters, velocity and

temperature of stack gases and particles, and primary pollutants versus chemical reactions on these primary pollutants.

5. Valuation studies used for nonfatal health effects.

A study by Krupnick and Burtraw (1996) reconciled the assumptions of the Department of Energy, New York State, and ExternE studies. Table E.5 illustrates the results of the reconciliation.

Table E.5: Reconciliation of Three Externality Studies: ExternE, DoE and New York (1999 U.S.D./kWh)

Study	Original Estimates	Reconciled Estimates
Study #1: ExternE	\$0.021	\$0.0079
Study #2: Department of Energy	\$0.001	\$0.0021
Study #3: New York State	\$0.0033-0.0042	\$0.0043

Source: (Krupnick and Burtraw 1996)

The Krupnick and Burtraw (1996) analysis showed that the large variations in damage estimates in the DoE, ExternE, and New York studies could, in large part, be explained by varying assumptions and site characteristics of the studies and once adjustments were made, the estimated damages did converge. The authors explained the remaining deviations as differences in air quality monitoring. The detailed reconciliation of the studies shows that there is a movement toward consensus on the general approaches for estimating dose-response and damages for the air-health pathway.

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