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Susan E. Selke, Ph.D.
Major professor

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RATING THE ENVIRONMENTAL IMPACTS OF VARIOUS SOLID WASTE MANAGEMENT METHODS

By

Mun-Ling Salina Fung

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ABSTRACT

RATING THE ENVIRONMENTAL IMPACTS OF VARIOUS SOLID WASTE MANAGEMENT METHODS

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This thesis assigns scores to the overall environmental impacts of various solid waste management methods for seven kinds of packaging materials. Contributing factors including the environmental impacts of disposal and production, the conventional costs of disposal, the energy consumption in production, and the depletion of exhaustible natural resources are quantified into monetary costs. The total environmental costs exhibited by different materials in different disposal options are converted to scores in the scale of 0 to 1; the higher the cost, the lower the score. The results show that, for most of the evaluated materials, recycling scores higher than incineration and landfilling and is therefore more beneficial to the environment. In addition, it is better for the environment to incinerate than landfill combustible packaging materials. The average environmental cost of landfilling of packaging materials is found to be $1112/ton, of incineration is $1037/ton, and of recycling is $520/ton.

Dedicated to Edwin Lee, for his
endless support and patience.

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LIST OF ABBREVIATIONS

ABS	Acrylonitrile-butadiene-styrene
BHET	Bis-hydroxyethyl terephthalate
BOD	Biological oxygen demand
BOF	Basic oxygen Furnace
BTU	British thermal unit
CBA	Cost-benefit analysis
DMT	Dimethyl terephthalate
EAF	Electric arc furnace
EPA	Environmental Protection Agency
EVA	Ethylene vinyl acetate
FDA	Food and Drug Administration
FML	Flexible membrane liner
HDPE	High density polyethylene
LCA	Life cycle assessment
LDPE	Low density polyethylene
LLDPE	Linear low density polyethylene
LPG	Liquefied petroleum gas
MCDB	Milk cartons and drink boxes
MMBTU	10^6 BTU
MRF	Materials recovery facility
MSW	Municipal solid waste
MSWLFs	Municipal solid waste landfills
NO_x	Nitrogen oxides
NREL	National Renewable Energy Laboratory
NSCC	Neutral sulfite semi-chemical
PBT	Polybutylene terephthalate
PE	Polyethylene

PET	Polyethylene Terephthalate
PM10	Particulate matter < 10 microns in diameter
PP	Polypropylene
ppm	Parts per million
PS	Polystyrene
PVC	Polyvinyl chloride
RCRA	Resource Conservation and Recovery Act
RDF	Refuse-derived fuel
SO_x	Sulfur oxides
TPA	Terephthalic acid
TPD	Tons per day
TSP	Total suspended particulates
VOCs	Volatile organic compounds
WTE	Waste-to-energy
WTP	Willingness to pay

CHAPTER I

INTRODUCTION

HISTORY

Managing municipal solid wastes (MSW) used to be a relatively trouble-free task when the volume of garbage was moderate and the population density was sparse. In the 1930's there was a depression and people did not have much money to spend, and later between 1939 and 1945, people had money but the war kept few products on the market (Platt et al., 1991). Therefore, the amount of garbage was of a manageable size.

It was after World War II that the per capita rate of solid waste generation began to increase significantly. Furthermore, more land was developed for residential and business purposes in order to accommodate the population explosion during that time. Less land was thus available for landfills than there used to be.

Alarming issues on solid waste disposal began to surface in the 1960's when the increase in refuse volume was coupled with the fact that the improper solid waste treatment caused contamination of the environment and posed threats to public health. During the 1960's, a lot of landfills were not operating in sanitary conditions. They attracted flies and rodents, created odors and polluted the air, water and soil. And more, no incineration plants at that time met the

California air pollution standards (Compost Science, 1968). The Federal government finally took part for the first time to support a national effort to solve the solid waste problem: President Johnson signed the Waste Disposal Act in October 1965. The main purposes of the Act included 1). the initiation and acceleration of a national research and development program for proper/ economic solid waste disposal and conservation of natural resources by reducing wastes and recovery of potential resources; and 2). the provision of technical and financial assistance to state and local governments in planning solid waste management programs (Black and Gilbertson, 1966).

In 1970, the Environmental Protection Agency (EPA) was formed. It joined in with the local governments and the Department of Health, Education and Welfare to handle solid waste issues. EPA's initial focuses were on waste reduction and materials recovery. However, it soon adopted industry's suggestion of dealing with incineration as well. EPA and the Department of Energy set a goal in the late 1970's to build more than two hundred incineration plants by 1992. Their plan was strongly opposed by the public. As a result, more plants were cancelled than were ordered during the mid 1980's (Platt et al., 1991).

"Leaking landfills" was another solid waste headline that the public was concerned about in the 1980's. New standards were then set for the structure and performance of landfills to ensure that landfills provided barriers to the migration of contaminants to groundwater. When a lot of the landfills that did not meet the new standards were closed, the number of landfills dropped by half from 1983 to 1986 (Platt et al., 1991). Many of those remaining, however, still did not meet the standards (Selke, 1994).

Landfilling and incineration used to be the major channels for solid waste disposal in the 1960's. According to the EPA's statistics, 63% of MSW was landfilled, 30% was incinerated without energy recovery and the remaining 7% was recycled in 1960 (EPA, 1992). Nationally speaking, while landfills always manage to absorb the largest amount of MSW, incineration experienced its ups and downs during the last three decades. Since the Waste Disposal Act was in effect in 1965, a lot of old incinerators equipped with no pollution controls were closed between the 1960's and the 1970's. Incineration was unwelcome by the public and the amount of MSW incinerated dropped to 10% by 1980. The incineration technology however, was not stagnant. Incineration with energy recovery became the new direction for burning MSW. In addition, new regulations that monitor the operation of incineration plants were in effect and pollution devices were installed in the plants. All these had led to a substantial growth of the operation. 16% of MSW was incinerated in 1990, a 6% increase since 1980 (EPA, 1992).

Recycling, including composting, has been designated as another alternative for waste management. Unlike incineration, recycling has always been approved by the public. The effect of recycling on the solid waste stream, however, was insignificant in the 1960's and 1970's when less than 10% of the garbage was recovered. It was not until the late 1980's that recycling became much more active. The overall recycling rate reached 17% in 1990, and there were communities that achieved recycling rates of 25-40% (EPA, 1992; Platt et al., 1991). A race is on in the 90's for states to achieve higher and higher recycling rates. At the same time, composting is catching up to absorb some of the organic wastes.

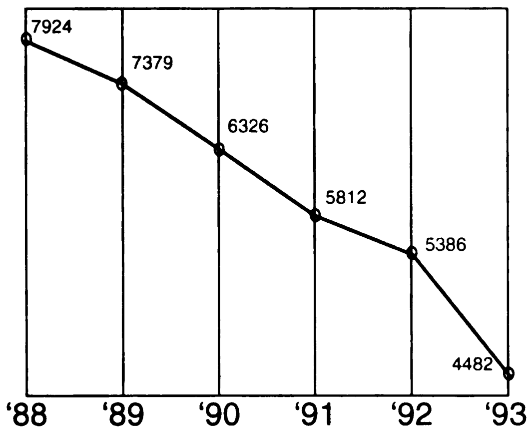
CURRENT DEVELOPMENT

In 1990, 195.7 million tons of MSW were generated, of which 67% was landfilled, 17% recycled and 16% incinerated. 32.9 wt.% of the MSW generated was containers and packaging. In landfills, packaging waste occupied 32.7 vol.%, which was the biggest share compared with other product categories (EPA, 1992).

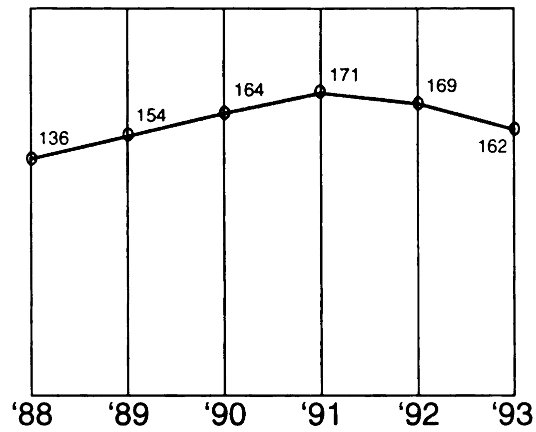
From an annual survey conducted by *BioCycle*, more current data are obtained for the development of MSW management in the nation. The survey, called 'The State of Garbage In America' (Steuteville, Apr 1994), reports figures collected primarily from state agencies and others from recyclers and trade associations. Figures 1.1(A) and 1.1(B) summarize some major statistics presented in that survey for 1988 to 1993. The survey focuses on MSW but some industrial waste is included because many disposal facilities handle both types of waste. Since EPA's statistics include strictly MSW, the disposal rates reported by the EPA differ from those by Biocycle by a few percent. Figures 1.1(A) and 1.1(B) are intended for demonstrating the trend in various solid waste management activities over the years, rather than pinpointing the numerical figures.

Figures 1.1(A) and 1.1(B) show that there has been a steady decline in the number of landfills as well as the landfilling rate. More and more MSW is diverted to recycling. Therefore, the number of curbside recycling programs and MRFs keep an upward climb. As a result, the recycling rate has more than doubled since 1989. Incineration, on the other hand, remains relatively stable in the last four years, keeping at a rate of around 10-11%. Table 1.1 illustrates the regional patterns of landfilling, recycling and incineration. The eastern states are the ones diverting the highest percentage of garbage to recycling and

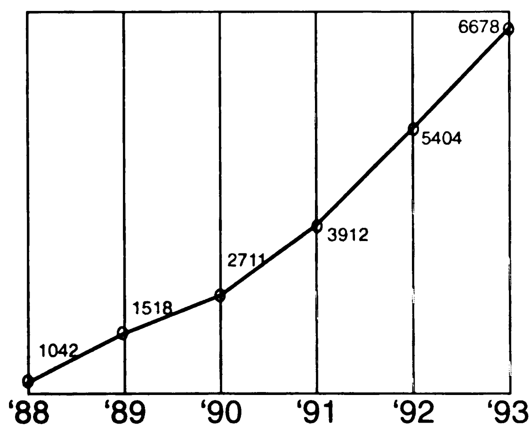
Figure 1.1(A) Statistics of Solid Waste Management Facilities in the U.S.



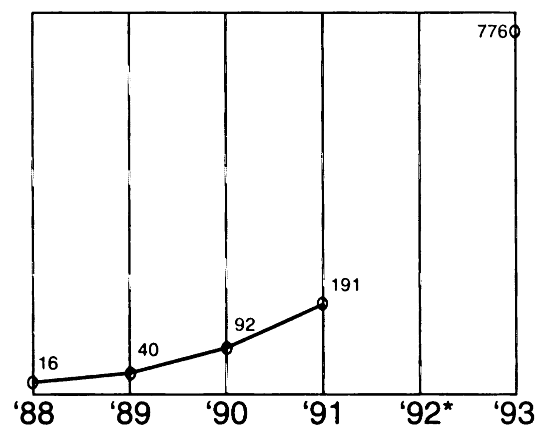
Number of Landfills



Number of Incinerators

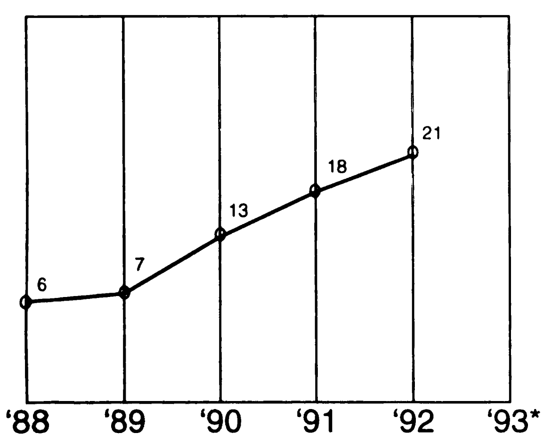


Number of Curbside Programs



Number of MRFs

* Information not available

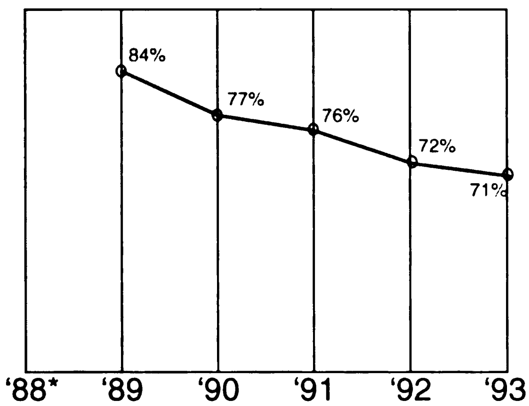


Number of Mixed Waste
Composting Facilities

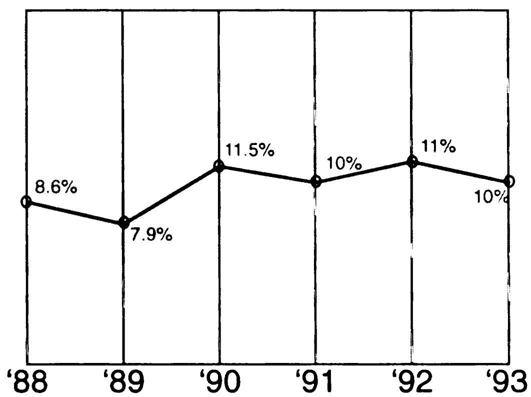
* Information not available

Source: Glen, 1989, 1990, 1991, & 1992, and Steuteville, May 1993 & Apr 1994

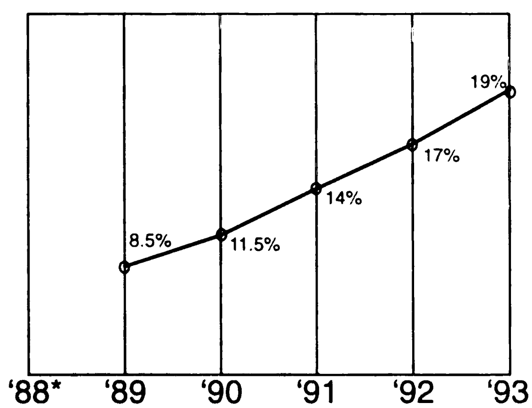
Figure 1.1(B) Disposal Rates of MSW in The U.S. (1989-1993)



Landfilling Rate



Incineration Rate

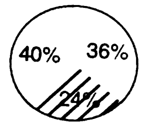
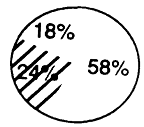
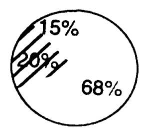
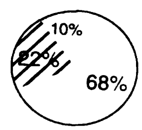
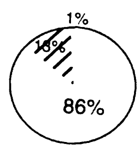
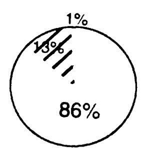
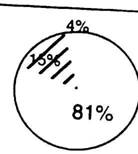


Recycling Rate
(includes composting)

* Information not available

Source: Glen, 1989, 1990, 1991, & 1992, and Steuteville, May 1993 & Apr 1994

Table 1.1 Disposal Rates in Different Regions of the U.S. in 1993

Region	State	Average Landfill Tip Fee	Landfilling, ▭ Recycling, ▨ and Incineration ▭ Rates
New England	Connecticut Maine Massachusetts New Hampshire Rhode Island Vermont	$46/ton	
Mid Atlantic	Delaware District of Columbia Maryland New Jersey New York Pennsylvania West Virginia	$53/ton	
South	Alabama Florida Georgia Kentucky Louisiana Mississippi North Carolina South Carolina Tennessee Virginia	$21/ton	
Great Lakes	Illinois Indiana Michigan Minnesota Ohio Wisconsin	$29/ton	
Midwest	Arkansas Iowa Kansas Missouri Nebraska North Dakota Oklahoma South Dakota Texas	$23/ton	
Rocky Mountain	Arizona Colorado Idaho Montana New Mexico Utah Wyoming	$15/ton	
West	Alaska California Hawaii Nevada Oregon Washington	$32/ton	

Source: Steuteville, Apr 1994

incineration. This solid waste management strategy is in response to the high tipping fees for landfilling in those regions (New England, \$46/ton, and MidAtlantic, \$53/ton). State-wise, Minnesota had the highest recycling rate (41%). 29% of Minnesota's waste was incinerated, and only the remaining 30% was landfilled. The District of Columbia reported the highest incineration rate (59%) while 30% of its MSW was recycled and a mere 11% was landfilled. The highest landfill rate was produced in Wyoming (96%). The remaining 4% was recycled (Steuteville, Apr 1994).

Except for Wyoming and a few other states, all states have an agenda on recycling. Forty-four states now have announced goals on recycling, 37 of which legislated their goals. Their percentages range from 20% (Maryland) to 70% (Rhode Island) (Steuteville, May1994). The average recycling goal from these 44 states is 39%. As a result of the surge of recycling mandates and recycling goals beginning in the late 1980's, an imbalance between supply and demand of recyclables was created. To remedy the situation, a lot of efforts were made in the early 1990's to create markets for the materials recycled. The recycling business finally began to see the light in 1994 when there was a substantial increase in revenues from recyclables. Corrugated cardboard leaped from a national average of \$25/ton at the beginning of 1994 to \$70/ton in October of the same year. Aluminum was \$0.20/lb higher in 1994 than the year before, and the price for HDPE doubled from 1992 to 1994 (Steuteville, Dec 1994).

Recently, a new type of package entered the recycling stream that is worth mentioning. MCDB (pronounced as McDub), which stands for milk cartons and drink boxes, is the type. It is not difficult to recall that drink boxes were identified as 'non-recyclables' in the late 1980's and Maine even banned their use in 1989. In a dramatic turn, these packages are now welcomed in the recycling stream because of the high quality fibers they provide. Recycled drink boxes command an attractive price of $90 to $150/ton. Seventeen states now include MCDB in their curbside programs. It is considered as one of the fastest growing additions to curbside programs (Steuteville, Mar 1994a & b).

Nineteen ninety-four was a critical year for landfill operation because Subtitle D of the Resource Conservation and Recovery Act went into effect to put more stringent requirements on liners and landfill management. The prediction was that landfill tipping fees would escalate as a result of the closures of a large number of landfills. According to Biocycle's survey, however, the national average landfill tip fee in 1994 was $28/ton, a $1 decrease from 1993. There were states that did report an increase in tipping fees, but there were also states which experienced a decrease. For example, the tipping fee in New Jersey dropped from $74/ton in 1993 to $61/ton in 1994, and Connecticut dropped from $65/ton to $30/ton. The decline was due to the opening of some massive landfills and the increased competition between private contractors (Steuteville, Apr 1994).

To help reduce the adverse environmental impacts of MSW landfills and to enhance recycling, disposal bans are enforced to keep hazardous and recyclable wastes from MSW landfills. Only four states do not have any bans on materials.The most popular banned wastes are batteries, tires, yard trimmings, motor oil and white goods. Five states have bans on certain packaging wastes (Steuteville, May 1994).

Compared to recycling and landfilling, incineration and composting do not make as much news. Incineration remains an unpopular choice for solid waste management because of the public's contention that it is a source of air pollution.

Solid waste composting is also striving for a place in the solid waste management system. The number of solid waste composting facilities has declined in the last few years: in 1990, 79 projects were in development and 9 facilities were in operation; in 1992, 61 projects were underway and 21 facilities were in operation; and in 1994, 17 facilities were in operation while only 34 projects were in development. Even though the composting technology and its management have improved over the years after lessons were learned from past failures, composting is still hurdled by cut-backs in state and municipal budgets, the divergence of enough waste to recycling and the competition from lower landfill tip fees. Experts say it is hard to predict how solid waste composting will evolve over the next few years (Goldstein and Steuteville, Nov. 1994).

OBJECTIVE

The objective of this project is to evaluate and quantify the environmental impacts of various solid waste management methods and then assign ratings to those methods according to the severity of their environmental impacts.

This objective is originated from the SWIPES project in which an equation was developed to calculate the relative volume of waste a package takes up in a landfill. The equation is shown in Figure 1.2. The equation requires the input of a list of data from a package and then generates a s value, which represents the relative amount of landfill volume occupied by the package. Among the variables, there are coefficients W, X, Y and Z, representing the environmental impacts for reuse, recycling, composting, and incineration respectively. These coefficients may take any values between 0 and 1. A score of 1 denotes the waste management option is of no or little cost to the environment and a value of zero

$$\underline{\mathbf{S}} = \frac{\left(\frac{m_{pkg}}{d_{pkg}}\right)\left[1 + f\left(\frac{m_s}{m_{pkg}}\right)\right](1 - X_1 r_r)(1 - X_2 r_c)\,[1 - (Yc)(1 - e)]\,[1 - (Zi)(1 - a)]\,(1 + b)}{1 + W(n - 1)} + P + S$$

$\underline{\mathbf{S}}$ = swipes
d_{pkg} = compacted density of package
d_{pr} = compacted density of product
m_s = mass or weight of scrap per package
m_{pkg} = mass or weight of package
m_{pr} = mass or weight of product
f = fraction of scrap disposed of

P = product correction factor = b (swipes value for product) or

$= b\left(\frac{m_{pr}}{d_{pr}}\right)[1 - (iZ)(\%\text{ combustible of product})]$

S = correction factor for manufacturing scrap

$= f\left(\frac{m_s}{m_{pkg}}\right)(X_1)(r_r)$

a = fraction of ash after combustion
b = average percent product damage in distribution
c = composting rate
e = residue after compost screening
i = incineration rate
n = number of uses
r_r = recycled content
r_c = recycling rate

W = arbitrary value for reuse ($0 \le W \le 1$)
X_1 & X_2 = arbitrary values for recycling ($0 \le X_1 \le 1$) ($0 \le X_2 \le 1$)
Y = arbitrary value for composting ($0 \le Y \le 1$)
Z = arbitrary value for combustion ($0 \le Z \le 1$)

FIGURE 1.2 The SWIPES Equation

Source: Saputo, 1991

denotes the option is of no benefit to the environment (Saputo, 1991). For example, if recycling is considered as imposing little harm to the environment, a value of 0.8 can be assigned to X. On the other hand, if incineration is considered as harmful to the environment, a score of 0.2 may be assigned.

Without a guideline to follow, the assignment of scores to these coefficients is rather arbitrary. The goal of this project is, therefore, to provide a methodology that would quantify the environmental impacts of those solid waste management options and then assign them scores that are scientifically based.

Those scores can be compared with the public perception and current trends of the various waste management methods to find out whether the current strategies on solid waste management are rationally based on environmental impacts or subjectively sided on public sentiment.

CHAPTER II
LITERATURE REVIEW

So far, no work is known to have been done on rating the environmental impacts of solid waste management methods in the United States. In Sweden, however, a study was carried out to rank the environmental impacts of reuse, recycling, incineration and landfilling of high density polyethylene. In the U.S., two other works are found on the evaluation and comparison of the cost, energy, and environmental impacts of various waste management methods. One paper was published by Yale University, entitled “Does the Solid Waste Management Hierarchy Make Sense?” (Schall, 1992). The other work, “Data Summary of Municipal Solid Waste Management Alternatives”, was published in a 12-volume report by the National Renewable Energy Laboratory (NREL) in 1992. All three studies used life-cycle assessment to identify the energy and material use, and the environmental discharge in various stages during the life-cycle of the waste.

The study performed by the NREL provides data on the energy analysis, air and water emissions, the amount of landfill space required and the capital and operating costs of eleven different combinations of waste management approaches. However, only landfilling, mass burn, and the RDF technology were studied independently as three separate alternative strategies. The remaining eight strategies combined recycling with one or two other waste management approaches in different proportions, with the amount of waste being recycled at most 20% in all cases. Therefore, little information can be drawn on the comparative impacts of individual waste management options. For this reason, this report will not be described in further detail. The other two studies are summarized as follows:

RECYCLING, INCINERATION, OR LANDFILLING OF POLYETHYLENE

"Recycling, Incineration or Landfilling of Polyethylene - A Comparative Environmental Assessment" was a paper presented at the ReC Conference in Geneva, Switzerland in 1993. This paper reports an environmental life cycle assessment (LCA) of different waste management approaches for high density polyethylene (HDPE). The LCA, carried out in Sweden, evaluated the energy requirements and environmental releases of HDPE from the extraction of the virgin raw materials to its final disposal. Impacts relating to additives and printing processes were not included. Processes that were carried out in all waste management options, such as packaging filling, distribution and use, were also excluded. The recycling rate of HDPE in Sweden was assumed to be 75%, and the remaining 25% was assumed to be landfilled. Residues from the recycling process were assumed to be landfilled as well. Pollutant data were based on actual measured emissions from processes wherever possible. Others were calculated using emission factors.

In this study, three evaluation methods were used; each allowed the energy requirements and emission data to be aggregated to one single figure for rating purposes. Those three methods were i) the ecoscarcity method (ECO), which relates the emissions to either an ecologically critical load or a politically determined emission target, ii) the weighted environmental theme method (ET), which is based on political targets and iii) the EPS method which is based on technical-environmental factors. All three methods originated in Europe. The paper refers to another source for the detailed description of those three evaluation methods. That source is a manuscript that was submitted for publication at the time this paper was released.

The results of the study are shown in Table 2.1. The evaluation of aluminum is listed as a reference to give the reader a feeling for the magnitudes of those values.

TABLE 2.1 Results of Environmental Impact Assessment of HDPE

Compared Cases	Differences by Evaluation Method		
	EPS	ET	ECO
HDPE:			
incineration - recycling	0.16	-2	-20
incineration - reuse	0.58	34	19
incineration - landfilling	N.A.	-133	N.A.
Aluminum:			
incineration - recycling	2.5	430	800

Note: '-' means minus
N.A.: not available

Source: Rydberg et. al., 1993

It was concluded from the results that product reuse is the most environmentally favorable. Incineration and recycling scored almost equally and landfilling of HDPE was worst for the environment. The study also found that the environmental impacts due to materials transportation during landfilling and recycling were the same, and such impacts contributed only a small fraction to the total environmental impact of the solid waste approach. It was noted that the results were specific to HDPE, and therefore should not be generalized to other materials. In addition, the results were generated from studying the disposal system in Sweden, therefore they should not be applied to other locations.

Even though this paper does not include detailed information on the method of ranking the waste management approaches according to their environmental impacts, it makes us aware that studies of this subject have been carried out in Europe, and evaluation methods have been developed to perform the task.

DOES THE SOLID WASTE MANAGEMENT HIERARCHY MAKE SENSE?

"Does the Solid Waste Management Hierarchy Make Sense?" written by John Schall, was published by the School of Forestry and Environmental Studies at Yale University. Mr. Schall was formerly the Director of the Solid Waste Group at Tellus Institute and was a visiting fellow at Yale during his work on this paper.

Like the question posed in the title, the theme of this paper is to justify the order of the waste management methods described in the hierarchy which prescribes source reduction as the most preferable, followed by recycling, then incineration, and landfilling being the last resort. Although the hierarchy exists as a popular political agenda, it has never been proven on scientific grounds. The author therefore utilized empirical data to provide the hierarchy with a technical, economic and environmental justification. In particular, this paper concentrates on demonstrating the priority of source reduction and recycling.

The data on which this study was based came mainly from three major research efforts that were conducted by the Tellus Institute: the Packaging Study; the California Disposal Fee Study; and the RPA Study, which analyzed the economic and the environmental impacts of several different 20-year solid waste management scenarios for the tri-state region in the Northeast.

Schall utilized the scenarios modelled in the RPA Study to evaluate the difference in the monetary costs of implementing different solid waste disposal systems. A baseline scenario modelled the existing solid waste management system throughout the tri-state area (New York, New Jersey, and Connecticut) in 1990. It described the tonnage distribution of solid waste in different collection programs and in different management options, and the respective percentages of waste being handled by each management option and various solid waste management facilities. No garbage was diverted by source reduction in this scenario. Scenario 1 modelled an integrated solid waste management system in the tri-state area through the year 2015 based on the state mandated goals in each of the three states. All the options including source reduction were carried out in this scenario. Scenario 2 eliminated source reduction and apportioned the previously prevented waste to the remaining disposal options at the rates modelled in scenario 1. Scenario 3 modelled for the region a system that utilized incineration and landfilling only. By comparing the costs of implementing scenario 1 and 2, the economic impact of source reduction becomes evident. Based on his model, Schall forecasts that a recycling rate of 47% would be achieved in the tri-state area in the year 2015. The cost difference between scenario 3 and scenario 2 thus displays the effect of implementing a recycling-intensive program.

The second type of costs that the author looked into was the environmental cost of disposal. The total environmental cost of each solid waste management method was calculated using the environmental impact cost of handling a ton of each waste material through each collection program and processing facility. These material-specific environmental costs were developed by the California Disposal Fee Study (CDFS) and the Tellus Packaging Study. A more detailed description of how these monetary costs for environmental impacts were derived can be found in Chapter III of this thesis. In addition to the packaging waste that Tellus analyzed, the CDFS also included twenty other non-food wastes in their analysis. Schall's paper included materials that were covered in the CDFS. The data used for the calculation of the environmental costs included the tonnage of each waste material that is handled by each disposal option from 1990 through 2015, the respective environmental costs for each type of collection program owing to collection truck emissions, and for each disposal facility, impacts due to emissions as well as environmental benefits from the prevention of production.

The third cost factor that was included in justifying the hierarchy was the environmental costs of the production of virgin and recycled materials. Air and water emissions generated from raw material extraction to materials manufacturing were quantified. Those data were developed in the Tellus Packaging Study and are illustrated in Table 2.2. The major findings of this study are illustrated in Tables 2.3, 2.4 and 2.5. In Table 2.3, the cost differences between scenarios 1 and 2 show that implementing source reduction in the waste management system can save $8/ton in the year 2000 and $15/ton in 2015. The cumulative reduction of waste from the year 2000 to 2015 was found to be 43 million tons, which amounts to a saving of $4.25 billion or $100/ton of waste prevented. In scenario 3, the disposal cost of incineration and landfilling only in the year 2015 is $131/ton, compared to a slightly higher cost of $134/ton if a 47% rate of recycling is integrated into the system as in scenario 2. The author pointed out that the data made no allowance for the rising disposal costs due to landfill depletion. The $3/ton difference was trivial considering the long term projection. In addition, a program such as scenario 2 would be able to recover 9 million tons of materials per year. The author therefore argued that a recycling-intensive solid waste management approach was no more expensive than incineration and landfilling alone.

TABLE 2.2 Environmental Impacts of Virgin and Recycled Production

Materials	Environmental Cost of Producing Virgin Materials ($/ton)	Environmental Cost of Producing Recycled Materials ($/ton)	Difference between Virgin & Recycled Material Use
Corrugated Cardboard	214	150	64
Boxboard	269	135	134
Glass	85	55	30
Ferrous	230	222	8
Aluminum	1933	313	1620

Source: Schall, 1992

Table 2.4 illustrates the environmental impact in each scenario. Again, a program that includes source reduction provides a significant benefit. A saving of $28.97/ton in scenario 1 is made possible by the prevention of emissions from collection trucks, from solid waste processing facilities and from material production facilities. For each ton of waste prevented, it was found that the

TABLE 2.3 Conventional Costs of Disposal

Scenario \ Year	1990 Baseline	2000	2015
Scenario 1 (Integrated)	$138/ton	$132/ton	$119/ton
Scenario 2 (no source reduction)	$138/ton	$140/ton	$134/ton
Scenario 3 (Incineration & Landfilling only)	————	$132/ton	$131/ton

Source: Schall, 1992

TABLE 2.4 Environmental Costs of Solid Waste Management Scenarios

Scenario \ Year	1990 Baseline	2015
Baseline (Integrated w/o s.r)	$2.83/ton	————
Scenario 1 (Integrated)	————	- $28.97/ton
Scenario 2 (no source reduction)	————	$4.03/ton
Scenario 3 (Incineration & l landfilling only)	————	$2.55/ton

Note: negative cost means benefit

Source: Schall, 1992

TABLE 2.5 Environmental Costs of Solid Waste Management Scenarios with Recycling Credits

Scenario \ Year	1990	2015
Baseline (Integrated w/o s.r)	-$5.46/ton	———
Scenario 1 (Integrated)	———	-$63.10
Scenario 2 (no source reduction)	———	-$36.59
Scenario 3 (Incineration & Landfilling only)	———	$2.55

Source: Schall, 1992

environmental benefit was \$221/ton, but to manage a ton of waste with scenario 2 costs \$4/ton. The author concluded that source reduction deserves its place at the top of the hierarchy. For the case of recycling, the environmental cost per ton for the disposal system with the recycling option (Scenario 2) is about \$1.50/ton more than for the incineration and landfilling option. The difference is due to the higher emissions per ton of waste generated by recycling trucks which do not compact waste as garbage trucks do. The author found that a recycling-intensive solid waste system was no more environmentally advantageous than a disposal-intensive system.

In Table 2.5, the environmental benefits of producing secondary materials as opposed to virgin materials production are added to the environmental costs. The result shows that an integrated solid waste management system (scenario 1) provides the most environmental benefit. A system that implements all methods but source reduction benefits less, and a system that implements landfilling and incineration alone incurs costs to the environment.

When the conventional cost of disposal in Table 2.3 for 2015 is combined with the total environmental costs in Table 2.5, scenario 1 costs the least overall (\$119 - \$63.1 = \$55.9), scenario 2 costs \$97.41 (\$134 - \$36.59) while scenario 3 costs \$133.55/ton (\$131 + \$2.55).

Based on the statistics generated in this study, the author concluded that source reduction is economically and environmentally justified to be placed at the top of the hierarchy. Schall's forecast of a 47% recycling rate in the year 2015 in his model suggests that it is technically feasible for the solid waste management system to manage an intensive amount of recyclables. It is also technically feasible for most of the packaging materials to incorporate at least 50% recycled content in their production. Taking into account these 2 attributes and the conventional and environmental cost data for recycling, Mr. Schall asserted that it is justified to rank recycling second to source reduction in the hierarchy.

Even though no scores were given to the waste management options and incineration and landfilling were not evaluated in detail, this paper provides a lot of quantified information to enable the comparison between source reduction and recycling. Since this thesis will use data from the Tellus Packaging Study for rating the solid waste management methods, Mr. Schall's work has provided some valuable insights on how to approach evaluating the hierarchy and what factors should be included in the rating criteria.

CHAPTER III
QUANTIFICATION METHODS

During the search for a method to rate the solid waste management options, a variety of methods were considered and three evaluation methods and a packaging analysis report were studied in detail. They include surveying, risk analysis, cost-benefit analysis and the Tellus Packaging Study. All three evaluation methods are capable of quantifying an event, which is the goal of this thesis; and the Tellus Packaging Study provides useful insights and valuable data for reference.

SURVEYING

A survey is a scientific study of people about their personal characteristics, background, and aspects of their behavior, knowledge, and opinions. It is a common technique used in the area of social science for studying the relationships between human characteristics and their behavior, in marketing research for learning consumer preferences, and in election activities for reflecting candidates' likelihood of winning. Carrying out a survey means eliciting information directly from people, either by telephone or face-to-face interviews, or by mailed questionnaires. The power of a survey comes from its ability to assign numerical values to non-numerical characteristics of human behavior in ways that permit uniform interpretation of these characteristics (Backstrom and Hursh-Cesar, 1981).

To perform a survey requires the following of a set of systematic procedures. Any deviation from these rigorous rules may produce survey results that are biased or invalid. The scientific nature of a survey enables replicability; in other words, other researchers using the same methods following the same procedures should obtain the same results. One basic characteristic of a survey is that it can generalize about a large number of people by studying only a sample of them. Therefore, good sampling is a very important factor in the success of a survey. First, the researchers should precisely define their survey objectives. Second, the target respondents should be identified. A sufficient number of respondents should be surveyed in order for them to represent the population under study. (A population in a survey could mean a collection of people of any size - a group, a state, or a nation.) Normally, there is no standard for sampling size, but it should be large enough to produce an acceptable margin of sampling error and a high level of confidence.

The next question is what sampling method to use in drawing the sample. Probability and non-probability sampling are the two types of sampling methods available (Fink and Kosecoff, 1985). In probability sampling, individuals are randomly selected from the population or from subgroups, or several groups of people are randomly selected from a number of groups of people. The particular kind of selection to be used depends on the research objectives and requirements. The key element is randomness. The effect of probability sampling is that people selected are the same as those who are not, so that the drawn sample is capable of representing the entire population considered. Non-probability sampling, on the other hand, is not concerned with the representativeness of the sample to the entire population. Instead, it is used when a specific group of people is to be studied. The selection of people may not involve randomness. The researcher may pick the first 100 people that pass by, or the researcher may use his/her own judgment in picking the respondents as long as the choices can be justified. Whether it is a probability or non-probability sample, it should be one that provides a satisfactory response rate. Choosing an adequate number of the right kind of people by a justified method is necessary but not sufficient in a sampling procedure. The researcher also has to make sure those who are selected are likely to respond to the survey. Otherwise, there will be insufficient data, and in turn, inconclusive results.

The next step in a survey process is to decide whether the survey should be conducted by interviewing by telephone, in person or by mail. Each of these three interviewing methods has its advantages and limitations. The underlying criterion for selecting an interviewing method is to choose one that produces the highest degree of reliability and validity for the specific purposes of the survey and yet meets the financial budget. For example, if the respondents to be interviewed are a group of people who would have difficulty in reading, it is obvious that a conversational interview is preferred to written questionnaires. Mailed surveys, however, allow respondents to answer at their own pace. In another case, a survey subject may be so complicated that it is difficult to answer on paper. The interviewer may require clarification and explanations from the respondents. A personal interview would be the best choice, but it is also the most expensive interviewing path to take. In that case, the quality of the responses has to compromise with the budget if cost is a limiting factor in the project.

One important part of a survey design is the construction of the questionnaires. The format and the wording of the questions have to follow specific guidelines, or else various kinds of biases may occur or the researcher would not obtain the necessary information. The English should be well written and the questions concrete. No ambiguity should be present in the questions. Sometimes, a certain term may mean different things to different people. The researcher has to define the terms being used in the questionnaires to make sure the respondents understand.

When writing survey questions, a lot of tact is needed in order to lead the respondents to surrender their true responses to the interviewer unawares. Respondents are being placed in an insecure position where a perfect stranger comes into their everyday lives and tries to dig into their personal details. It is not surprising that some people choose to give falsified or exaggerated responses which they think are more socially acceptable or which would enhance their self-image. There is no guaranteed way of eliminating such mischief. Following the rules of writing and interviewing for a survey will help minimize it.

Writing of questionnaires may involve writing of responses as well. There are formats of close-ended and open-ended questions. The former provides respondents with forced choices of responses and the latter asks the respondents to express the responses in their own words. Close-ended questions are more popular because the researcher only has to deal with a fixed number of responses. These responses are easy to analyze. Responses from open-ended questions may be more in depth but are usually extremely difficult to interpret (Fink and Kosecoff, 1985).

The sequence of the questions is also important. Questions of different difficulties and sensitivity ought to be put in a certain order. For example, sensitive questions should be placed toward the end of a survey followed by easy questions at the end. Objective questions should be asked before subjective ones. The idea is to make the respondents feel comfortable with the flow of the interview and hence be willing to provide their responses (Fink and Kosecoff, 1985).

After the survey is written, it should be pretested by presenting the survey to a small scale replica of the sample. This is an opportunity for researchers to rehearse for the survey, and to reveal possible weaknesses before performing the main study.

Telephone and face-to-face surveys require interviewers to be adequately trained. Interviewers who are unfamiliar with interviewing techniques are likely to induce biases in both themselves and the respondents, consequently jeopardizing the survey results.

Responses that are collected are then categorized and coded. Each category of responses is given a code to enable computer analysis (Backstrom and Hursh-Cesar, 1981). Here, statistical techniques are involved. Converting collected responses to statistical values is the ultimate goal of a survey. It is the quantitative values that the final user of the survey results depends on for comparison and decision making.

The overall framework of surveying was discussed above. To actually carry out a survey, there are a lot more detailed procedures to follow. It must be kept in mind that surveying works best when information has to come directly from people (Fink and Kosecoff, 1985). Researchers should take into consideration other methods if the information needed does not necessarily come directly from people. It all depends on the research subject. When there is a choice, surveying may not be the best approach to take. Performing a survey can be a straining task when the budget is limited. It can be very expensive to follow the strict rules of surveying and to interview a large enough sample. Due to the limited cost and time, the survey result may not penetrate deeply enough into people's minds. There is also no guarantee that the people who were interviewed are telling the truth. Surveying can be obtrusive to some people and it may stimulate the respondents to respond differently from what they really think. The ability of a survey to quantify human behavior and to generate contemporary results has led survey studies to proliferate in a lot of areas, from the government to the academic sector, and from commercial institutions to the news media. There are so many survey results that we are told of, some of us may start to question their accuracy and their usefulness. It is therefore important for survey researchers to perform meaningful and professional surveys to justify their purposes as well as their results (Backstrom and Hursh-Cesar, 1981).

RISK ANALYSIS

"Risk"- we may not think about it or realize it, but it exists and we are all exposed to it many times each and every day. In our own home, we might be electrocuted by turning on the light if the wiring is old and worn. We might be unfortunate enough to have taken some spoiled food and become poisoned. Going out for work or to school, there is always a chance of being involved in an automobile accident. Though it is true that those risks are too remote to be worried about, there are, however, risks that do catch our attention. A lot of the risks our society is concerned about nowadays are technology related: Is the air clean enough to keep us from being harmed by the various kinds of pollutants that are emitted from industrial activities? Would a nuclear power plant be a threat to our lives? What is the chance of getting killed by flying on a plane? People who are exposed to those health risks may suffer from injuries, illnesses, or they may even lose their lives.

Human beings are not the only species to bear the hazardous trade-offs of technologies. The natural environment faces the hazards as well. Environmental risks can take the form of deforestation, damage of soils, pollution of air, contamination of water and many other forms of degradation of the environmental quality. In light of the serious consequences that could be brought about by technologies, a branch of study is devoted to evaluate risks of that nature.

The study of risks involves three parts: risk assessment, risk abatement and risk management. Risk assessment is the process of determining the adverse consequences that may result from the use of a technology or some other actions (Conservation Foundation, 1985). It involves the quantification of risks. Risk abatement looks for techniques to be used to regulate or otherwise limit the levels of the risks assessed; and risk management determines the level of risks to be controlled and chosen (Glickman and Gough, 1990).

There is no such thing as zero risk. The elimination of one risk arisen from one event will automatically increase the risk that is associated with the substitute. Moreover, our society may not be able to afford the immensely high costs to lower all risks to their minimum levels. The elimination of all units of air pollutants can serve as an example. In the absence of the pollutants, our health is at zero risk of being harmed by the pollutants. However, the current affordable technologies can only abate the amount of pollutants down to a certain level. To abate the remaining pollutants in the air requires more sophisticated and expensive technologies. The cost, however, may not justify the benefits if abating more pollutants will not make us much safer. We therefore have to decide what is the acceptable level of risk we would want to achieve, the level of risk that would provide us with "enough" safety. Governmental agencies are the ones responsible for this task. They make use of risk analysis to set policies that protect public health and the environment. They also use risk analysis to justify proposed regulations. It is their responsibility to provide the society with an agreeable quality of life. In addition to the public sector, private sectors such as insurance companies, the chemical product industry and the energy utility industry also practice risk analysis in their decision-making processes (Wathern, 1988).

Since the goal of this thesis is to select a quantification method for valuing the various waste management approaches, risk assessment, which is a process of quantifying risks, is the part of risk analysis that is close to the interest of this thesis. Therefore, the remainder of this section will be focused entirely on risk assessment.

Risk Assessment

There are typically three principle elements included in risk assessment of an activity:

1). a determination of the types of hazard posed,

2). an estimate of the probability of a hazard occurring, and

3). an estimate of the number of people, wildlife, or other environmental elements likely to be exposed to the hazard and the number likely to suffer adverse consequences (The Conservation Foundation, 1985).

Data collected for the above items for each technology or activity assessed are the basis for risk quantification and risk comparison. Data are aggregated and analyzed. A single risk index will result, which characterizes the riskiness of the technology. If several technologies or activities are under analysis, the technology bearing a higher risk index is considered to be riskier.

Following are the procedures for a risk assessment:

1). Define the scope of the risk assessment by selecting the adverse consequences that are relevant to the current analysis. Since a large number of hazards, both direct and indirect, immediate and long-range, could result from an activity, it is impossible to assess all the adverse consequences. Thus it is necessary to select those that are major contributors to the riskiness of the activity, and those that match the objectives of the analysis.

Adverse consequences of an activity may include: the loss of human lives, reduction in life expectancy, loss of human health, material losses, environmental damages, and societal disturbances (Hovden and Singleton, 1987).

2). Determine the units of measurement for each consequence selected to be assessed. Let the quantifiable variable representing each consequence be denoted by x_i.

For example, for a given technology:

Let x_1 (for consequence 1) be the number of accidental deaths/ unit of the product produced/ year.

Let x_2 (for consequence 2) be the number of trees destroyed/ unit of the product produced/ year, and so on.

The prerequisite for establishing those variables is that corresponding data have to exist. Even so, one has to think carefully if the choices of those units are appropriate for the assessment. It is important to note that using different units could affect the relative riskiness of the technologies or activities. For example, instead of quantifying the consequences with respect to the 'per unit of product produced', 'per employee' could be chosen. The interpretation of the riskiness will then be changed to a different perspective. The choice depends heavily on the point of view of the analysts and the objectives of the assessment. In all cases, the units of measurement predetermine the value of the resulting risk index.

3). Obtain all the possible outcomes of each consequence according to the units determined in procedure 2. Determine also the respective probability of the occurrence of each outcome. If there are insufficient data, the "worst-case" outcomes may be used.

4). Score the values of x_i's.The actual values of x_i's are not used for the calculation of the risk index. Instead, a utility function is defined for each consequence that would convert all possible outcomes of the consequence to scores that lie within a predetermined scale. For instance, a scale of 0 to 100 could be used. One could define the scale as follows:

Let "0" be the least extreme possible consequence which represents:

no casualties for x_1, no trees destroyed for x_2, and so on.

Let "100" be the most extreme possible consequence, which represents:

10 deaths for x_1, 10,000 trees destroyed for x_2, and so on.

In this example, the utility function is a linear function, thus intermediate scores are assigned in linear proportion to the defined scores.

5). The risk index R can then be expressed as

$$\sum_{i=1}^{n} w_i y_i = R \qquad \text{Equation 3.1}$$

where n is the number of consequences,

y_i is the summation of the products of a given score and the probability of it being incurred for consequence x_i.
(y_i is the expected score of consequence x_i), and

w_i is the weighing factor which expresses the importance of consequence x_i relative to other consequences
(Glickman and Gough, 1990).

The risk index R of a given technology is equal to the summation of the products of the expected score of each consequence and its relative importance to other consequences.

To evaluate the probability of a consequence being incurred could be a straining task. To simplify the problem, some experts choose to treat the utility as a certainty. The uncertainty factor is taken into account by inserting a range of possible outcomes into y_i. Different sets of results are then compared and analyzed by the experts. They will then finalize the conclusion of the assessment.

Limitations and Difficulties

Risk assessment employs science but it is not a completely scientific process. Science cannot decide on the relative importance of one environmental hazard to others. Science does not provide a complete guideline for scientists to interpolate laboratory results of animals to humans and high dose to low dose situations. There are a lot of occasions where science does not suffice, especially those in which the cause-and-effect relationships of the events are not known clearly. Disagreements are unavoidable whenever human opinions emerge. That is why risk assessment renders room for arguments and controversies. Human judgments involve subjectivity. Scientific experts have their own set of beliefs and may hold strong opinions on them. They could subject themselves to biases and misjudgments in some cases.

Scantiness of data also forces the use of value judgments. It is one major difficulty that clouds risk assessment. A lot of toxicity data are simply unavailable. It is even harder to obtain exposure data. The exposure to a given hazard could come from multiple sources instead of the single technology that is under investigation. There are antagonistic hazards that reduce the overall effect of exposure, and synergistic hazards that increase the overall effect of exposure. Some people or some environmental species may be more sensitive to a hazard than others. When data are difficult to obtain due to those circumstances, assumptions are usually made by employing the “worst-case” scenarios. There have long been debates over the usefulness of assuming the worst situations when in reality the most probable scenarios are of much more concern. Unfortunately, the prediction of the probability of a hazard occurring and the degrees of its effects is yet another laborious task. Experts have to decide on values for all those unknowns and uncertainties. Some scientists are forced to ignore the uncertain parts when they cannot be quantified.

Risk assessment is an applied policy analysis that emphasizes human health hazards, especially those concerning the potential mortality due to cancer or technological catastrophes (Wathern, 1988). There is another form of policy analysis called environmental impact assessment which is very close to the nature of risk assessment, but it emphasizes more the evaluation of hazards incurred by natural ecosystems. It is, however, a less structured and less sophisticated method in terms of its ability to quantify uncertainties, and its capability to estimate the magnitude of the impacts. Although risk assessment allows the evaluation of environmental hazards, the technologies for assessing engineering and chemical hazards are much more well developed than those for assessing environmental hazards. In Risk Assessment and Risk Control (Conservation Foundation, 1985), it was stated:

> "Assessment of hazards to the natural environment are even more difficult to make than assessment of human health hazards because environment assessments are apt to involve a wide variety of hazards and to engage numerous scientific disciplines. Environmental hazards can range themselves from threats to a particular species of plant or animal to changes in the upper atmosphere that affect climate. Among many relevant academic disciplines, the science of ecology potentially could be the most useful and could provide an integrated framework for assessing natural hazards. But ecology is not sufficiently developed to serve these functions well. One report summaries the situation with the comment, 'environmental assessments may, at times, require much more data than a health effects assessment, yet provide an answer that is more tenuous or at least less quantitative'"

Even being burdened with its numerous difficulties and limitations, risk assessment remains as one of the major policy analysis methods carried out by governments and industries to evaluate the riskiness of new projects and to justify regulations. The quest for certainty in risk assessment will continue, but it may never reach perfection, as there are too many cause-and-effect relationships that even scientists do not have the knowledge of, and there are too many chain reactions to be tracked down in this enormous and open environment.

COST-BENEFIT ANALYSIS

From its name, one can easily tell that cost-benefit analysis (CBA) is economics related and it uses dollar value as the unit of measurement. But being more than that, CBA is actually not as materialistic as it sounds. To explain what CBA is, one should begin with some economics concepts:

Economics is the study of the allocation of scarce resources among competitive uses (Anderson and Settle, 1977). Were there unlimited supplies of goods that fulfill our needs, an economic structure would not be necessary. We could simply consume as much as we desire at whatever time we choose to do so. That is not the case; so we need the knowledge of economics to help allocate our limited resources the best that we can in the attempt to make the most people in the society happy. Welfare economics is that branch of economics that deals with how a society can allocate its scarce resources so as to maximize social welfare. CBA is a tool applied in welfare economics to evaluate whether a specific public project would increase the social welfare. Such an evaluation is based on the economics criterion: an activity enhances social welfare if that activity results in a net increase in the value of goods and services produced throughout the economy (Anderson and Settle, 1977). The value of goods and services produced by the economy is measured by people's actual willingness to pay (WTP) for those goods and services. The prices may not equal the prices that people are asked to pay on the market. Another version of this criterion is the Hicks-Kaldor criterion which states that an increase in general welfare occurs if those that are made better off from some change could, in principle, fully compensate those that are made worse off, and still achieve an improvement in welfare (Anderson and Settle, 1977). Applying these two criteria in CBA, a public project is analyzed as economically efficient or socially favorable if its social benefits outweigh its social costs. Social benefits refer to any social advantages which may involve marketed and non-marketed goods and services associated with the implementation of the project. Likewise, social costs refer to any social disadvantages involving marketed and non-marketed goods and services generated by the project.

In welfare economics and, in turn, CBA, goods and services include both the marketed type, which means the ones being traded on the market and therefore having dollar values attached to them, and the non-marketed type which does not have a market price. Clean air is a common example of a non-marketed good. There is no market established for clean air. Even though individuals may have a high WTP for clean air, no actual cash transaction is possible for this good.

When carrying out a CBA, analysts have to determine the WTP for both marketed and non-marketed goods and services that are effects of the project. WTP for marketed goods can be derived from their market prices. Different approaches are used to establish WTP for non-marketed goods. Most of the approaches are directed toward the valuation of environmental goods because it is a major type of non-marketed goods being evaluated in CBA. Those approaches are:

1). Contingent Valuation Method: it directly solicits from a sample of consumers their WTP for a change in the level of environmental service flows, in a carefully structured hypothetical market (Hanley and Spash, 1993).

2). Hedonic Pricing Method: it infers consumers' WTP for the environmental goods from their consumption behavior on housing which is related to that goods.

3). Travel Cost Method: this method uses consumers' spending on the travel costs, entry fees or on-site expenditures on the environmental goods as a proxy for WTP for that good.

4). Control Cost Approach: It infers the cost that society attributes to pollution from the regulations that it imposes on itself (Tellus, 1992). This is the method employed by the Tellus Packaging Study which will be used as the major data source for this thesis.

Examples of public projects that make use of CBA are health care and prevention programs, soil and water conservation projects, national forest planning, and projects that evaluate environmental policies.

CBA can be divided into seven main stages: definition, identification, quantification, monetization, discounting, decision-making and sensitivity analysis. Each of these stages is discussed below.

1). Defining the Project

The very first step of carrying out a CBA is to define what is to be appraised in the analysis. The scope of the analysis should also be determined at this stage. At the same time, the population which the CBA is covering should be determined. One key assumption of CBA is that individuals are the best judge of their own interests. Individuals conglomerate to form a society. Therefore, the aggregation of individuals' valuation represents society's preference. Costs and benefits in a CBA are then aggregated costs and benefits over individuals. However, most of the time, the effects of a project spreading over a long distance are capable of affecting a large population in different degrees. A boundary must be set on the population of gainers and losers to be aggregated in a CBA.

2). Identifying Impacts of the Project

Once the objective and the scope of the project are defined, CBA should proceed with identifying all the impacts, positive and negative, resulting from the implementation of the project. Using an example of whether it is socially beneficial to construct a waste-to-energy facility in a selected location, the listing of impacts should include all resource inputs into the construction of the facility, effects on employment levels, effects on environmental quality, local property prices, effects on avoidance of landfill disposal, and effects on energy production.

The list of impacts resulting from the project should exhaust all kinds of relevant effects and not be restricted to tangible ones only. Impacts that are known to be intangible should also be included because cost-benefit analysis is designed to describe those effects in the report even if they are not quantifiable or monetized. By this means, decision-makers are able to understand the pros and cons of the project in a fuller perspective. When analyzing projects which affect the environment, environmental impacts count as long as they either cause at least one person in the relevant population to become more or less happy, and/ or change the level or quality of output of some positively valued commodity (Hanley and Spash, 1993).

In environmental CBA, a type of impacts called externalities is an important factor to be addressed. Externalities refer to impacts that an individual or an organization produces as costs arising from its activities, but where that individual is not liable for the costs. Externalities may also refer to benefits that are conferred to others by an individual, but this individual is not fully compensated for the benefits. The former, known as negative externalities, can be exemplified by the pollution that is generated from an aluminum-making plant. The pollutants are released from the plant into the open air, which is then consumed by other individuals who bear the subsequent environmental cost of the pollutants. The latter form of externality is a positive one, an example of which is the beautiful Christmas decoration that you put up outside your house. It provides your neighbors with enjoyment and yet they pay nothing for it. Some externalities are difficult to monetize. Those that are monetized are called external costs. Since externalities have great influence on society's welfare, environmental CBA tries its best at least to describe them, if not to monetize them.

3). Quantifying the Identified Impacts

Following the identification of relevant impacts, the amount of those impacts and the time of their occurrence should be determined. In the waste-to-energy facility example, this quantification stage may include the amount of solid waste to be handled by the facility, the energy input and output, the number of laborers and trucks required, the amount of pollutants to be emitted, etc. When the data required are difficult to predict, the analysts may use probabilities to determine an "expected value".

4). Monetization

The goal of CBA is to quantify the social impacts of a project or alternative projects, and then attempt to reduce them into a commensurable unit so that the benefits versus the costs of the project, or the net benefits of alternative projects, can be compared. Using dollar value as the common unit does not originate from materialistic reasons, but because of its convenience for valuation. If there are other units that are appropriate for the purpose, CBA will adopt them as well. However, in the meantime, dollar value remains as the most plausible unit of measurement in the analysis.

At this stage, CBA analysts have to

i). adjust market prices, if necessary, for impacts that involve marketed goods;

ii). estimate and predict prices for future benefits and costs; and

iii). calculate prices for unpriced impacts.

In a perfectly competitive market, the supply curve of a product represents the opportunity costs of production of that product, and the corresponding demand curve represents people's willingness to pay. The equilibrium price, which is the intersection between the supply and demand curve, indicates both the marginal social cost and marginal social benefit of the production of that product. In this case, the analyst can simply use the market price without making any correction. In many cases, however, market prices do not reflect the marginal social benefits and costs. Those cases include imperfect market competition that is born by monopolists, government subsidies and taxation, and unemployment. CBA analysts have to adjust the market prices in order to find out the true social costs or benefits in those situations.

Not all costs and benefits of a project are immediate. Some will not emerge until years later. CBA has to predict prices for the future costs and benefits. For example, in a water conservation program, one of the benefits is the increase in the productivity of fishes twenty years in the future. The analysis then requires an estimation of the fish prices over this time span in order to fill in the blanks for this benefit.

The most controversial part of this monetization stage is to price the unpriced. Environmental impacts are the major elements in this subject area. In the previous section, four pricing methods are named. It is arguable whether those methods are able to provide accurate valuations. However, they at least provide an explicit standard for measurement when none used to exist.

5). Discounting

Discounting is the calculation of the present value of some future sum of money. The formula for converting future values (FV) to present values (PV) is as follows:

$$PV = \frac{FV_n}{(1 + r)^n} \qquad \text{Equation 3.2}$$

where r is the discount rate and n is the number of years from now the future value is associated with.

In CBA, discounting is very important because it is this process that converts all dollar values of future costs and benefits into present values so that all the values of present and future benefits can then be aggregated and compared.

6). Decision-Making Rules

Two of the most commonly used criteria for decision-making by CBA are described as follows:

i).

$$NPV = \sum_{i=1}^{n} \frac{B_i}{(1 + r)^i} - \sum_{i=1}^{n} \frac{C_i}{(1 + r)^i} \qquad \text{Equation 3.3}$$

where NPV = net present value of a project
B_i = expected net annual benefits
C_i = expected net annual costs
r = social rate of discount per annum
n = project life (in years)

In this criterion, a project is considered favorable if the discounted net benefits exceeds the discounted net costs, in other words, NP is positive. If several alternative projects are being evaluated, the one yielding the highest NP, with other things being equal, would be preferred.

ii)

$$BCR = \frac{\sum_{i=1}^{n} B_i}{\sum_{i=1}^{n} C_i} \qquad \text{Equation 3.4}$$

where BCR = benefit cost ratio

The decision rule for this criterion is that the project is overall beneficial to the society if BCR exceeds unity.

The findings of CBA are only an aid to the decision making process. Besides referring to the values of NPV or BCR, decision makers may take other considerations into account when making a decision on a public project.

7) Sensitivity Analysis

Similar to the problem of risk analysis, a lot of uncertainty factors are involved in CBA. Predictions are made on future impacts and their associated costs and benefits. What if different sets of possible values are input? Will the NPV change drastically? And to what extent? To better understand the relationship between the inputs and the outputs, sensitivity analysis is important. It requires the recalculation of NPV or BCR with changes in the values of parameters, such as the discount rate, physical quantities and qualities of inputs and outputs, prices of these inputs and outputs, and project life span (Hanley and Spash, 1993).

Critiques of CBA

Cost benefit analysis is widely practiced in appraising public activities and policies. The issues may range from concerns with human health to the quality of natural resources. Critics question the ethics of placing dollar values on intangible items such as the human life. Some comment that CBA is an impersonal analysis of matters which bear personal feelings. Others point out that putting a dollar value on a benefit reduces the value of that benefit (Glickman and Gough, 1990).

Pricing the unpriced is not the only criticism CBA receives. The list goes on with the validity of the prices obtained. CBA uses various valuation methods to elicit individual's WTP on goods, basing on the assumption that individuals are the best judges of their own interests. Critics doubt that this is always true. Even if it is true most of the time, there are cases in which individuals display different behavior in private than in public situations. For example, an individual may agree that recycling is beneficial to the society, but at the same time he/she may not save the soda bottles. It is therefore unreliable to infer public preferences from individuals' private behavior. If the project to be evaluated is environmentally related, the complexity of the ecosystem may not allow individuals to comprehend, nor be correctly informed of the situation. Their reaction and viewpoint on environmental impacts may then be laden with false judgment.

Other philosophical and argumentative issues exist, but the context of this thesis can only allow a brief discussion of the major criticisms.

As to the response of CBA analysts, they admit that CBA is far from flawless, but its positive roles should not be ignored. CBA is systematic. It provides a set of standards that makes the major costs and benefits of a project explicit. Its method directs analysts' and policy makers' attention to specific aspects of a project, and hence provides solid information for the policy makers to base their decisions on. Its use of dollar value as the measuring unit is solely for convenience. There are no other units that can carry out the task as easy as the monetary unit does.

It may be objectionable to put a dollar sign on human lives or on the amenity of the environment, but a standard is needed if we would like to work toward a system that can fairly and efficiently allocate our resources. Without some sort of quantification and commensurability, the evaluation method is reduced to an approach similar to surveying or political voting in which valuation is made by implicit and individual judgments. It is believed that CBA contributes some exclusive benefits to the decision-making of public activities and policies.

TELLUS PACKAGING STUDY

In 1989, the Tellus Institute, a non-profit public interest research organization, began extensive research on the life-cycle environmental impacts of different packaging materials. The goal was to develop a firm scientific basis for policy makers to use when formulating packaging policies. The Tellus Packaging Study was supported by the U.S. EPA and the Council of State Governments. After a three-year study, the results were published in 1992 in a series of five reports. The reports contain detailed description of the production and disposal of major packaging materials. Since the goal of the study was to investigate the scientific aspects of packaging and be able to compare their effects on the waste stream and on the environment, quantification methods were used.

In the Tellus study, three categories of costs were determined for packaging materials. These costs are the means for comparison between different packaging materials. They are the conventional and environmental costs of disposal and the environmental costs of production. Conventional costs of disposal include the land, labor and capital incurred in collecting, transporting and processing packaging waste and the revenues received from the output in a given waste management method. The environmental cost of disposal is the cost associated with the environmental releases during the entire process of a waste management method including emissions from collection trucks and from waste management facilities and leachate from landfills. Environmental cost of production covers the cost of environmental releases during packaging production, from raw material extraction to the manufacturing of the packaging material. These three categories of costs are external costs of the packaging material, meaning that they are costs that the packaging producers do not have to pay but are passed on to others in the society. Tellus realized the magnitude of these costs varies among materials and among waste management methods. They therefore designed a scheme that identified these costs for each packaging material in each waste management method.

Conventional Cost of Disposal

The eight types of packaging materials that were studied for their conventional costs of disposal include aluminum, glass, paperboard, corrugated cardboard, ferrous containers, HDPE and PET containers, and non-recyclable plastic containers. The disposal options considered were recycling, incineration, landfilling and solid waste transfer. Whenever possible, marginal costs were used as measures of the conventional costs of disposal. Marginal cost is the increased cost required for the handling of an additional amount of packaging waste in the solid waste system. Marginal cost is the unit of monetary measure commonly used in the evaluation of public projects. It is the incremental change, not the average cost, that the government responds to when making or updating policies. In some cases, however, where it is difficult to calculate the marginal costs, Tellus used average costs as the second best alternative.

Disposal practices may differ considerably in different locations. To collect data on disposal activities in every part of the country is an immense task and the result generated would be too generic to be useful. Tellus therefore selected New Jersey as the state they investigated the disposal of the packaging waste in. New Jersey is one of the northeastern states where there is an impending crisis in landfill space. The information generated from the study provides some insights to other states of their potential problems in waste disposal when their amount of garbage and the scarcity of landfills measure up to New Jersey's situation.

Tellus began this part of the study by dividing New Jersey into seven county scenarios; each practiced a different combination of disposal options. Only residential waste data were incorporated because, according to Tellus, commercial waste was difficult to analyze due to its heterogeneity in terms of size and composition.

Tellus developed a solid waste management computer tool called the WastePlan model. By inputting the required data obtained from each region in New Jersey, WastePlan generates individual scenarios. All the scenarios combined serve as the basis for the calculation of the marginal costs of handling the major packaging materials in New Jersey. All the data input to the WastePlan were collected from New Jersey state agencies, surveys of county solid waste and recycling coordinators, as well as individual periodicals and reports (Tellus, 1992). The most recent data they used was dated summer to fall of 1990.

There are three modules contained in the WastePlan; each requires the input of relevant data:

i). The generation module calculates the total waste stream size and composition based on demographics of the region, ii). The collection module calculates the total quantities of waste handled in each collection system, the number of trucks and containers needed to collect the materials, and the associated capital and operating costs, through the input of information such as the percentage of materials diverted to recycling and garbage collection, truck type and its costs, crew size, average miles to facilities, collection schedule and efficiency.

iii).The last module analyzes the different facilities that process and dispose the collected materials. Facilities include drop-off recycling, recycling processing facilities, incinerators and landfills. The module calculates the number of facilities needed, land area and building size for each facility, the type and amount of equipment required, materials flow through each facility, the annual facility costs and the amount of revenues generated, and the quantity of residue produced. It will also generate the cost required for each waste management system to handle a ton of each material.

After generating output data for the existing scenarios, the WastePlan model estimated the marginal costs of handling each material in each waste management option. The estimation was achieved by adding an incremental volume equivalent to 15% of the entire waste stream to each material in all the 7 scenarios, while holding other materials' quantities constant. The unrealistically large increment ensures there has to be a change in equipment needs for all the solid waste programs. WastePlan then recalculated the costs. The marginal cost per ton of each material equals the cost increase divided by the additional tons of material added to the system.

Table 3.1 lists a summary of the marginal costs per ton for specific waste management options. The marginal costs for recycling and garbage collection

TABLE 3.1 Marginal Costs Per Ton for Packaging Waste Collection and Processing

	Aluminum	Ferrous	Glass	Corrugated	Paper	PET	HDPE
Recycling Collection	180.74	88.95	19.41	102.58	0.00	357.24	366.97
Garbage Collection	96.14	41.57	12.57	39.20	34.12	168.30	164.69
Recycling Facility	-734.24	12.49	7.71	19.72	0.00	0.52	-0.59
Incineration	93.10	93.10	94.23	54.42	51.74	-2.54	-2.54
Landfill	183.51	86.29	21.52	80.21	72.85	170.82	171.13
Transfer Station	115.01	115.16	115.19	115.06	113.37	115.51	115.51

Source: Tellus, 1992

were re-evaluated by recalculating the number of trucks needed and the number of households collected/hour when the amount of packaging waste was increased by 15%.

The marginal cost of a recycling facility is the processing cost minus the revenue earned from the materials.

Costs of incineration include the processing cost, the cost for ash disposal, and the revenues from energy generation based on the Btu content of the materials.

Since landfill is filled up by volume, the landfill cost per ton of a material is determined by the following equation:

$$\text{Landfill Cost/ton of Material A} = \text{Average Cost/ton} \cdot \frac{\text{Average Waste Density}}{\text{Density of Material A}} \qquad \text{Equation 3.5}$$

Lastly, transfer station costs were assumed to be based on tonnage measures only, even though the transfer costs by trucks, which account for one-fourth of the total transfer costs, were based on volume. The final disposal costs of the transferred waste were, however, based on tonnage.

By combining the collection cost/ton of the materials with the marginal cost at each disposal facility for each material, the total per ton marginal costs of waste management alternatives from different materials result. Table 3.2 lists those costs, which are referred to as the conventional costs of disposal.

Limitations

Numerous assumptions were made before the conventional costs of disposal could be estimated. For example, assumptions were made on the container size, number of collection sites, average distance to facilities, truck

TABLE 3.2 Per Ton Marginal Costs of Waste Management Alternatives for Different Materials

	Aluminum	Ferrous	Glass	Corrugated	Paper	PET	HDPE
Recycling	-553.50	101.44	27.12	122.31	0.00	357.76	366.38
Incineration	189.24	134.67	106.79	93.62	85.86	165.76	162.15
Landfill	279.66	127.86	34.09	119.41	106.97	339.13	335.82
Transfer Station	211.15	156.73	127.76	154.25	147.49	284.03	280.20

Source: Tellus, 1992

requirements, and costs and the number of households collected per hour in curbside collection. Other estimates and assumptions were detailed in the report. These assumptions were made because it was very difficult to keep track of their actual statistics. Users of the results should be aware of these assumptions.

Environmental Costs of Production and Disposal

The methodology used for valuing environmental damages derived from packaging material production and disposal is entirely different from that used for finding the conventional cost of packaging disposal. While conventional costs can easily be obtained from market prices of the goods and services considered, there are no explicit market prices for environmental impacts. Environmental costs have to be determined through indirect sources. Tellus Institute chose the control cost approach for the monetization.

As described before in the Cost-Benefit Analysis section, control cost approach infers the social costs of pollutants from the pollution abatement regulations that the society establishes. The level of pollutants our society decides to abate provides an indication of the costs we ascribe to the presence of the pollutants. The emission standard for a particular pollutant is usually set at the level at which the marginal cost of abatement equals the marginal benefit of abatement. This economic strategy allows an efficient allocation of the society's resources. This marginal cost of abatement associated with the regulation is the cost that society places on the pollutant. It is also the cost that the control cost approach ascribes to that pollutant as a valuation of the degree of environmental damage this pollutant causes.

The procedure for valuing the environmental cost of production and disposal of packaging started with the identification of pollutants emitted during those processes, proceeded with the finding of regulations that addressed the abatement of those pollutants in the context of solid waste management and industrials practices, and finished with estimating the costs of meeting those regulatory standards.

Three categories of pollutants were monetized for the environmental costs of production and disposal. They are: EPA's criteria air pollutants, which were defined in EPA's Clean Air Act regulations, greenhouse gases, and hazardous substances. The prices for EPA's criteria air pollutants were based on the costs of meeting EPA's standards estimated by the Southern California South Coast Air Quality Management District. There were no regulations established for greenhouse gases, but the prices of greenhouse gases were determined with reference to the reforestation cost for carbon dioxide developed by the California Energy Commission. Prices for other greenhouse gases were determined by multiplying the price of carbon dioxide by the global warming equivalence of a given gas with respect to carbon dioxide.

Since a lot of hazardous substances are not regulated, Tellus combined the control cost approach with a health effect ranking in order to develop prices for hazardous substances. This category of pollutants was first subdivided into carcinogens and non-carcinogens. The cost evaluation considered the damage of these pollutants to human health only. The reason that no environmental damage was investigated might be due to the complexity of tracing and identifying the effects of hazardous substances on the ecosystem.

In the subcategory of carcinogens, pollutants were ranked according to their cancer potency factors with respect to isophorone. A separate rank list was developed for non-carcinogens based on the level of maximum daily exposure without harm. Xylene was the pollutant used as a baseline reference. Tellus then employed the OSHA standard to relate the health impacts of isophorone to xylene so that the two lists could be combined in proportion. The combined ranking scores for pollutants in this category can be found in Table 1.3 of Report 4 in the Tellus Packaging Study. Lead is one of the pollutants on the list. The average control cost for lead found by Tellus was $1600/pound of lead controlled. Tellus used this pollutant price as the base for pricing the rest of the pollutants on their list. Those prices are listed in Table 1.4 of Report 4 in Tellus' report.

Environmental Production Cost

In Report II, "Inventory of Materials and Energy Use and Air and Water Emissions from the Production of Packaging Materials", Tellus identified the controlled and uncontrolled emissions generated by the production of each packaging material. Since no pollutant prices were established for uncontrolled emissions, only controlled emissions were used for calculating environmental costs. The environmental costs of producing a ton of a particular packaging material were calculated by multiplying the emission factor (pound of pollutants/ ton of packaging material) of each pollutant a packaging material generates during production by the pollutant price ($/pound of pollutants) of that pollutant. The total environmental cost of producing a packaging material is the sum of all the environmental costs of the pollutants emitted during its production, from raw material extraction to the manufacture of the packaging material. The impacts from the production of additives are also monetized. Forming, filling and transportation of packages are omitted since the focus is on materials rather than packages.

Environmental costs of production for HDPE, LDPE, PP, PET, PS, PVC, bleached kraft paperboard, unbleached coated folding boxboard, linerboard, corrugated medium, unbleached kraft paper, and folding boxboard from waste paper were calculated.

Table 3.3 lists the environmental costs of production of each material.

Environmental Cost of Disposal

Figure 3.1 illustrates the sources of pollutants that were emitted during MSW collection and processing and indicates the types of pollutants that were evaluated for the New Jersey scenarios in the Tellus Packaging Study.

The primary source of environmental impact due to garbage and recycling collection is the air emissions from collection trucks. Tellus referred to the U.S. EPA report, "Compilation of Air Pollutant Emission Factors II: Mobile Sources" and the studies published by the California Air Resources Board for truck emissions data. Emission factors in the U.S. EPA Compilation report were converted from pollutants emitted per ton-mile to pollutant per cubic yard-mile because collection trucks are filled up by volume instead of by weight. Assumptions were made in order to estimate the amount of air emissions apportioned to each ton of material collected by the trucks. A few of the assumptions include the following:

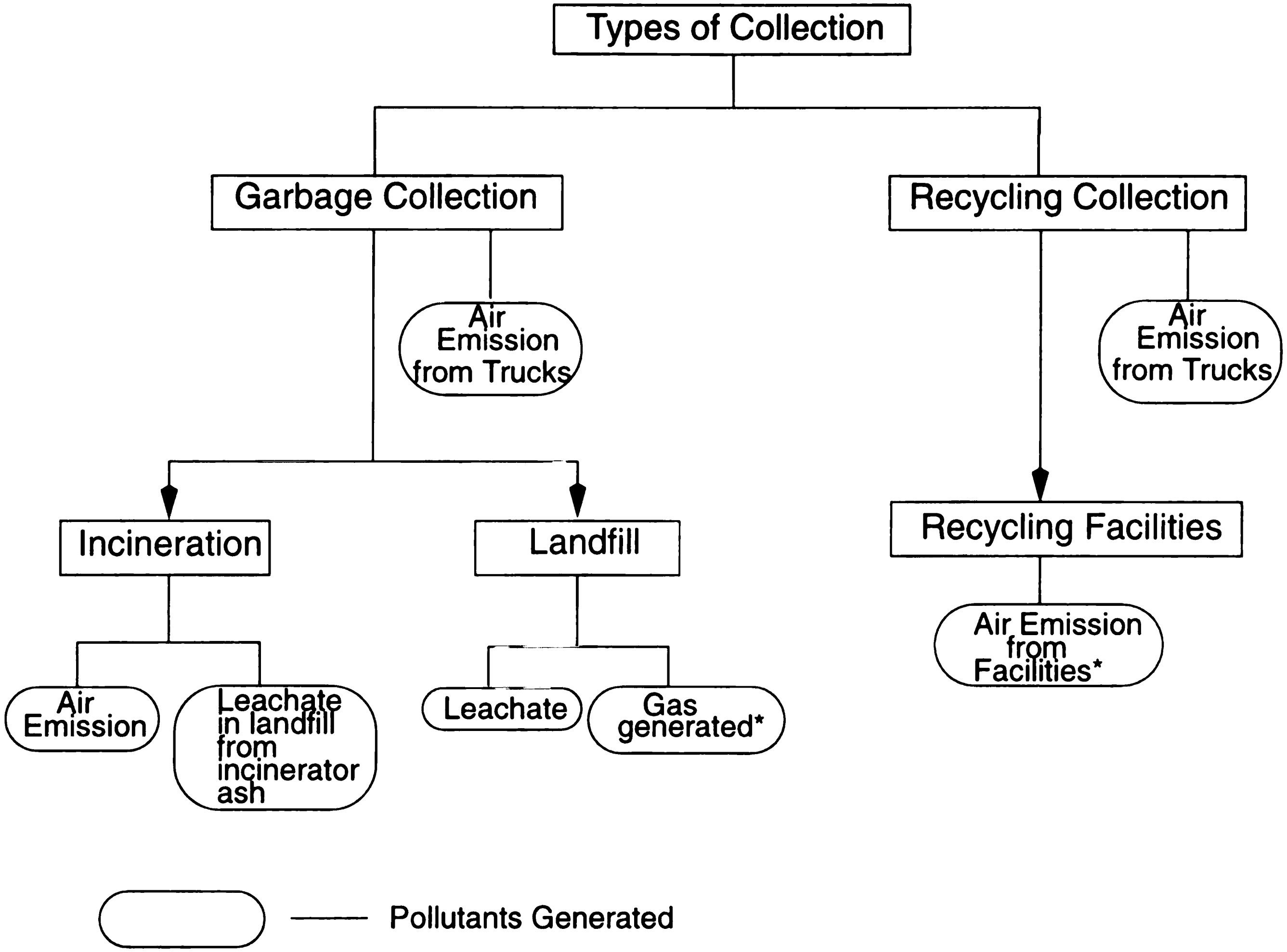
3 pounds of waste generated/person/day
2.6 persons/per household
15 wt.% of materials are recycled
recycling collection rate equals 80 households/hour
garbage collection rate equals 60 households/hour

Tellus made some adjustments on the adopted emission factors since the

TABLE 3.3 Environmental Costs of Material Production

Material	Criteria Air Pollutants ($/ton material)	Toxic & Carcinogenic Pollutants ($/ton material)	Total Environmental Cost of Production ($/ton material)
Virgin Aluminum	1511	423	1933
Recycled Aluminum	312	1	313
Virgin Steel	74	156	230
Recycled Steel	74	147	222
Virgin Glass	83	3	85
Recycled Glass	54	0	55
Unbleached Coated Folding Boxboard	187	82	269
Folding Boxboard from Wastepaper	120	14	135
Linerboard	193	80	273
Linerboard from Wastepaper	121	15	135
Corrugating Medium	77	6	83
Corrugating Medium from Wastepaper	162	21	183
HDPE	170	122	292
PET	261	593	854

Source: Tellus,1992



* Pollutants that were not included in Tellus evaluation on environmental costs of disposal

FIGURE 3.1 Sources and Types of Pollutants Emitted during MSW Collection and Processing

U.S. EPA's Compilation report addressed trucks traveling at high average speed while garbage collection involves a lot of idle time during collection. Emission factors of the latter case were assumed to be higher.

The emissions emanating from recycling facilities were studied. However, the sampling methods and results were not satisfactory. Tellus therefore decided not to include this information in the disposal costs. However, their preliminary findings involved emissions of various pollutants on the order of 10^{-10}-10^{-7} lb of pollutant/ton of material, which is small compared with the emissions generated from packaging production.

Two types of environmental impacts in landfills that were studied were leachate and gas generation. It was found that packaging material had little contribution to landfill gas emission. Therefore, only leachate generation was quantified.

To identify the amount of pollutants released by each packaging material into landfill leachate, first, the annual amount of leachate generated (gallon/ton waste/year) in a generic controlled landfill was determined through the use of U.S. EPA's "HELP" model for water balance in addition to some assumptions in regard to the landfill size and capacity. Second, the concentration of various pollutants in leachate (ppm or lb/gallon) was found by using some national data (Tables 2.2 and 2.3 Report 4 of Tellus Report). Third, the pollutant concentration was converted to pollutant factors (lb pollutants/ton of MSW). Finally, the amount of various pollutants was allocated to each packaging material by composition analysis of the materials. The results for both inorganic and organic pollutants in leachate range in the order of 10^{-10} to 10^{-7} pounds/ton for paper, plastics, glass and metal packaging.

The last type of pollutants investigated was emissions from MSW incinerators. Incinerators equipped with a scrubber, fabric filter baghouse and Thermal $DeNO_x$ for air pollution were the kinds evaluated. Pollutants due to the incineration of MSW could be emanated from air emissions during burning as well as from leachate of solid waste incinerator ash. The emission factors for solid waste incinerators ranged from the order of 10^{-9} to 1 pound/ton MSW depending on the pollutant in question. The emission factors for leachate originated from incinerator ash was on the order of 10^{-10} to 10^{-7} pounds/ton of MSW, which was much lower than air emissions. Again, the allocation of these pollutants to various packaging materials was determined by composition analysis.

An itemized presentation of the environmental impacts of disposal is shown in Table 3.4. The zero costs shown in the column of leachate in controlled landfill were because of the infinitesimal costs (in the order of $\$10^{-5}$) that resulted when pollutant prices were multiplied with the emission factors for leachate constituents. Likewise, leachate caused by incinerator ash was found to be minimal; therefore no environmental cost was associated with those pollutants.

EVALUATION OF METHODS CONSIDERED

To decide on what method to use in rating the environmental impacts of the various waste management options is not an easy task. Assigning scores to environmental impacts amounts to quantifying the intangibles. Valuation methods such as risk assessment and cost-benefit analysis have been used to generate numerical values for environmental effects, yet they are far from perfect to deter criticisms. Controversies often center around the validity of these approaches and the ethics of putting numbers on some abstract and sensitive items. Such

TABLE 3.4 Itemized Environmental Costs of Disposal ($/ton)

	Recycling Collection	Controlled Landfill (LF)			Incineration (IC)		
		Collection	Leachate	Total LF	Collection	Emission	Total IC
Aluminum	11.53	3.04	0.00	3.04	3.04	0.90	3.94
Ferrous	3.46	1.20	0.00	1.20	1.20	0.90	2.10
Glass	1.15	0.28	0.00	0.28	0.28	1.04	1.32
Paper	6.92	1.10	0.00	1.10	1.10	1.63	2.73
Corrugated	3.07	1.21	0.00	1.21	1.21	0.63	2.84
HDPE	19.77	2.56	0.00	2.56	2.56	1.44	4.00
PET	23.06	2.61	0.00	2.61	2.61	1.44	4.05

Source: Tellus, 1992

controversies may not be avoided in this project, but the selection among various valuation methods is carefully rationalized. The ultimate method adopted is believed to be the most feasible and appropriate among all methods considered for this project.

First, the choice of using surveying was given up. Surveys are best performed by experienced survey researchers. Amateurs trying to design a survey are likely to experience great difficulties and induce errors along the process. The survey outcomes, as a result, may be superficial, if not biased.

If a survey was performed for this project, the sample respondents should include selected experts from various fields such as environmentalists, packaging specialists, solid waste personnel and ecologists. Based on their expertise and judgments, different specialists may hold different views on the same issue. Interviewing experts from different fields can therefore explore all the possible variation of responses so that the survey topic can be studied at all angles to yield a fair result. The difficulties begin with the question of exactly how many different kinds of experts should be surveyed? For this current project, other than those experts mentioned, should biologists, toxicologists, geologists and environmental engineers also be interviewed? Should citizens be included in the survey as well? And how many experts in each category should be surveyed? Will the response rate of each category be the same? If not, their responses will be biased toward the groups with a high response rate. Even if the response rates are comparable among groups, one group may have a particularly strong opinion on an issue, and therefore provide more extreme responses to some of the questions. If those are questions that require the interviewee to rate on an issue, those extreme ratings will overshadow the responses provided by others who are not as fervent on the issue.

Surveying works best if information must come directly from people. In the case of this project, soliciting experts' opinions on the environmental impacts of disposal methods seems to be an indirect approach compared to other possible alternatives. If surveying was done for this project, the factual information on environmental impacts would be first absorbed by and then filtered through the minds of the experts. It is the filtered information that they provide in the survey. Those survey responses are still to be analyzed and interpreted by survey researchers. More direct methods are those that allow direct analysis of the first-hand information without going through the numerous media. With these shortcomings in mind, surveying was decided not to be the best approach for this project.

Risk assessment is a more direct approach than surveying: it assesses adverse consequences of an activity by looking directly into those consequences. The usefulness of risk assessment for this project is its ability to quantify events and reduce the corresponding data into one single index. If sufficient data is obtained on the adverse environmental consequences of each waste management method, risk assessment can aggregate the consequences for each waste management method and generate a risk index for each method. A low risk index means the corresponding disposal method causes less harm to the environment than other methods, as opposed to a high index which means more harm. These indices can then be reduced to the 0 to 1 scale for use in the SWIPES equation.

Unfortunately, risk assessment does not fall short of difficulties when implemented. As mentioned in the risk assessment section, both the scoring of the outcomes of consequences and the determination of the weighing factors for the consequences require human judgement. Without a standard to follow, the assignment of these two sets of values becomes rather arbitrary. While their ultimate values influence the measure of the risk index, there may be disputes on what numbers to be assigned to those parameters. Another difficulty is the lack of data. To evaluate environmental hazards in risk assessment, specific data are needed and the probabilities of the occurrence of each possible outcome for the consequences have to be known. Many such data are unavailable for this project. Risk assessment is geared toward assessing health related consequences such as the number of people who contract a particular disease as a result of inhaling a given amount of pollutants emitted from an incineration plant. Those data may even be further broken down according to the seriousness of the illness. If this kind of data is to be established, extensive research is needed, a lot of assumptions have to be made, and above all, the task is difficult to accomplish without a team of manpower.

In conclusion, even though risk assessment enables the direct evaluation of the scientific and technological aspects of various environmental impacts, if employed, the difficulties of having limited data and a relatively high degree of uncertainties have to be dealt with.

THE ADOPTED METHOD

As a result, the methods discussed above are not used directly in this thesis. However, both the cost-benefit analysis and the Tellus Packaging Study have inspired the design of the adopted method. This current method rates the solid waste management options based on the external costs of packaging materials quantified in the Tellus' research and two other environmental costs. Relevant costs incurred in each option for each material are aggregated, compared with the costs of other options, and converted to scores between zero and one. This method bears similarities to that in CBA, which aggregates costs and benefits of an activity for comparison.

There are five major factors contributing to the rating of the environmental impacts of various solid waste management methods:

1) the environmental releases during materials production and disposal,

2) the conventional costs of disposal,

3) the depletion of non-renewable natural resources,

4) the energy consumption in virgin versus recycled production and

5) the depletion of landfill space.

The first two factors are quantified as external costs for packaging materials in the Tellus Packaging Study. External costs are used as the basis for the rating because these are costs that our society and our environment are burdened with for handling the packaging materials once they are produced.

The external costs include the conventional cost of disposal and the environmental costs of disposal and production. Tellus has assessed the conventional and environmental costs of disposal for landfilling, recycling and incineration for each type of packaging waste. The environmental cost of production was quantified for the production of packaging materials using virgin resources; and for some materials for which information was available, the impacts of production using recycled materials were also quantified.

In addition to the external costs, the energy cost of production and the cost relating to raw material depletion are included for the ratings. The energy cost of production is determined by multiplying the amount of process energy required in the production of a packaging material by the average energy price per million Btu. The costs relating to the raw material depletion are derived from the market prices of the exhaustible natural resources that are input for material production. A more detailed discussion of these two costs is presented in Chapter V. The cost of depleting landfill space is not determined because it is very difficult to assign a price to it when the total capacity of landfill in the country is not known and the ease of siting for a new landfill cannot be quantified.

It is conceivable that for each material, different options of solid waste management incur different environmental costs. The itemized costs listed in Table 3.5 for aluminum serve as an example of the types of environmental costs that are included in the ratings for each material.

Table 3.5 Per Ton Environmental Costs of Aluminum

	Landfilling	Incineration	Recycling
Conventional Cost of Disposal*	279.66	189.24	-553.5
Environmental Cost of Disposal*	3.04	3.94	11.53
Environmental Cost of Production*	1933	1933	1933
Environmental Impact Benefit for Using Recycled Materials*	—	—	-1620
Cost of Natural Resources	128.36	128.36	0
Energy Cost of Production	1110.72	1110.72	44.43
Total Environmental Cost	3454.78	3236.90	-184.54

Note: Positive Values represent "Cost", Negative Values represent "Benefits"
*Source: Tellus, 1992

The benefit of using recycled material in production is the cost difference between the environmental impact of virgin material production and recycled material production. This benefit is credited only to recycling because only this waste management approach makes the production of recycled material possible.

Each material carries a set of environmental costs for three waste management methods as shown in Table 3.5. These environmental costs, as will be shown in Chapter VII, range from a low of $-184.54/ton of material for the recycling of aluminum to a high of $3454.78/ton for landfilling of aluminum. The total environmental costs of other materials lie between these two figures. All these total environmental costs can be converted to the 0 to 1 scale only if a high cost can be anchored at the value of 0 and a low cost be assigned to a value of 1. The rest of the costs can then be converted proportional to those costs. In this work, zero cost is determined to correspond to the index of 1, which means the disposal method is charging no costs to the environment. This disposal method is "reduce" - an option that ranks highest in EPA's hierarchy of solid waste management. Here, source reduction refers to reducing the amount of packaging materials that enters the solid waste stream by buying fewer products, designing products with longer useful lives and reducing the amount of packaging material used. There is a claim stating that source reduction has negative effects on the economy for consumer goods and services, but this cost is out of the scope of solid waste management. The current notion of source reduction bypasses this cost and considers only the environmental impacts and the basic conventional costs required to implement the program. By simply producing less packaging wastes, source reduction causes no harm but only benefits to the environment. In the paper, "Does the Solid Waste Management Hierarchy Make Sense?", it was noted that source reduction education programs cost $9/ton. Compared to the hundreds of dollars of external costs charged by other disposal options, this cost is relatively small. Therefore, source reduction can be assumed a zero cost and corresponds to an index of 1.

An index of 0, as defined in the SWIPES project, states that the corresponding waste management is considered as providing no benefit to the environment (Saputo, 1992). In practice, it is difficult to assign to that value an environmental cost that would match that definition. Even landfilling, which is given the lowest priority in the EPA's hierarchy, is not completely useless to the environment. It is one of the feasible and necessary approaches that helps manage solid waste. Without it, a large part of our garbage would be sitting by the curbside. Therefore, for the convenience of assigning a baseline cost that corresponds to this index of 0, the average of the total environmental costs due to landfilling is used. This cost turns out to be $1112/ton, as will be shown in Chapter VII. Using a landfill cost as the lower bound of the scale is in agreement with EPA's hierarchy which prescribes landfilling as the least desirable option.

Now that a value of 1 corresponds to a zero cost and a value of 0 corresponds to the average cost of landfilling, all the in-between total environmental costs can be converted to this 0-1 scale by proportion. The detailed procedure will be discussed in Chapter VII.

Unlike EPA's hierarchy in which source reduction includes reuse, here, the two methods are assessed separately. Large scale reuse of packages is usually carried out by manufacturers or packagers. Reusing packages requires the transportation of the packages back to the packagers. Those used packages are then subject to thorough cleaning before they can be reused. The factors that differentiate source reduction from reuse are the environmental impacts such as pollution due to transportation and pollution generated from the energy required for cleaning the packages for reuse. An environmental cost thus exists for reuse. Therefore, it should be rated differently from source reduction and be given a score less than 1.

Reuse, recycle, composting and incineration are the waste management options this project has to assign values to. However, the Tellus Packaging Study assessed recycling, incineration and landfilling only. There is little information on the reuse of packaging materials. The only consumer package known for large scale reuse was returnable glass bottles, but this practice has been stopped nowadays. Composting of packaging material is not carried out extensively either. Paper products are the only type of packaging materials that can be composted, but current composting programs handle mostly yard wastes. Reuse and composting therefore cannot be quantified at this stage. However, the known environmental effects of reuse and composting can still be described and discussed wherever possible. In addition, two proposed equations are written for the calculation of environmental costs for reuse and composting when data become available in the future. These descriptions and the equations will be presented along with other numerical results in Chapters VI and VII. Similar to the approach of CBA, describing the unquantified offers the user of the analysis an explicit and most comprehensive view of the subject possible for the time being.

Rationales for the Method Adopted

There are two approaches to producing the ratings. One can list and then evaluate the various impacts of the disposal options. A score can be assigned to each disposal option based on implicit assessment and comparison of those impacts, which may be presented in different units. Alternatively, one can first standardize the impacts to one common unit, enabling an explicit aggregation and comparison, then assign final scores based on the values of the disposal options in terms of that common unit. In the earlier part of this thesis, it has been implied that the use of an explicit method is preferred. That is why surveying, which is based on implicit judgment, was rejected. The question now becomes what common unit should be used for the explicit comparison. Is the monetary unit the only choice?

For the same reason as that in CBA, the use of dollar amounts as the unit of measurement in this project is really inevitable. There are no other units more practical and convenient in application. It is true that not all matters amount to a price, but money is so common and widespread in our everyday lives, it makes the conversion of effects from other units to dollar values easier than if other units are used.

Once it is decided that dollar values be used, CBA becomes a potential candidate for this project. To apply CBA means to list the environmental advantages and disadvantages of each waste management option, quantify these effects, convert them into monetary units, and calculate the net present value of each option. These net present values can then be converted to 0-1 scores. If sufficient data can be found for the list of environmental impacts, the rest of the CBA procedure is straightforward. Unfortunately, collecting data for such items and finding a scheme to monetize them is far from easy.

The search for environmental data for this project was dissatisfactory. Scanty and scattered data cannot fulfill what is required of this project. *Data*

Summary of Municipal Solid Waste Management Alternatives (NREL, 1992) contains data for existing waste management technologies on costs, environmental releases, energy requirements and production. However, the 12-volume report provides information on each disposal technology as a whole and does not include any research on the impacts of different materials on the disposal system. Some waste management journals have reported disposal costs of specific packaging materials, but their research aimed at providing general ideas rather than developing an in-depth database for the subject. Thus, the quality of those data is deemed questionable. The Tellus Packaging Study is the only source found that contains detailed information on the conventional and environmental impacts of managing various kinds of packaging waste by different disposal methods. The objective of this project to rate the disposal options can be best performed by using data from a common source. Rather than trying to compile data obtained from different researchers, the use of a coherent source of data would make comparing subjects easier and the rating results more reliable. Since the research findings provided by Tellus Packaging Study meets the need of this project, and its results are the most elaborate among all other sources found, it is in this project's best interest to use Tellus' data to carry out the rating.

It turns out a thorough CBA does not need to be carried out because Tellus has already provided most of the environmental impacts of the disposal activities in monetary units. Tellus' research approach was based on environmental economics methodology, the same stream of approach that will be used if CBA is applied. Even though no actual cost benefit analysis is carried out in this project, the concept of assessing alternative options of a public activity based on costs and benefits of priced and originally unpriced impacts is, however, extracted from CBA and transferred to use here.

In the Tellus Packaging Study, the impacts of various waste management options were identified for each type of packaging material because different materials will exhibit different impacts in different disposal environments due to the materials' differences in physical and chemical properties. In other words, each packaging material will have its own set of ratings for the disposal options that they are eligible for.

As it is listed in Table 3.5, the rating is based on six types of costs: conventional costs of disposal, environmental costs of production and disposal, the benefits of using recycled materials, energy cost of production and the monetary costs of exhaustible raw materials. The conventional cost of disposal includes monetary costs of collecting, processing and disposing of the solid waste. Without packaging waste, our society does not have to spend money and efforts to manage the waste. Having to pay for such costs is thus one type of negative impact. This impact, even though it is more economic than environmental, should be included as one of the factors in the rating. In actual practice, economic and environmental factors are interdependent in determining the applicability of a solid waste management approach. When an environmental impact is recognized as severely hazardous to human health and to the ecosystem, our society will place a high priority and be willing to allocate more resources to reduce the impact, but at the same time, the fact that our resources are limited does not allow us to spend whatever it takes to carry out a remedial activity. The point is that if there is an environmentally benign method that can effectively manage the solid waste, but this method is exceptionally costly to implement, the chance is the society will not be able to afford it, but will look for a less expensive alternative. Therefore, there is usually a compromise and yet a tie between the environmental and economic considerations. Identifying the degree of environmental soundness of a waste management method alone does not determine its practical existence. The conventional cost has to be taken into account. In addition, part of the monetary cost of disposal is attributed to reflect the scarcity of the landfill space through high tipping fees, and the use values of the waste through the amount of revenue being generated from recycling and electricity generation. For this and the above reasons, the conventional cost of disposal should take a part in determining the ratings of the disposal options.

The environmental cost of production also has to be included in the ratings because through different approaches of solid waste management, the production input and output of materials and the associated environmental impacts will be affected. If a package is prevented at its source, environmental releases due to its production are prevented. When a package is being reused, depending on the total number of uses, its production impact per use will only be a fraction of the production impact of a single-use package. If the material is recovered by recycling, the supply of recyclable materials to the recycled material production avoids virgin material extraction and therefore, lowers the total amount of environmental releases due to material production. For this reason, recycling is credited with the benefit that is the difference between environmental costs for virgin and recycled material production. This benefit is the cost avoided due to the supply of recyclable materials by recycling. In conclusion, the environmental cost of production is interrelated to disposal management methods. The evaluation of the environmental impacts of disposal methods cannot be completed without taking the environmental cost of production into account.

The energy cost of production is included in the rating because energy is produced from exhaustible natural resources like coal, petroleum and natural gas. The saving of energy from recycled production is equivalent to the savings of natural resources, and therefore is beneficial to the environment. The energy cost is thus a measure of the environmental cost in that aspect.

Other natural resources that can be saved through recycled production are the minerals, petroleum or wood that would have been consumed during the production of the virgin material equivalent. An environmental cost should be charged to the production that leads to the depletion of these limited resources. The market prices of these raw materials are chosen to be an indicator of this type of environmental cost.

The last cost item that should be included in the rating, with little argument, is the environmental cost of disposal. This cost, as will be shown later, turns out to be small when compared to the other costs mentioned above.

CHAPTER IV

MATERIALS PRODUCTION TECHNOLOGIES AND DATA

During the lifecycle of a packaging material, the pollutants emitted during materials production are the major source of environmental releases. The levels of these releases are significantly higher than those from packaging disposal. The method that the Tellus Institute used to monetize the environmental releases from materials production has been explained in Chapter III. In this chapter, the production processes for each of the materials evaluated in this project are described. These processes and the corresponding data are summarized from the Tellus Packaging Study unless otherwise stated. The Tellus study adopts the most predominant method of production when several exist for a material. The emissions associated with the production processes are also reported in this chapter wherever information is available.

PAPER

Virgin and Recycled Paper Production

The processes of paper and paperboard production are summarized in Table 4.1, and the associated environmental releases are also presented in the table with indications of whether those releases are included in the environmental

Table 4.1 Production of Virgin Paper and Paperboard

Process	Pollution	Remarks
1). Cut Trees from stump	- particulates from burning branch and leaf wastes	not quantified
2). Transport logs to a truck	- pollution from energy use by trucks, cranes, loaders, etc.	quantified
	- erosion that causes siltation of lakes, and streams if logging roads are not good	not quantified
3). Transport logs to mills or woodyard	- emissions from trucks and/or railcars	not quantified
4). Wash logs in a mechanical conveyor system	- water effluent	not quantified
5). Debark i). Dry debarking by friction, or ii). Wet debarking by friction, or iii). Hydraulic debarking	- water effluent	not quantified
	- pollutants from burning bark to recover energy	not quantified
6). Cut debarked logs into chips	- wood-dust	not quantified
7). Screen chips i). Accepted ---> pulping ii). Screened-out fines ---> combusted	- air emissions due to burning	quantified

Process	Pollution	Remarks
8). Pulping		
A). Kraft Pulping		
i) Cook chips in a digester with $NaOH + Na_2S$ solution	- odor	not quantified
Cooked pulp ---> blow tank		
Relief gases ---> turpentine condenser	- air emissions	quantified
ii) Send pulp to brown stock washers		
Pulp ---> screening	- air emissions	quantified
Black liquor (i.e. spent cooking liquor) ---> black liquor recovery system♣		
B). NSSC Pulping	- controlled air emissions (only uncontrolled emissions are quantified)	not quantified
i) Cook chips in digester with $Na_2CO_3 + Na_2SO_3$ or $(NH_4)_2SO_3$ solution		
ii) Send softened chips to a blow tank		
iii) Break chips into fibers with a disc refiner		
iv) Wash pulp in a pulp washing system		
9). Screen pulp		
10). Centrifugal cleaning of pulp		
11). Thicken pulp on a decker		
12). Bleaching (if needed)	- chlorine, chloroform (only uncontrolled emissions are quantified)	not quantified

♣ See “Black Liquor Recovery System” below

Process	Pollution	Remarks
13). Sheet and Board Forming		
A). Refining		
- blend 2 to 3 types of pulp to meet specifications		
B). Dilute the pulp		
C). Add additives to achieve properties		
D). Send stock (refined pulp) to paper-making (by cylinder or fourdrinier machine)		
i) Form a web of fiber on a paper machine wire	- air emissions and water effluent	quantified
ii) Press water out of the web	- air emissions and water effluent	quantified
iii) Drive off remaining water by heat	- air emissions	quantified
iv) Apply coating as needed	- air emissions	quantified
♣ **Black Liquor Recovery**		
1). Weak black liquor is directed from brown stock washers in kraft pulping to an oxidation tower to lessen sulfur emissions	- no controlled air emissions detected	
2). Concentrate weak black liquor to ~55% solid content in multiple effect evaporator system	- no controlled air emissions detected	
3). Further concentrate black liquor by a direct or indirect contact evaporator i) Strong black liquor ---> storage tank ii) Flue gases ---> electrostatic precipitator		

Process	Pollution	Remarks
4). Burn strong black liquor in recovery furnace i) Organic fraction provides heat credit ii) Na_2S, Na_2CO_3 and other inorganics form a molten smelt	- air emissions	quantified
5). Smelt is treated in a Kraft White Liquor System✧		

✧ White Liquor System

Process	Pollution	Remarks
1). Put smelt from black liquor recovery into a dissolving tank ---> form a green liquor	- air emissions	quantified
2). Send to a green liquor clarifier i) Dregs (suspended solids) to dreg washer to recover Na_2CO_3 and sulfide. Send remains to sewer ii) Mix green liquor with water and re-burned lime. Send to slaker to form a slaked-green liquor. Remove unreacted particles for disposal after washing.		
3). Send the mixture to a causticizer (agitated tanks) for converting Na_2CO_3 to NaOH.		
4). Transfer overflow to white liquor clarifier i) white liquor overflows to storage tank for reuse in the pulping process ii) lime sludge remains at bottom		
5). Wash lime sludge		
6). Dewater lime mud		
7). Dry lime mud in lime kiln and convert to CaO	- air emissions	quantified

cost of paper production in the Tellus study. Emissions released from energy production are not listed in the tables. These emissions are, however, included in the environmental cost. Sources of energy include gasoline and diesel fuels used by trucks and mobile equipment, electricity generated from steam, natural gas, oil, wood waste and coal.

In addition to the information given in Table 4.1, a few remarks have to be made: pulping can be classified into three categories, namely chemical, semi-chemical, and mechanical pulping. Kraft pulping is one type of chemical pulping in which the extraction of cellulosic fibers from a mat of fibers and lignin is achieved by the chemical reaction between the lignin and the chemicals. Semi-chemical pulping softens wood chips by chemical reaction, and is followed by mechanical action to convert chips into pulp. NSCC (neutral sulfite semi-chemical) pulping is the primary type in this category. Mechanical pulping uses grinding action to fiberize pulpwood. This method does not separate lignins from fibers, therefore, the paper produced is of a lower grade with inferiority in brightness and strength. Since almost all paper and paperboard packaging are made from kraft pulping and NSSC pulping, Tellus modelled the environmental impacts of paper production based on these two methods.

As described in step 8ii in Table 4.1, the solution that has been used in digesting the wood chips, called the black liquor, has to pass through a recovery system. The purposes of this effort are to

a). remove the water,

b). convert the sodium sulfate to sodium sulfide for further recovery, and

c). burn residual organic materials in the liquor for energy recovery.

Included in the black liquor recovery process is the kraft white liquor recovery process. A white liquor recovery system converts the unburned inorganic smelt from the black liquor recovery system to reusable white liquor for kraft pulping. Comparing to kraft pulping, NSSC pulping does not have to employ extensive recovery processes because of the relatively low organic content of its spent liquor and the low quality of its recovered products.

In addition to the principle pulping processes, the production of five types of chemicals used in the pulping and bleaching processes are also included in the quantification. Table 4.2 lists those chemicals and the types of associated pollutants quantified. Emissions due to energy generation for use in producing these chemicals are also quantified.

Table 4.2 Pollutants from Production of Chemicals for Papermaking

Chemical	Type of Emissions Data	Types of Pollutants
Lime	Controlled	Air Pollutants: TSP, PM_{10}, SO_x, NO_x, VOCs, CO, Pb, and CH_4
Chlorine	Controlled	Air Pollutants: TSP, PM_{10}, SO_x, NO_x, VOCs, CO, Pb, CH_4 and Cl_2 Water Pollutants: Cl_2, Sb, As, Cd, Cr, Cu, Pb, Hg, Ni, Ag and Zn
Caustic Soda	Controlled	Same as those in chlorine production
Soda Ash	Uncontrolled only	
Sodium Sulfate	No controlled or uncontrolled data	

Pulp production from wastepaper is much simpler than virgin pulp production, and the processes are described in Table 4.3.

Among the five types of paper and paperboard evaluated by Tellus, the data on unbleached, coated folding boxboard, linerboard, and corrugating medium are used in this project.

Folding Boxboard: Unbleached, coated folding boxboard: this type of virgin paper is generally made from unbleached kraft pulp and then clay coated. Its recycled equivalent can be made 100% from waste paper of various kinds.

Linerboard: Virgin linerboard is generally made from kraft pulp and is used as a facing material for corrugated boxboard. Recycled linerboard can be made 100% from old corrugated boxboard, corrugated box clippings and some combination of unbleached kraft pulp and waste paper.

Corrugated Medium: Used as the flute between facing materials in a corrugated box, this material is usually made by NSSC pulping. Its recycled equivalent can be made from waste papers of various grades such as old corrugated cardboard, old newspaper and mixed waste. Unfortunately, only uncontrolled air emission data are available for NSSC pulping, therefore, the environmental impact of this material is likely to be underestimated.

<u>Environmental Impacts</u>

Table 4.4 presents the process energy needed for virgin and recycled paper packaging production. The environmental costs of production can be found in Table 3.4. Virgin corrugating medium takes less energy to produce than other

Table 4.3 Pulping and Papermaking from Wastepaper

Process	Pollutants	Remark
1). Repulp		
A). Mix wastepaper with heated water in a hydropulper to separate fibers i) contaminants removed from surface ii) send slurry to a centrifugal cleaning system to remove glass, gravel, etc.	- water pollutants (from inks and coatings)	quantified
B).Pass clean slurry to vibrating or high pressure screens to remove non-fibrous materials (may also use magnetic screening devices)		
C).Pulp may be washed and bleached to remove inks, clays, chemicals if required		
2). PaperMaking Send pulp to papermaking machine. The papermaking process is same as the same as virgin paper production	- air pollutants from various stages of recycled paper production are assumed to be the same as those in virgin production	quantified

papers because NSSC pulping, the method from which its pulp is produced, is more energy efficient than kraft pulping. The total environmental cost of producing virgin corrugating medium is also substantially lower than other papers partly because of the lower energy input, which leads to lower emissions associated with energy production. At the same time, kraft pulping, from which linerboard and folding boxboard are made, has a high environmental cost because of particulate emissions. Also, controlled air emission data for NSSC pulping are not included in the quantification because they are unavailable. All these factors contribute to the low environmental production cost for virgin corrugating medium when compared with other papers on the list.

Table 4.4 Process Energy for Paper/Paperboard Production (MMBtu/ton)

Paper Type	Virgin	Recycled
Unbleached, coated folding boxboard	29.96	22.89
Linerboard	30.73	23.12
Corrugating Medium	22.30	22.13
Corrugated Boxboard (31% corrugating medium, + 69% linerboard)	24.91	22.44

While folding boxboard and linerboard are showing lower environmental costs for recycled production, corrugating medium shows an opposite trend. Recycled corrugating medium costs more than twice as much to produce as virgin production. The large difference is due to the fact that recycled production of this material showed a substantial amount of heavy metal emissions during the deinking process for wastepaper. In addition, more energy is needed for its pulping process because old corrugated cardboard is more difficult to break up than other papers. As a result, NO_x and SO_x emissions that are released primarily during energy production are 120% higher in recycled than in virgin corrugating medium production. Particulate emission is also 23% higher in recycled production. The slightly lower emissions of CO and VOCs (volatile organic compounds) in recycled production are not enough to offset the higher environmental costs charged by the increased amounts of other pollutants. It should be noted that the use of less toxic ink nowadays should significantly lower the high cost that is imposed by heavy metal emissions.

When comparing the environmental releases between virgin and recycled paper production, all three types of paper packaging show higher air emissions of SO_x, NO_x, and heavy metals such as Cr, Cu, Pb, Hg, Ni and Zn in their water effluent. The environmental costs of producing the virgin corrugating medium and all three recycling papers are dominated by SO_x and NO_x (50% and 20% respectively of the total costs). High emissions of SO_x and NO_x in recycled production are due to the need for recycled production facilities to purchase electricity, whereas virgin production obtains some of its energy from its waste products. The high levels of heavy metals found in the water effluent from the recycling processes originate from the liberation of inks and coating that are contained in the wastepaper during the re-pulping process.

Nevertheless, folding boxboard and linerboard are associated with a substantial decrease in particulate emissions and lower CO and VOCs emissions. Recycling folding boxboard also decreases HS releases to a large extent. These decreases in emissions are enough to offset the higher emissions of NO_x, SO_x, an

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and heavy metals, therefore resulting in lower environmental production costs for boxboard and linerboard.

ALUMINUM

Virgin Aluminum Production

Aluminum is obtained from bauxite, a raw material that contains mainly hydrated alumina (Al_2O_3) and various amounts of iron oxide, titanium oxide, silica, and other impurities.

To extract aluminum from bauxite, three major processes are carried out:

1). bauxite ore mining and refining (Figure 4.1),

2). alumina production (Figure 4.2), and

3). aluminum production (Figure 4.3).

The production steps involved in each of these three processes are described in the figures indicated.

In bauxite refining, the drying of the processed bauxite is the most energy intensive step, which accounts for about 80% of the energy used in this process. Controlled air emissions are identified for this process. Total suspended particulates (TSP) are the only controlled emission detected in bauxite mining and processing. Also, various types of air pollutants (TSP, PM_{10}, SO_x, NO_x, VOCs, CO, Pb, & CH_4) are released due to the energy generation for the process.

Refined bauxite proceeds to the Bayer Process, a method that purifies bauxite to a high grade Al_2O_3. Lime and caustic soda are the two additives required in this process.The emissions associated with lime and caustic production and those associated with their process energy are quantified. Their total level of environmental impacts, however, is much lower than that imposed by

Figure 4.1 Step I: Bauxite Mining and Refining

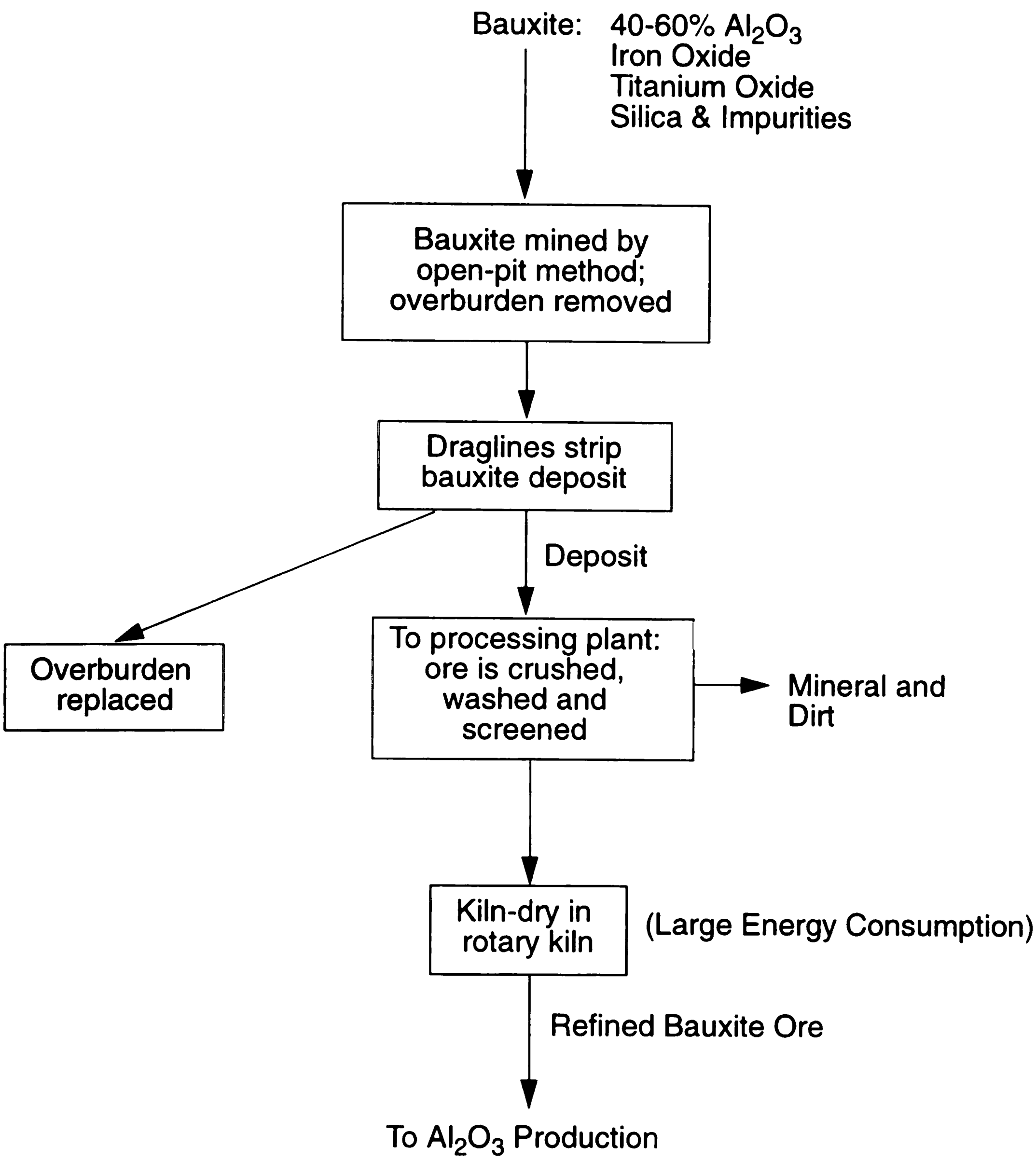


Figure 4.2 Step II: Al_2O_3 Production (The Bayer Process)

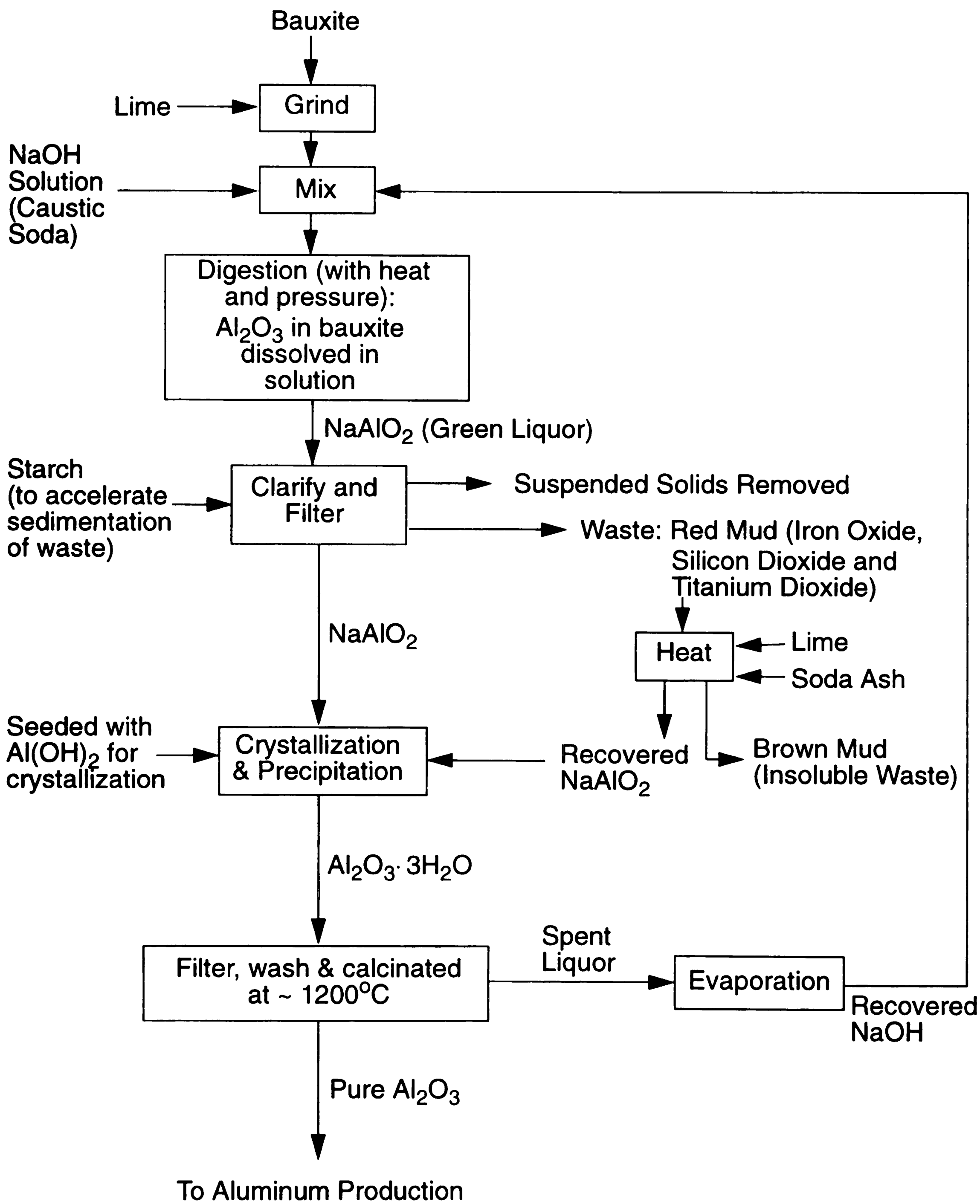
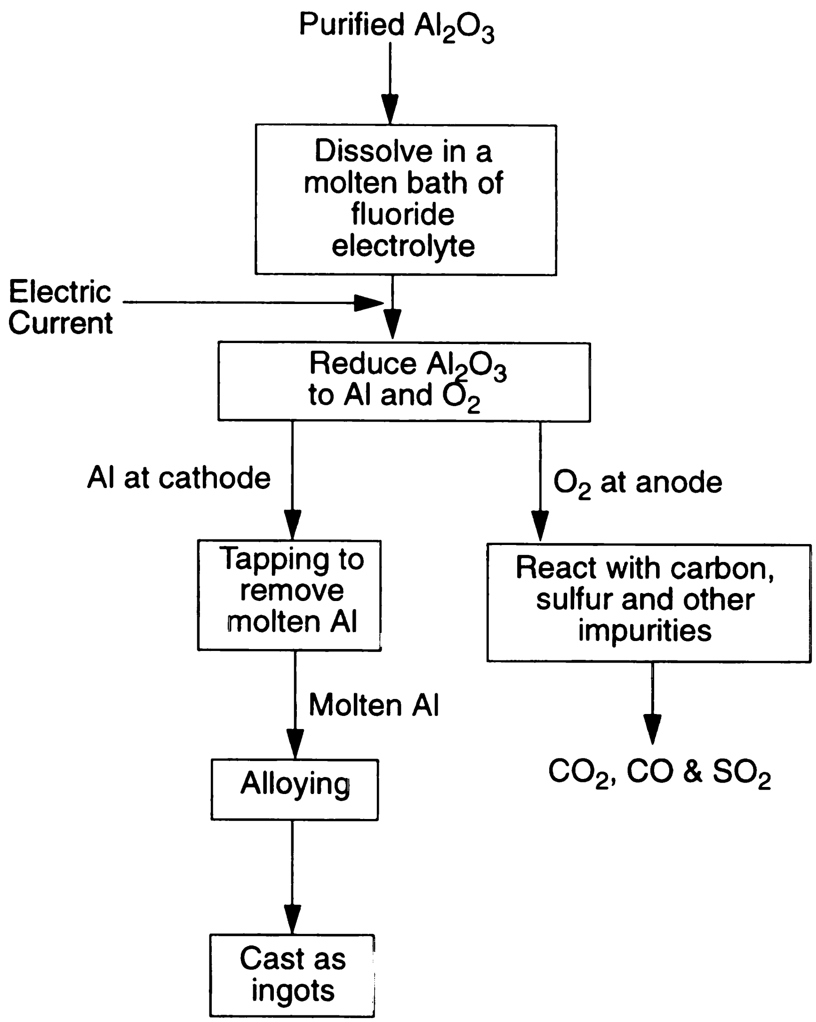


Figure 4.3 Step III: Aluminum Production (Hall-Heroult Process)



the alumina production and its associated process energy. TSP emitted from alumina production is over 200% higher than TSP emission from lime and caustic production combined. The levels of PM_{10}, SO_x, NO_x, lead and methane emissions associated with the process energy of Al_2O_3 production are also substantially higher than those from the production of the additives. The environmental impact of the Bayer Process is attributed to its intensive use of energy.

Once pure Al_2O_3 is formed, it is reduced to aluminum by the Hall-Heroult Process. This process begins with the anode and cathode production for the electrolytic cell. Petroleum coke and pitch binder are the two materials involved in the anode production. Cathode production involves anthracite and pitch binder. The energy inputs for the electrode production and the air and water emissions associated with the petroleum coke production are quantified. Tellus does not indicate any inclusion of emissions from the production of anthracite.

When a low-voltage direct current is applied to the molten bath in the Hall-Heroult Process, Al_2O_3 is reduced to aluminum and deposited at the cathode while oxygen reacts with the carbon anode and other impurities to form CO_2, CO, and other by-products. Molten aluminum is then removed from the cathode by "tapping", a process in which the aluminum is sucked up with a ladder or a crucible. Alloying and casting are the procedures that follow.

The Hall-Heroult Process is an energy intensive operation. Considering a total of 208 MMBtu/ton of aluminum used for the entire aluminum making process, from bauxite extraction to the reduction to aluminum, 67.8% of this energy is spent on the electricity required in the electrolytic reduction process alone. (9.5% is used in the cathode and anode production). 22% of the total energy is allocated to the mining and refining of bauxite and the forming of Al_2O_3. The remaining energy is consumed by the additives involved.

Recycled Aluminum Production

As indicated in Figure 4.4, the production of recycled aluminum cans does not involve bauxite extraction or alumina forming. Recovered aluminum is remelted and the melt is converted into the required chemical specifications. Aluminum beverage cans consist of two portions, one the body and the other the lid. The latter is alloyed with a higher magnesium content to enhance the strength. When an entire can is recycled into a body stock, the melt is composed of a higher magnesium content than required. Demagging is the process in which chemical agents are used to remove the excess amount of magnesium.

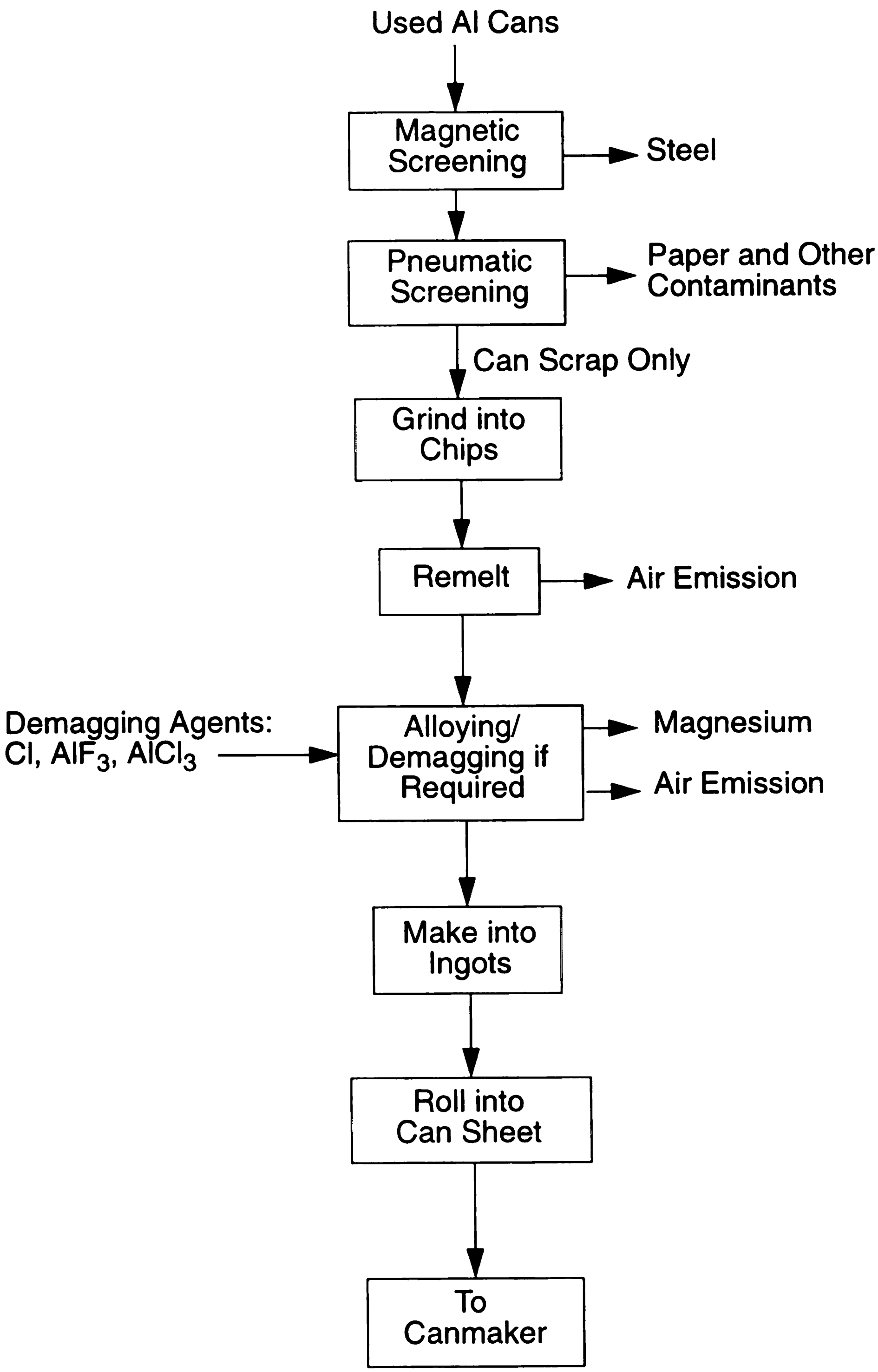
About 78% of the energy used in recycled aluminum production is spent on demagging and melting. It is noted that demagging produces noxious halogen and halogen compound emissions, as well as PM. Emission factors for these compounds are, however, unavailable. The only controlled air emission factor quantified for recycled aluminum production is TSP.

Environmental Impacts

The amount of energy required for aluminum recycling is only 8.32 MMBtu/ton. Recycling aluminum thus saves energy by 96%. This significant saving is due to the avoidance of bauxite mining and refining, alumina forming, and the electrolytic reduction of alumina to aluminum. Most of the energy used in recycling aluminum is for delacquering and melting.

The environmental costs of aluminum production can be found in Table

Figure 4.4 Recycled Aluminum Production



3.4. The cost associated with the criteria air pollutants is reduced by 79%, and that with the toxic and carcinogenic pollutants is reduced by 99.8% when aluminum is recycled. Since the recycling process eliminates alumina and aluminum formation and requires much less energy, the environmental costs associated with various pollutants are reduced substantially. In virgin aluminum production, its high overall environmental cost is contributed by the high costs produced by NO_x, SO_x, particulates, fluoride, and lead emissions.

GLASS

Virgin Glass Production

Glass containers for packaging are made from soda-lime glass, which is typically composed of 70% SiO_2, 15% Na_2O, 12% CaO, 2% Al_2O_3, and 1% of other minor constituents.

Four major processes are involved in manufacturing glass containers:

1). mining and processing of raw materials,

2). mixing raw materials,

3). melting and refining raw materials; and

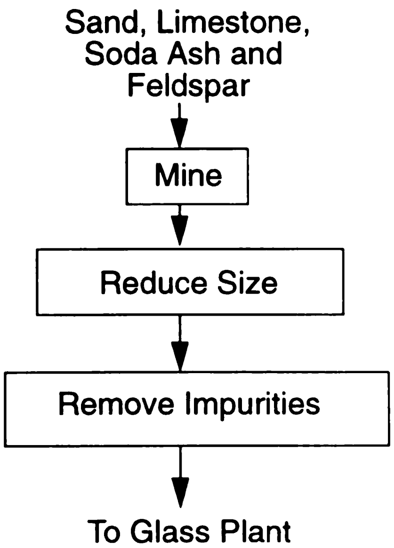
4). forming molten glass and manufacturing glass containers.

The details of each process and the environmental releases included in quantifying the environmental production cost are described in Figure 4.5.

When the processed raw materials are fed into the furnace, they are melted at a section of the furnace, called the melter, where the temperature is kept at around 2800^{o}F. Molten glass is then flowed through a narrow opening from the bottom to the refiner. This mechanism holds back impurities that flow on the surface of the melt. At the refiner, refining agents are added to reduce seeds and

Figure 4.5 Manufacturing of Virgin Glass

Step I: Raw Material Mining and Processing

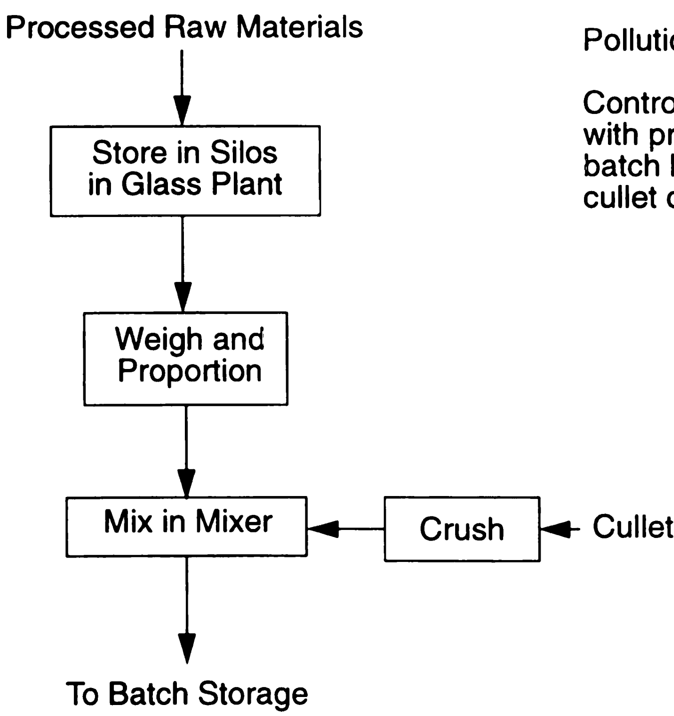


Pollution Data Included:

1) Controlled emissions of limestone mining and processing
2) Controlled emissions associated with limestone process energy
3) Emissions associated with synthetic soda ash process energy

Other emissions data are unavailable

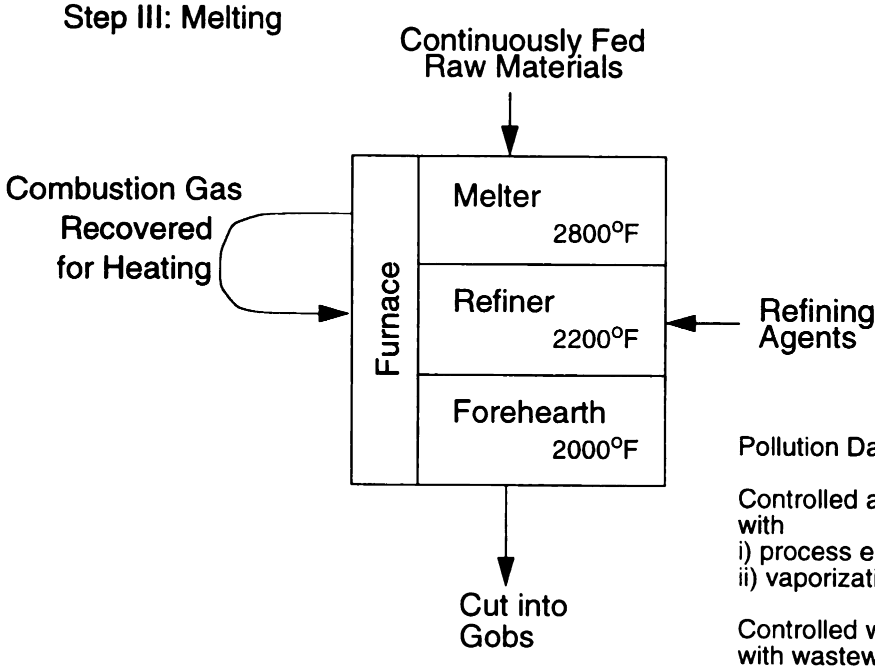
Step II: Raw Material Mixing



Pollution Data Included:

Controlled emissions associated with process energy used in batch handling, mixing and cullet crushing.

Step III: Melting

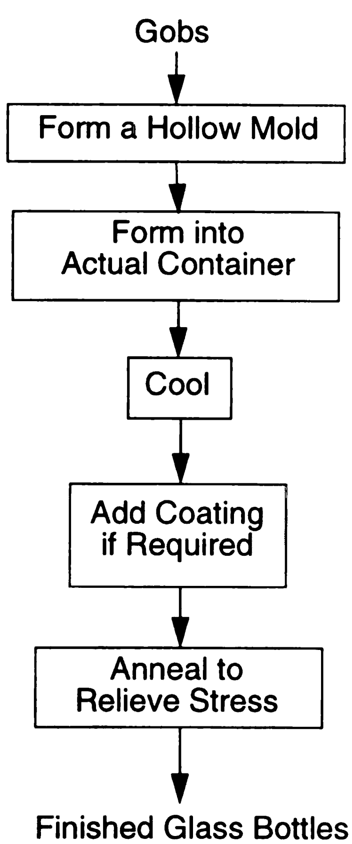


Pollution Data Included:

Controlled air emissions associated with
i) process energy
ii) vaporization of raw materials

Controlled water emissions associated with wastewater from non-contact cooling in the melting process.

Step IV: Forming



Pollution Data Included:

Controlled air emissions associated with process energy for fining and conditioning; forming and annealing.

blisters. The temperature of the molten glass is gradually lowered to 2200°F at this stage. The molten glass then enters the forehearth where temperature is at 2000°F. From here, the molten glass is cut into gobs, each will then be formed into one bottle.

By the time the bottles are formed, a lot of stresses are arrested in the glass due to prior heating and cooling. Annealing is a thermal process that eliminates those stresses: glass bottles are placed at about 1000°F for an interval of time and are then allowed to slowly cool to room temperature.

Glass Recycling

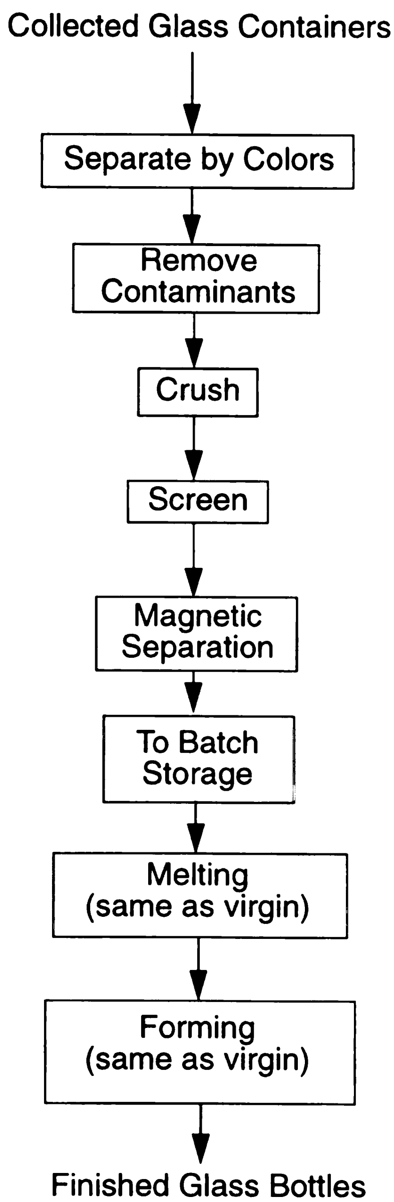
In virgin or recycled glass manufacturing, cullet is used as an input material. Around 30% cullet is used in the production of virgin glass containers. In the Tellus Packaging Study, it is assumed that recycled glass is made of 100% cullet.

When recycled glass is used, mining, processing, and mixing of raw materials are eliminated. Instead, the production begins with bottle sorting, contaminants removal and size reduction (Figure 4.6). Once the collected glass is processed for remanufacturing, it is melted and formed into new bottles the same ways as virgin glass.

Environmental Impacts

The amount of energy input for manufacturing glass containers is lower than other packaging materials. The total energy required for virgin glass production is 13.5 MMBtu/ton and that for recycled glass is 9.92 MMBtu/ton.

Figure 4.6 Recycled Glass Production



Pollution Data Included:

Same as those listed in steps III and IV of 'Virgin Glass Production'

Manufacturing from recycled glass saves energy due to the use of cullet which melts at a lower temperature than the raw materials. It has been found that every 1% increase in cullet provides 0.25% in energy savings in the melting process. According to Tellus' data, 16% of the energy input in virgin glass production is attributed to raw material mining and processing, and 50% is used in the melting process. In conjunction with the lower energy use is the reduction of the associated air emissions. The lowering of the furnace temperature also lowers the vaporization of volatile materials in the melt.

Seven types of controlled air pollutants are found to contribute to the environmental cost of both virgin and recycled glass. They are CO, NO_x, particulates, SO_x, VOCs, and lead. Recycling glass reduces CO and SO_x releases by more than 54%, NO_x by 38%, particulate emissions by 68% and lead by 90%. As a result, the environmental cost of recycled glass is much lower than that of virgin glass. This cost, however, may be lower than the true environmental cost since no data were found for the controlled emissions associated with feldspar, sand, and soda ash mining and processing.

STEEL

Virgin Steel Production

Steel cans that are used for food and beverages are one major type of steel packaging. It is the category that Tellus Packaging Study focused on when evaluating steel packaging.

Two types of furnaces are predominant in the current steel making industry: the basic oxygen furnace (BOF) and the electric arc furnace (EAF). While an EAF can accept up to 100% steel scrap, BOF can allow only 30-40% of scrap. The steel that is made of 100% scrap, however, does not have the properties required for the production of thin steel sheets that are used to make steel cans. Consequently, steel cans are made primarily from BOFs. Tellus thus modelled the production of steel packaging with the use of BOFs and assumed the recycled content of recycled steel cans to be 40%.

The production of steel requires the input of several raw materials. The first few processes of the production include the acquisition of these raw materials:

1). iron ore mining and processing,

2). limestone quarrying and lime formation,

3). coal mining and processing, and

4). coke formation from coal.

Limestone and lime are used as fluxing agents. Coal is used for the production of coke, a major source of energy in the steel industry. Steel is an alloy of iron and carbon. Coal is therefore used as a source of carbon in the steel production as well. The production processes for the raw materials and the processes for steel making can be understood from Table 4.5. The following processes follow raw materials processing:

5). sintering,

6). pig iron production, and

7). steelmaking.

Sintering is a thermal process that allows particles of materials to agglomerate into one piece of product. The process typically begins with applying pressure to some powdered materials or materials having fine sizes. The pressure helps draw particles close together. The materials are then heated in the

Table 4.5 Virgin Steel Production

Process	Pollution	Remarks
I) Raw Materials Mining and Processing		
A. Iron Ore Mining and Processing		
i). Open-pit mining of iron ore	- overburden as solid waste	not quantified
	- air emission from mining and ore handling equipment	not quantified
ii). Beneficiation		
- wash and screen ore to remove impurities	- emissions from concentrate production	quantified
- crush and grind		
- magnetic separation		
- produce iron ore concentrate by floatation cells	- residue disposed as slurry	not quantified
iii). Pelletization		
- concentrate is pelletized with a binder	- particulate emissions	quantified
- harden in a furnace		
B. Limestone Quarrying and Processing		
i). Limestone quarried by the open-pit method	- overburden as solid waste	not quantified
ii). Crush	- particulate emission during during processing	quantified
iii). Screen		
a. obtain usable limestone or	- emissions from kiln: particulates, SO_x, NO_x, VOCs and CO	quantified
b. limestone to a rotary kiln for converting to lime		
C. Coal Mining and Processing (Coal is used as an input for coke production)		
i). mined by either surface or underground mining	- pollutants from drilling, blasting, and from coal hauling vehicles	quantified
ii).clean at a preparation plant to lower its ash and sulfur content (may use classifiers, screens, centrifuge, etc.)	- releases from coal cleaning	not quantified

Process	Pollution	Remarks
II) Coke Manufacturing		
(Coke is used as fuel and an O_2 reducing agent) i). Coal is heated in a series of ovens ii). Pyrolysis removes the volatile fraction - coke is the main product - collect volatile fraction, other gases and vapors to recover useful chemicals that will go back to ovens or other parts of the steel plant iii). Coke is quenched	- Emissions throughout the process (air and water)	quantified
III) Sintering (Optional)		
i). Recycle small iron ore fines, fine coke, limestone, etc. by sintering them into an agglomerated piece	- water and air emissions	quantified
ii). Sinter is crushed and screened - acceptable particles go back to blast furnace - undersized is sintered again		
IV) Pig Iron Production		
i). Processed Iron Ore (from IA), lime (from IB) and coke (from II) are charged to a blast furnace		
a) close to furnace bottom: highest temperature; molten iron pool is called pig iron	- pollutants from blast furnace	quantified
b) above (a): lime, coke and ore impurities are formed into slag	- blast furnace sludge and slag as solid waste	not quantified
c) furnace top: oxygen removed from ore by coke gases		

Process	Pollution	Remarks
V) Steel Making (by Basic Oxygen Furnace)	- air and water emission	quantified
i). BOF furnace is tilted		
ii). Steel scrap is added		
iii). Then add molten pig iron		
iv). Return BOF to the upright position		
v). Oxygen is supplied into the furnace to oxidize impurities and provide heat	- largest amount of pollutants released at this stage (iron oxide, fluoride, particulates and carbon monoxide)	quantified
vi). Add lime, and fluorspar to form a slag layer with impurities		
vii). Molten steel is removed		
viii). Add alloying elements		
ix). Form steel into ingots or cast		

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furnace. Particles coalesce by diffusion to form into a dense mass.

Pig iron is the name given to the molten iron formed during the pig iron production. It is the major input to the steel making process in the BOF.

Recycled Steel Cans

A tin coating is electroplated on steel cans to protect the steel from corrosion. When steel cans are recycled, the tin coating is usually removed prior to infeed to the furnace. Tin, if charged with steel to the furnace, will make the steel become brittle. Most steel product specifications limit a tin level of 0.02% to 0.04%.

Detinning is therefore the major process in steel recycling. The two methods of detinning, chemical, and electrolytic, are outlined in Figure 4.7. Detinned steel is then mixed with other raw materials to produce new steel.

Environmental Impacts

19.6 MMBtu/ton is required to produce virgin steel and 16.9 MMBtu/ton is needed for recycled steel production. The energy input for virgin steel production is dominated by pig iron production, which accounts for 79% of the total process energy. The reason that there is only a decrease of 2.7 MMBtu/ton in energy use when recycled instead of virgin steel is produced is because of the low percentage of used steel cans that can be put into the BOFs. Technically, 20-30% of steel scrap is required for any steel production in the BOFs. Tellus assumes that 28% steel scrap is used in virgin steel production. When recycled production is considered, an extra 12% of steel can scraps are added to make up the maximum allowable percentage of 40% that can be put in this type of furnace. As

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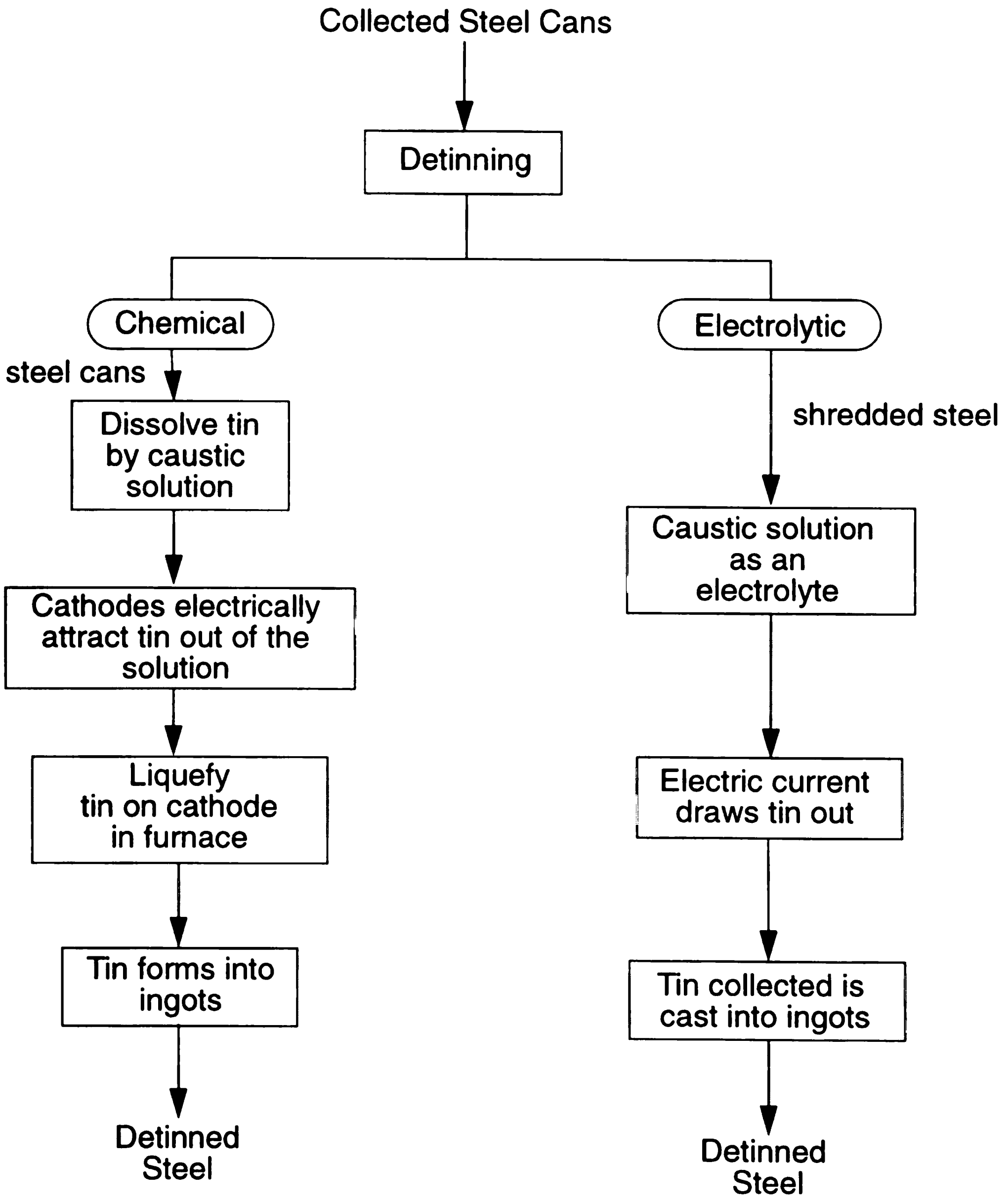
Figure 4.7 Steel Can Detinning



Pollutants Included:

Only air pollution is involved

a result, the 12% recycled material does not have a significant impact on the energy or environmental cost reduction. The environmental releases examined are either equal to or only slightly decreased when post-consumer steel cans are added as scrap. The fact that detinning causes air emissions is responsible for the improvement being small as well.

PLASTICS

In 1993, 17.3 billion pounds of resins were used for packaging in the United States, which accounts for 24.6% of the U.S. resin sales in that year. Of this amount, HDPE was the most used plastic, taking 31.7%, followed by 31.4% from LDPE (including LLDPE and EVA), 10.5% from PP, 10.3% from PS, 10.2% from PET, 2.85% from PVC and 3.1% others. Among these resins used in packaging, PET achieved the highest recycling rate, 25.6%. HDPE was recycled at 8.9%. Other post-consumer resins were recycled in minuscule amounts (Modern Plastics, 1994).

The Tellus Institute has examined the production impacts of all the above resins mentioned, but only PET and HDPE were studied for their impacts in the disposal system since they were and still are the only two post-consumer plastics being recycled at significant percentages. While the rating scores of this project are based on both the production and the disposal impacts, only PET and HDPE exhibit sufficient information to be considered for the ratings. In turn, only the production processes of these two plastics are described in this section.

Table 4.6 First Three Steps of Virgin Plastics Production

Process	Pollution	Remarks
1).Natural Gas and Crude Oil Extraction (only onshore sources were studied) i) Develop oil/gas wells - drill wells for extraction	- drilling waste (mostly disposed on-site)	quantified uncontrolled emissions only (q.u.e.o)
	- wastewater from drilling pits (mostly untreated before disposal)	q.u.e.o
	- air pollutants from diesel engine used for drilling	q.u.e.o
ii) Production - oil and gas are brought to surface - crude oil and natural gas are separated - water is removed from oil and gas - crude oil is transferred to a refinery - natural gas is transferred to a gas processor	- produced water	q.u.e.o
	- air pollutants from generators that motivate electric pumps and other equipment	q.u.e.o
	- associated wastes such as tank bottoms, sludges, oily debris, etc.	not quantified
iii) Well Closure		

Process	Pollution	Remarks
2). Natural Gas Processing & Petroleum Refining		
i) Natural gas processing - ethane and propane are the two constituents to be obtained for plastic feedstocks	- air pollutants (SO_x emissions originated from HS present in natural gas)	q.u.e.o
a. remove water and hydrogen condensate (recyclable)		
b. removal of non-hydrocarbon gases such as hydrogen sulfide		
c. liqufication - major constituent (CH_4) is removed, other gases are compressed and liquefied		
d. distillation (or fractionation) - different hydrocarbons are separated according to their boiling points		
ii) Petroleum refining - propane, liquefied petroleum gas (LPG), naphtha and gas oil are to be obtained for plastic feedstocks		
a. desalting - dissolved salt in crude oil is removed to prevent corrosion and poisoning of catalysts used in later petroleum refining processes	- water effluent	controlled emissions quantified

Process	Pollution	Remarks
b. distillation - separate lighter hydrocarbons such as gasoline, naphtha and light gas oil out for use (by atmospheric distillation)	- wastewaters (pollutants include sulfides, ammonia, oil, cyanides and phenols)	quantified
	- air pollutant (VOCs)	quantified
- heaviest hydrocarbons (gas oil and residuals) distilled by vacuum distillation to obtain valuable components		
c. catalytic cracking - gas oil from both distillation methods cracked in a catalytic cracker to separate hydrocarbons into gases, naphtha, gasoline, and light and heavy gas oil	- air pollutants include TSP, PM_{10} SO_x, NO_x, VOCs, CO, Pb, NH_4, aldehydes and benzo(a)pyrene	quantified
d. gases obtained from b and c are processed into refinery gas and LPG		
	- additional air pollutant sources:	
	i). from blowdown systems that are used to dispose waste or excess gas; and	quantified
	ii). from fugitive emissions due to accidents, insufficient maintenance or poor planning	quantified

Process	Pollution	Remarks
3). Organic Chemicals Production i) Products from natural gas processing and oil refining (ethane, propane, LPG, naphtha, and gas oil) are cracked into organic constituents of benzene (for PS), butene, ethylene (for PE, PS, PET & PVC), methane, propylene (for PP), toluene, and xylenes or paraxylenes (for PET)	- ethylene production:	
	i). benzene and VOCs are the major air pollutants	q.u.e.o for VOCs
	ii). water effluent from thermal cracking	unquantified
	- propylene production:	
	i). VOCs	q.u.e.o
	ii). water effluent from thermal cracking	unquantified

Production of Virgin Plastics

Plastics are derived from crude oil and natural gas. Different types of plastics are manufactured from different kinds of hydrocarbons contained in the crude oil and natural gas. Beginning from crude oil and natural gas extraction, there are a few production steps that are generic to all plastic types, including:

1). natural gas and crude oil extraction,

2). natural gas processing and petroleum refining, and

3). organic chemicals production.

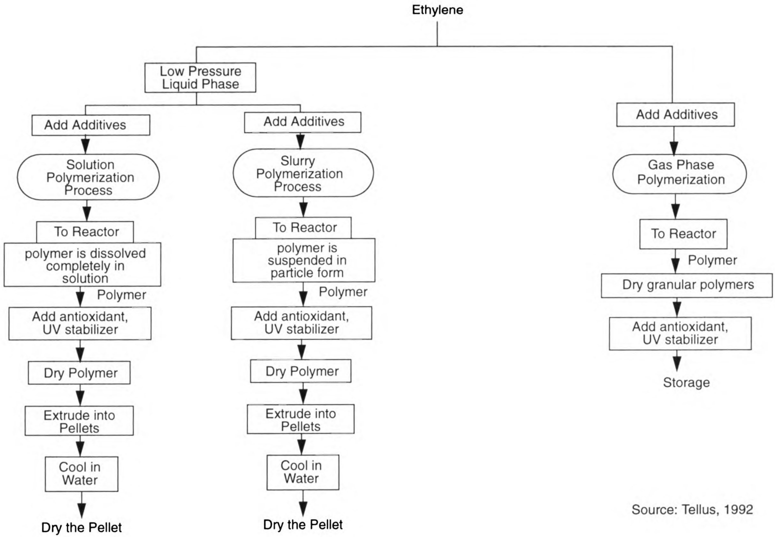
A summary of these three steps is outlined in Table 4.6.

Virgin HDPE Production

Both HDPE and LLDPE are produced from ethylene obtained from the organic chemicals production described in Figure 4.8. Both polymers are produced in low pressure reactors. The production can be achieved by three different processes: solution polymerization process, slurry polymerization process or gas polymerization process. It is assumed that 74% of HDPE is produced from slurry polymerization, 16% from solution polymerization, and 10% from gas phase polymerization. An overview of these production processes is outlined in Figure 4.8.

Environmental Impacts

Of the 39.45 MMBtu/ton of process energy used in HDPE production, 29% is used in ethylene production, 33% is spent on polymerization and 38% is consumed during the early stages of the production. The environmental cost of producing virgin HDPE is dominated by sulphur oxides emission ($123.27/ton



Source: Tellus, 1992

Figure 4.8 Production of High Density Polyethylene

HDPE) and naphthalene ($103.04/ton HDPE). Sulfur oxides are released from the use of natural gas as a feedstock and the burning of residual oil as an energy source. Naphthalene is emitted as air pollutants from petroleum refining and during monomer production. Pollutants from waste water is also quantified into the environmental production cost.

Virgin PET Production

PET is produced by polycondensation of ethylene glycol and dimethyl terephthalate (DMT) or terephthalic acid (TPA). Since no data were found on the energy and material flow for TPA, Tellus modelled PET production based on the use of DMT. The production processes are outlined in Figure 4.9.

The major air pollutant released during virgin PET production is VOCs. VOCs are emitted from ethylene oxide, ethylene glycol and DMT production. Water effluents are also generated during various stages of the PET production.

Environmental Impacts

54.37 MMBtu is required to produce one ton of PET. This amount of energy is split into 38% consumed by the formation of paraxylenes, 30% used for producing ethylene glycol, and the remaining 32% used for the condensation polymerization process.

The environmental cost of producing PET is shown in Table 3.4. This cost is dominated by the release of antimony ($384.16/ton PET), followed by sulfur oxides ($190.19/ton PET) and naphthalene ($154.04/ton PET). Antimony-based catalysts are commonly used in the U.S. in the polycondensation process for bottle-grade PET. The resulting high cost is due to the high cost that is charged for

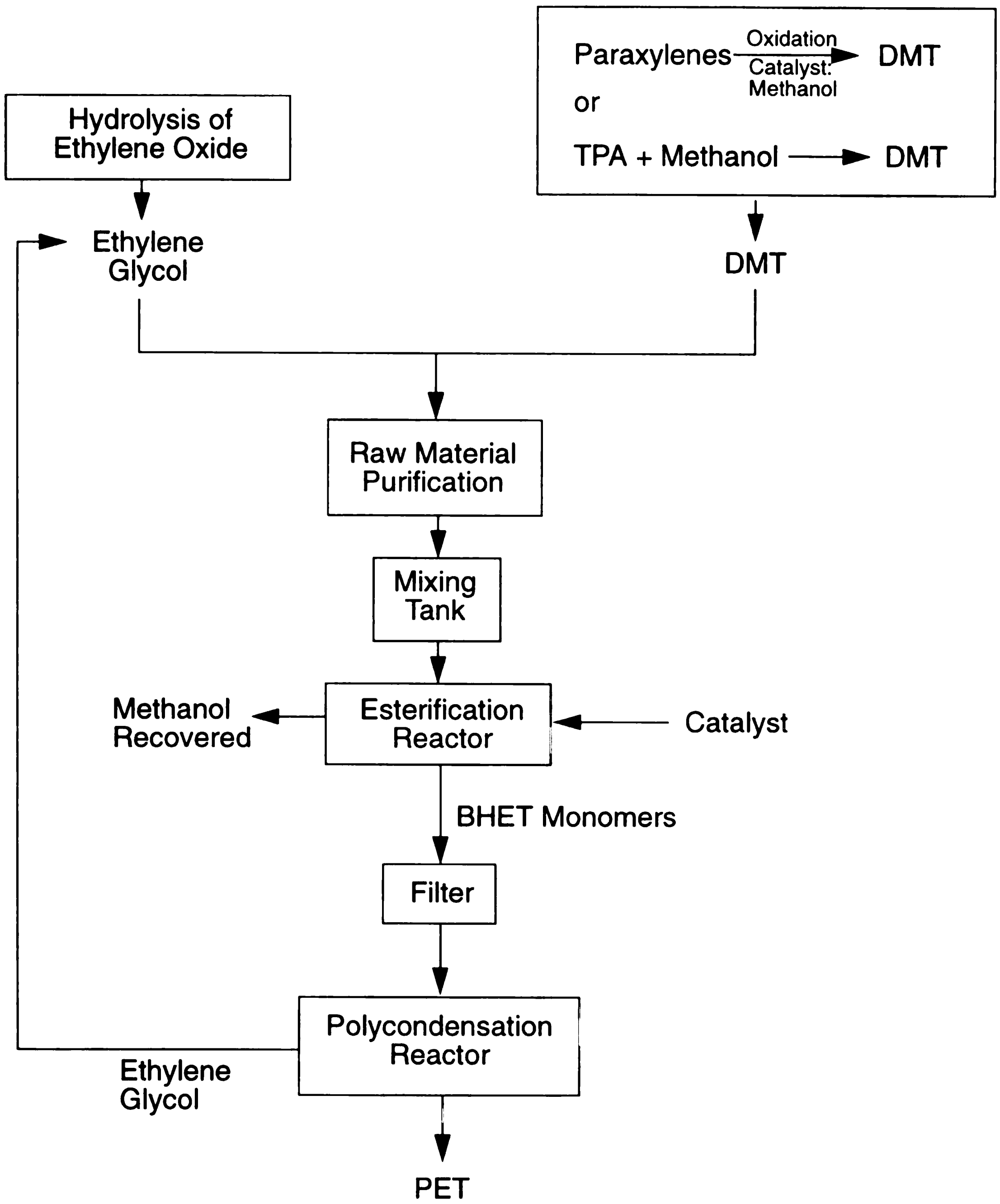


Figure 4.9 Production of Virgin PET

the release of antimony ($5600/lb of antimony).

Plastics Recycling

In general, plastics can be recycled by primary, secondary or tertiary recycling. Primary recycling usually refers to the reuse of in-house scraps from the production processes. Unlike other packaging materials, post-consumer plastics are seldom suitable for primary recycling because of their degradation and contaminant level. Instead, their second lives begin with secondary recycling, in which plastics are physically reprocessed by grinding, washing, pelletizing or flaking, and then are remelted to form new products (Armstrong and Thorsheim, 1993). PET and HDPE are the two most recycled postconsumer plastics. Recycled PET is most used to produce fibers for products such as ski jackets, carpets and sleeping bags. Other usages include household-industrial-chemical (HIC) containers, strapping, and car bumpers. Secondary recycled PET has very limited use in food-contact containers. One such use that is favorably reviewed by the FDA is the making of quart and pint-size baskets for fresh fruit and vegetables (Armstrong and Thorsheim, 1993). By applying methanolysis, a method of feedstock recycling, Coca-Cola is able to implement up to 25% of recycled material into 2-liter Coke bottles with approval from the FDA and comparable authorities in the European Community (Layman, 1993). Recently, a 100% recycled PET bottle is developed. The technique involved is a physical process which 'super-cleans' the plastic for reuse in the production.This process also receives no objection from the FDA. Recycled HDPE finds its use mostly in irrigation pipes, HIC containers, crates, cases and pallets. Its involvement with food containers is the base cups that support beverage bottles. Its use to fabricate harvesting crates is also favorably reviewed by the FDA (Armstrong and Thorsheim, 1993).

Besides recycling single-type resins, postconsumer plastics can also be recycled commingled. The E.T.1 extruder manufactured by Advanced Recycling Technology SA of Belgium is a type of commercially available machine that specializes in handling commingled plastics (Barlaz et al., 1993). Products made from this type of recycled plastics are usually of simple shapes. Plastic lumber is one major product. The cost to produce this lumber is, however, higher than that of wooden lumber, therefore some analysts predict that the market for commingled molded products will not improve dramatically in the short term (Barlaz, 1993). Other low performance products made from mixed plastics are trash cans, corrugated pipes and flowerpots.

To recycle postconsumer plastics, the material is first shredded and ground. Usually, only clear plastics are accepted while mixed colors are trashed. The plastics are reduced to a size of about 1/4 inch. These plastic chips are then sent to the wash line where labels, adhesives and other foreign fines are removed by washing the chips in cold and then hot baths with detergent. After repeated washing and rinsing, the cleaned chips are sent to the hydrocyclones in which resins of different specific gravities are separated for single-resin recycling. PET has a specific gravity of 1.2 and that of HDPE is 0.96. If aluminum is present as a contaminant, an electrostatic separator may be used to separate the plastics from aluminum (Previd, 1994).

Recycled plastics are identified as having less superior properties than virgin plastics. Recycled plastics are more susceptible to environmental stress cracking which may lead to bottle leakage. In addition, a study demonstrated that recycled PET exhibits a lower viscosity, lower molecular weight and higher carboxylic end group concentration than virgin PET. The extent of those effects depends on the purity of the recycled PET. PVC and impurities such as adhesives are causes of the degradative effects. There are methods to reduce some of the degradative effects such as adding anti-oxidants. New "hot melt" adhesives are chemically inert and thermally stable and therefore can prevent chemical degradation of the recycled resin. For mixed plastics, compatilizers can be added to enhance the blending of different polymer resins to increase the resulting strength (Giannotta et al., 1994).

Tertiary recycling of plastics, also commonly called feedstock recycling, is the depolymerization of polymers into monomers or oligomers followed by the purification and then regeneration of new polymers. Cleaning is more thorough in this type of recycling. Therefore the final product can be made into food containers without worries of hygienic or structural problems. Tertiary recycling of single plastics has been developed for years. The new interest is in its recycling of mixed plastics. Testing and pilot operations are carried out in Europe for this interest: Veba's coal liquefaction technology to hydrogenerate the plastics to a syncrude for a starting material for the petroleum industry, and the gasification of plastics by ESPAG to produce methanol and ammonia as well as heating fuel. Also, BP and Shell have patents on their feedstock recycling technologies on mixed plastic waste. A few other companies are also experimenting with this new recycling challenge and have plans to commercialize their technologies. The major hindrance to tertiary recycling is its high cost of operation compared to other types of plastics recycling (Layman, 1993).

The plastics recycling industry as a whole is still an early stage of development because the operation is hampered by the high cost of collecting and transporting the postconsumer waste due to its light weight and bulkiness. The difficulties of sorting very similar-looking plastics, the problem of contamination, and the degradation effect due to repetitive processing also have to be overcome. Since the industry is still working on those problems, little data are available for plastics recycling. As a result, the Tellus Institute did not describe or evaluate the production of recycled plastics.

Nevertheless, secondary recycling is expected to save energy by avoiding the production of monomers and polymerization. Secondary recycling consumes energy in sorting, grinding, cleaning and melting only. If a method is developed to reduce the conventional and environmental costs of transporting plastic wastes, secondary recycling of plastics should have a significant benefit to the environment.

In the paper, “Are Plastics Really the Landfill Problem” (Lantos, 1990), the author estimated the recycle potential of various kinds of plastics (Table 4.7). These estimates are based on the applications of each plastic and the length of its service lifetime. It is evident that plenty of plastics such as PET, LDPE, HDPE and PS have a high potential for recycling. In other words, it is now up to the technology and the infrastructure to catch up with such attractive potentials.

Table 4.7 Recycle Potential of Plastics

Plastic Type	Recycle Potential (% of Plastic)	Plastic Type	Recycle Potential (% of Plastic)
PET	98	Cellulosic	22
LDPE	78	PVC	20
HDPE	66	Polycarbonate	15
Polystyrene	61	ABS	14
Polypropylene	40	Acrylic	10
Nylon	27	Polyacetyl	10
		PBT/PET	10

Source: Lantos, 1990

CHAPTER V

ENERGY COST AND NATURAL RESOURCES DEPLETION COST

In addition to the environmental cost of production, the energy cost and the cost associated with the depletion of exhaustible natural resources are also derived from packaging material production and are costs that are included in the ratings.

ENERGY COST OF PRODUCTION

Energy consumption depletes exhaustible natural resources including coal, natural gas and petroleum; it is thus a cost to the environment and is therefore incorporated into the ratings. The amount of energy consumption is converted to a monetary cost so that it can be aggregated with other environmental costs used for the ratings.

To determine the energy cost of production for each material, the process energy from raw material extraction to the production of that material is first determined. In the Tellus Packaging Study, the energy inputs for packaging production are listed, but those values include the intrinsic energy values of the raw materials. Since most of those energies are not responsible for the production process, they are taken out of Tellus' figures. Only the amount of process energy required for production is counted.

Once the amount of process energy for each material's production is determined, it is multiplied by a unit price for energy. The U.S. energy price (dollar/ million Btu) charged to the industrial sector in 1991 is used. This price, $5.34/ million Btu, is a weighted average of the unit prices for electricity, petroleum, natural gas and coal consumed by the industry sector in 1991 (EIA, 1992).

Table 5.1 lists the process energies and the corresponding energy costs for virgin and recycled production. Recycled production appears to consume less energy than virgin production.

Table 5.1 Energy Costs of Packaging Production

Material	Energy Required (x10^6BTU/ton)	Energy Cost ($/ton)
Virgin Boxboard	29.96	159.99
Recycled Boxboard	22.89	122.23
Virgin Corrugated Box	24.91	133.04
Recycled Corrugated	22.44	119.81
Virgin Aluminum	208.00	1110.72
Recycled Aluminum	8.32	44.43
Virgin Glass	13.50	72.09
Recycled Glass	9.92	52.97
Virgin Steel	9.99	53.35
Recycled Steel (40%)	8.90	47.53
Virgin HDPE	39.45	210.66
Virgin PET	54.37	290.34

COST OF NATURAL RESOURCE DEPLETION

The underlying reason for our society to have concerns for the depletion of certain raw materials, or the deterioration of certain environmental qualities is because we have needs for them. If a mineral is in scarce supply but it is of no use to human kind, an evaluation of its depletion will be unneccessary. Similarly, the study of the environmental impacts of various solid waste management in this project is actually the study of the environmental impacts that ultimately affect human lives. The quantification of the depletion of a raw material alone is therefore not sufficient to include it as an environmental impact. Its demand by the society should also be taken into account. In other words, the social cost of a raw material should be used to characterize its depletion in relation to its demand. A raw material having a high social cost means it is in high demand and its depletion is of great concern to the environment and in turn to our society. This social cost can be reflected by the market price of the raw material. Strictly speaking, the economic price, which is the market price minus taxes, subsidies and any other forms of market distortions, represents the true social cost. However, since those distortions are difficult to determine, the market price is a good substitute for the social cost.

The depletion of virgin materials due to packaging production is represented by the market prices of the exhaustible raw material inputs. Table 5.2 presents a list of prices for those major raw material inputs used for producing the seven kinds of packaging materials discussed in this thesis. Table 5.3 reports on the social costs of raw material depletion for each ton of packaging material production. The derivation of these costs can be found in the Appendix.

Table 5.2 Unit Costs of Raw Materials

Raw Material	Price/ton (unless stated otherwise)	Remarks
Bauxite	$17	From 1.
Feldspar	$36.93	From 1.
Fluorspar	$190-195	From 1. Illinois, bulk, acid grade.
Limestone	$ 4.30	From 2.
Sand	$13.65	From 1.
Salt (for Caustic Soda)	$ 3.95	From 1.
Soda Ash	$ 89.21	From 1.
Anthracite	$ 34.24	From 3.
Natural Gas	$ 1.49/1000 ft^3	From 4. Wellhead price.
Crude Oil	$ 0.048/lb	Derived from 4. Domestic first purchase price. See Appendix for conversion.
Iron Ore	$ 31.18	From 1.
Concentrates	$20.09	From 1. Only import price available.
Roundwood and Chips	$ 22.95	From 5. An average price for softwood and hardwood and chips. See App.

[1] United States Department of The Interior, Bureau of Mines, Mineral Yearbook Vol. I. Metals and Minerals 1992

[2] United States Department of The Interior, Bureau of Mines, Mineral Yearbook Vol. I. Metals and Minerals 1991

[3] Energy Information Administration, Coal Production 1992, Oct. 1993

[4] Energy Information Administration, Monthly Energy Review May-Aug 1992

[5] Michael Howell, United States Department of Agriculture, Forest Service, Pulpwood Prices in the Southeast, 1992

Table 5.3 Social Costs of Raw Material Depletion

	$/ton	
Material	Virgin	Recycled
Aluminum	$128.36	0
Steel	$39.29	$33.15 (40% recycled content)
Glass	$29.53	0
Boxboard	$72.50	0
Corrugated	$61.72	0
HDPE	$88.91	---
PET	$101.28	---

CHAPTER VI

SOLID WASTE MANAGEMENT TECHNOLOGIES AND DATA

SOURCE REDUCTION

As stated by EPA, source reduction is the "reduction in the generation of waste, by redesigning products or by changing societal patterns of consumption and waste generation" (EPA, 1975). Source reduction aims at reducing not only the amount but also the toxicity of the material entering the solid waste stream. Applying it to packaging, source reduction includes reducing the amount and toxicity of materials used in a package, increasing the permanence of a package, reusing packages and buying fewer material goods. In this thesis, since source reduction is rated separately from reuse, package reuse is taken out from the list of approaches to source reduction and will be discussed in a separate section.

To reduce the amount of materials used in a package is always the goal of product manufacturers. With the given functions that they require their package to perform, they aim at delivering those functions with the lowest cost by using the least amount of materials. Source reduction, to product manufacturers, is thus driven by cost reduction. Table 6.1 illustrates that packaging material cost is expensive compared to landfill cost.

Table 6.1 Comparison Between Packaging Materials Costs and Landfill Cost

Landfill Cost	$10 to $140/Ton National Average $26.50/Ton
Packaging Material Cost to Food Company	
Foil	$6000/Ton
Paper Cartons	$800 to $1800/Ton
Plastic Wrap	$2500 to $4000/Ton
Plastic Bottles and Cups	$1400 to $4000/Ton
Glass Bottles	$200 to $400/Ton

Source: Erwin and Healy, 1990

The cost savings of using one less ton of any packaging material is far beyond the per ton avoided landfill cost. It is therefore obvious that the manufacturers have a strong incentive for waste reduction at its source.

The best savings seen by manufacturers, however, may not be agreed by others. In order to make the product sell itself, it has to stand out from other similar products. Manufacturers may produce a package with a sophisticated and glamorous surface design which uses much ink in the printing process. They may attract consumers by producing convenient packages, such as individual packs, which use more material. Such design criteria, while considered as essential by the manufacturers, may be regarded as excessive by environmentally concerned consumers. It is difficult to draw the line between the two and it is an area from where source reduction of packaging brings up controversies.

Government's involvement in source reduction includes implementing bans on certain types of packages and the control of toxic substances in packaging. To cite a few examples, in 1989, Maine enacted a ban on multi-material aseptic beverage packaging (Levy, 1993). California enacted a law, in effect on Jan 1, 1995, that bans rigid plastic containers if they do not reach any of the specified rates of recycling, reusability or source reduction (Levy, 1993). Eleven states had laws banning or restricting the intentional addition of lead, cadmium, mercury or hexavalent chromium in inks, dyes, pigments, adhesives and other components of packaging (Fishbein and Gelb, 1992). This number has increased now (Selke, 1995). There are also source reduction schemes which include government assistance programs that provide grants and technical assistance.

Government's regulatory actions and initiatives on source reduction are felt mostly by manufacturers. On the consumer side, source reduction is not heard as often as recycling or other methods of disposal. Asking consumers to produce less waste means requiring them to alter their spending habits and to change their life-styles. On the other hand, recycling requires consumers to alter only their disposal habits after the waste is produced. Therefore, from the consumers' standpoint, it is easier to perform recycling than source reduction. Local solid waste managers find it difficult to plan for source reduction for their communities because, while they know about acquiring more collection trucks, establishing MRFs, building incinerators and landfills, for carrying out recycling, incineration and landfilling, they are not sure how to carry out the less tangible event of source reduction, which is associated with not producing, not collecting and not building. As a result, a lot of communities establish so called waste reduction programs, which in fact include both source reduction and recycling. By combining the two solid waste management options together, solid waste planners can save the trouble of having to specifically identify the steps for implementing source reduction. The consequence is that recycling always becomes the center of the attention in these waste reduction programs. Since source reduction is the most highly ranked solid waste management approach, the government should put more emphasis on establishing well-planned strategies in promoting and implementing it separately.

The result of source reduction is that less waste is present in the environment; hence no hazard, health or environmental, is created relating to the avoided waste. The implementation of source reduction therefore carries no external cost to the environment. According to the rating scheme, source reduction should then receive a score of 1. This score places source reduction on the upper bound of the rating scale for disposal options. Any other disposal options will be compared with source reduction and be assigned a score that is less than 1 if they are identified to cause negative environmental impacts.

LANDFILLING

In brief, landfilling is the management of garbage by burying it beneath soil. In detail, landfilling, from its planning phase, to its post-closure, requires the following of strict rules and regulations because if performed inappropriately, it can induce environmental problems.

Landfill Technology

a). Site Location

Landfills should be located in a stable area, above ground with a safe distance to the natural groundwater surface. To site a landfill, the climate, hydrology, hydrogeology, soil characteristics and other site-specific conditions have to be evaluated (Christensen, 1987). Those are important factors that determine whether the landfill would induce negative environmental impacts.

b). Landfill Configurations

Three methods of landfilling are presently in use: the area method, the ramp method and the trench method (O'Leary and Walsh, 1991). The site contours determine the method used. The area method is used to fill an excavated area; the ramp method is used on a sloping site, and the trench method is used on a flat or gently sloping site with successive excavated trenches.

c). Compaction and Degradation

Each day, waste delivered to the landfill is compacted. At the end of the day, an intermediate cover of earthen material is placed on top of the waste to prevent odors and vector attack. As-delivered waste has a bulk density of 450 - 600 lb/cubic yard. Compaction equipment increases its density to 800 -1400 lb/cy so as to enhance the settlement of the landfill (O'Leary and Walsh, 1991). More compaction will be achieved as more waste and soil are added to the landfill and when the organic portion of the waste decomposes over time.

There are four stages in the degradation of organic waste. The first stage is carried out by aerobic bacteria when there is still oxygen in the landfill. Carbon dioxide, water and nitrate are the decomposition products. As oxygen is depleted, facultative and anaerobic microorganisms take over. This second stage is characterized by high volatile acid production, a low pH of 4-5, high conductivity, high chemical oxygen demand (COD) and low methane production. In the third stage, methane-producing bacteria predominate. They reduce volatile acids to methane and carbon dioxide, resulting in a methane to carbon dioxide ratio of approximately one to one, and causing a rise in the pH and decrease in COD and conductivity. This stage may take years to complete. The final stage marks the stabilization of the landfill. When degradation is completed, the landfill may have a 15% reduction in its final depth. Poor initial compaction can cause a reduction of as much as 25% (O'Leary and Walsh, 1991).

d).Leachate Collection and Liner Systems

Leachate is generated by precipitation or groundwater percolating through the surface cap or the landfill mass and by the fluid contained in the MSW (NREL, Vol VIII, 1992). The placement of a bottom liner and a low-permeability surface cover can greatly reduce the amount of leachate produced, but total elimination cannot be achieved because, as time goes by, physical or mechanical degradation of liners and surface cover due to freeze-thaw and wet-dry cycles or from subsidence of waste give a way for infiltration to take place. A leachate collection system is now a mandatory requirement in all operating landfills. This system prevents leachates from entering into the groundwater, which is the source of drinking water for more than half of all Americans (Cadwallader, 1988). Leachate collected can be discharged to a sewer with or without pretreatment, or to land or a surface water body after treatment. Leachate can also be recirculated into landfills. Landfills usually take twenty to thirty years to complete decomposition. Findings indicate that by recirculating leachate into landfills, the biological breakdown of organic matter in the landfills can occur up to ten times faster than usual. It is due to the fact that moisture enhances decomposition through supporting the survival of bacteria. Currently, twenty landfills with leachate recirculation, three of which being supported by EPA, are under study (Lipkin, 1994).

When designing a liner system to prevent the external migration of leachate, the following criteria should be met:

i).Low permeability: the liner material and the system should have the ability to resist the seepage forces of leachate generated within the landfill (Christensen, 1987).

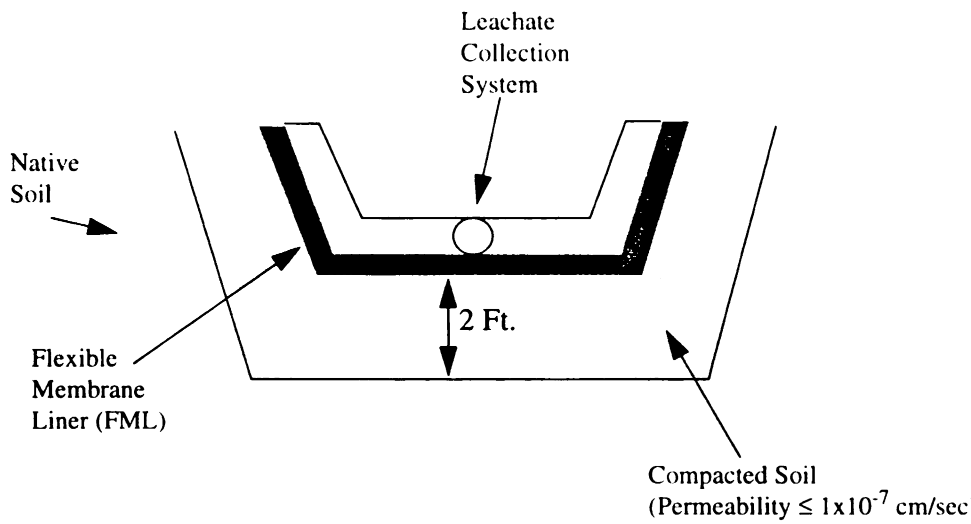
ii). Damage resistance: during the construction of the liner system and later, the filling of the landfill, damage can be induced to the liners. This may occur because of the unloading of wastes, waste compaction, and displacement due to settling. Liner materials, therefore, should be able to sustain those potential mechanical attacks. Clay liners are more resistant than synthetic liners in this requirement.

iii). Resistance to Chemical Attack: because of the complex composition of MSW, it is very important for landfill liners to have good chemical resistivity. There is, however, no one liner that is chemically resistant to all components of MSW. Therefore, landfill liners do not provide a completely safe system.

In addition, an effective liner system must offer the above properties at high and low temperatures as well as maintain a long-term performance.

Figure 6.1 provides a schematic drawing for the composite liner system required by the EPA:

Compacted soil has widely been used as a liner material. It is mixed with clay and yields a permeability in the order of 10^{-7} cm/sec (Christensen, 1987). The thickness of its application depends on particular site conditions, but the EPA requires a minimum of two feet to be underlain to the flexible membrane liner.



Source: NREL, Vol VIII, 1992

FIGURE 6.1 An Illustration of a Composite Liner System

FMLs are polymer liners such as HDPE, chlorinated PE or PVC. HDPE is the most commonly used material. These synthetic liners provide long term resistance to chemical attack and low permeability in the range of $1x10^{-14}$ to $1x10^{-12}$ cm/s. During the construction of a liner system, defects are usually induced in the liner which may increase the permeability to $1x10^{-10}$ to $1x10^{-7}$ cm/s. When an FML is coupled with compacted soil, the barrier performance becomes superior to a single liner system.

A leachate collection system typically consists of a drainage layer of inert material with high permeability and a series of drain pipes. The system is placed above the liner system at the base of a landfill, which is shaped either like a series of wedges or with intermittent trenches between level subsurfaces. Drainage pipes are then placed at the low points of the base to collect leachate and discharge it out of the landfill for treatment and disposal.

e). Landfill Gas

Besides leachate, landfill gas is another type of emission that has to be managed in MSWLFs. Landfill gas, typically consisting of 55% methane, 44% CO_2 and 1% trace contaminants, is produced during the anaerobic decomposition of organic waste. At a concentration of 5 to 15% in air, methane is flammable and explosive. It has caused fire and explosion on-site and off-site of landfills before. When landfill gas migrates and displaces oxygen in soil, it can ruin the plant root zone.

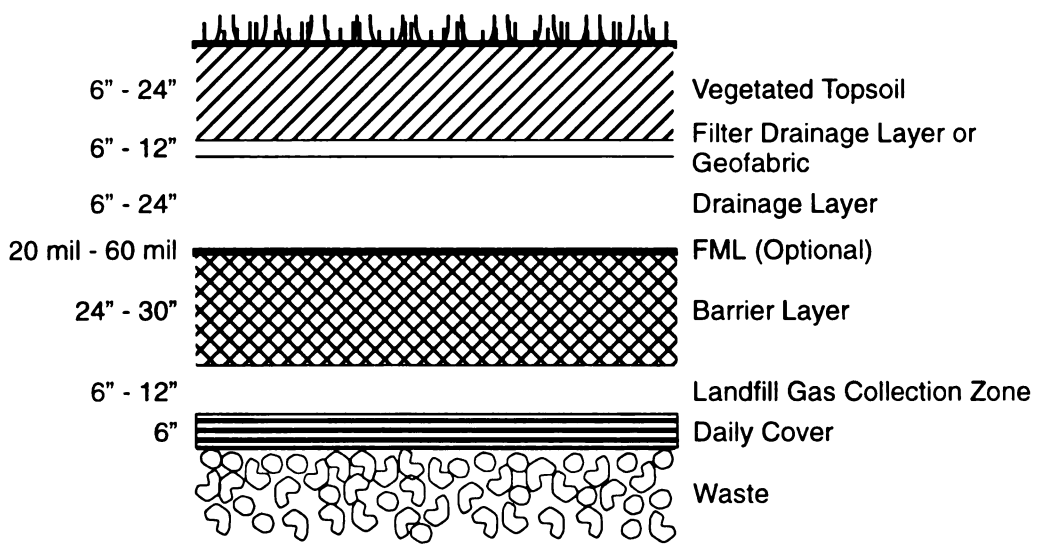
The generation rate of landfill gas ranges from 50 to 500 cubic ft/ton/year. During the lifetime of a landfill, it typically produces 6000 - 12,000 cubic feet of gas per ton of waste. It usually takes one to two years before a significant amount of methane is produced, and the generation rate will decrease as the organic waste is decomposed (NREL, Vol VIII,1992).

Passive and active systems are the two types of methods available for capturing landfill gas. A passive system generally consists of a number of gas extraction wells to vent landfill gas to the atmosphere uncontrolled. Some wells are connected to a flare which is used to consume the flammable gas collected.

An active system is the more commercially utilized method. In this system, landfill gas is moved from the extraction wells, through a header pipe to a control device by a blower. A pressure gradient is created by the blower, which prevents the infiltration of air from the surface and the sides of the landfill and enables a more efficient collection (NREL, Vol VIII, 1992). Once the gas is collected, it can go through minor processing such as the removal of water. The gas can then be utilized as fuel. At this stage, the gas has a heating value of about 550 Btu. It can also be upgraded to yield a Btu content of 900 - 1000 using techniques including solid absorption, liquid absorption and membrane separation (NREL, Vol VIII, 1992).

f). Final Cover

When a landfill is full, a final cover has to be placed on top to keep the landfill content away from precipitation, to prevent bird, insect and rodent invasion, to allow for vegetation and to keep the waste out of sight. With the above purposes in mind, capping should be equipped with the same properties as liner materials. Figure 6.2 shows the number of layers available for a landfill cap design. It has to be noted that not all layers shown are required in all landfills.



Source: NREL, Vol. VIII, 1992

FIGURE 6.2 An Example of the Number of Layers of a Landfill Cap Design

The permeability of the capping should be carefully selected because precipitation percolating from the top cover is the direct and determining source of leachate production. Surface caps should also demonstrate good physical resistance. The irregular and large movement of the underlying waste due to settlement may cause the surface cap to rupture if it does not have enough elasticity and strength.

Placing the final cap is not the final chapter of landfilling. Post-closure care continues for decades to follow. When landfill wastes complete their decomposition, perhaps twenty to thirty years from the closure date, the landfill can be mined to recover valuable materials, and landfill space can then be regained.

External Costs of Landfilling

The external costs of landfilling packaging materials extracted from the Tellus Packaging Study are listed in Table 6.2. As shown in the table, the environmental cost of production and the conventional cost of disposal are much higher than the environmental cost of landfilling. As will be shown later, the same trend is true for all other disposal options.

The low environmental cost of landfilling indicates that packaging materials do not pose a problem in landfills. Leachate and landfill gas are the two major sources of environmental cost to landfilling. Possible leachate contributed by packaging materials comes from ink and pigments or additives such as stabilizers or plasticizers, but the amount is minute. The Tellus Packaging Study reports the environmental cost of leachate being zero for all packaging materials. It is due to the fact that the concentration of emissions released by packaging materials as leachate is in the order of 10^{-11} to 10^{-7} lbs/ton of material. When

Table 6.2 External Costs for Landfilling Packaging Materials

Material	Environmental Cost of Production	Conventional Cost of Disposal	Environmental Cost of Disposal		Total External Cost
			Collection	Leachate	
Aluminum	1933	279.66	3.04	0.0	2215.7
Ferrous	230	127.86	1.20	0.0	359.06
Glass	85	85	0.28	0.0	119.37
Paper	269	269	1.10	0.0	385.07
Corrugated	214	214	1.21	0.0	330.62
HDPE	292	292	2.56	0.0	630.38
PET	854	854	2.61	0.0	1195.74

Source: Tellus, 1992

multiplying these factors with the pollutant prices, the dollar amounts are 10^{-7} to 10^{-5}, which are rounded off to zero cost. This indicates leachates originated from packaging materials pose minimal threat to the environment.

Only wood and paper based packaging waste will contribute to methane formation as they slowly degrade, but the contribution is far less than other organic materials. Tellus therefore did not evaluate landfill gas emission from packaging materials.

The cost data in Table 6.2 reflect that the lighter and bulkier the material, the higher the conventional cost of landfilling. The environmental cost of landfilling is also related to the bulkiness of materials since emissions of collection trucks are apportioned to different materials according to their occupied volume in trucks.

Taking into account the environmental cost of production, the last column of the table lists the total external cost of landfilling. These external costs will be added to the energy costs of production and cost of raw materials depletion described in Chapter V. The resulting total environmental costs will be used for the ratings.

INCINERATION

The major contribution of incineration to solid waste management is its ability to burn away as much as 90% of MSW by volume, leaving behind a much smaller amount of waste to be disposed to landfills. Two types of MSW incinerators are commonly used: mass burn and refuse-derived fuel (RDF) incinerators. In mass burn incineration, incoming waste is burnt with little or no preprocessing, whereas in RDF incineration, the waste is screened and shredded prior to feeding it into the furnace. Statistics on the number of incineration plants utilizing each of the two technologies vary from source to source. Roughly speaking, 75 - 85% of municipal waste combustors are mass burn, and 15-25% employ RDF incineration (Selke, 1990; BCEALR, 1993). Except for about 26% of the mass burn facilities, all other municipal waste combustors recover energy (NREL, Vol IV, 1992). The majority that performs energy recovery is sometimes referred to as waste-to-energy (WTE) facilities. In 1990, there were 128 WTE facilities operating in the U.S. (Kiser, 1990). Energy recovery is made possible by burning refuse that has a high fuel value. Table 6.3 compares the Btu values of some common MSW components with fuels.

Combustible materials burn readily and leave less than 10% of their volume as ash, but incombustibles have an ash content of 99%. A combustion process can be more efficient if incombustible materials are removed prior to burning. It can also reduce the amount of ash to be managed and lower certain air emissions. Preprocessing of MSW, however, requires more floor space and demands higher costs for equipment and labor.

TABLE 6.3 Btu Values of MSW Components and Fuels

Product	Btu/lb.
Aluminum, Ferrous, Glass*	50
Wood	4,700
Coal	10,500
#2 Fuel Oil	19,565
Mixed MSW	5,000
PVC	8,250
PS	17,250
PE	19,000
Magazines	6,320
Junk Mail	7,200
Corrugated Paper Boxes	7,500
Newspapers	8,040
Waxed Paper	9,250

Source: Alexander, 1993; *Tellus Institute, 1992

Mass Burn Technology

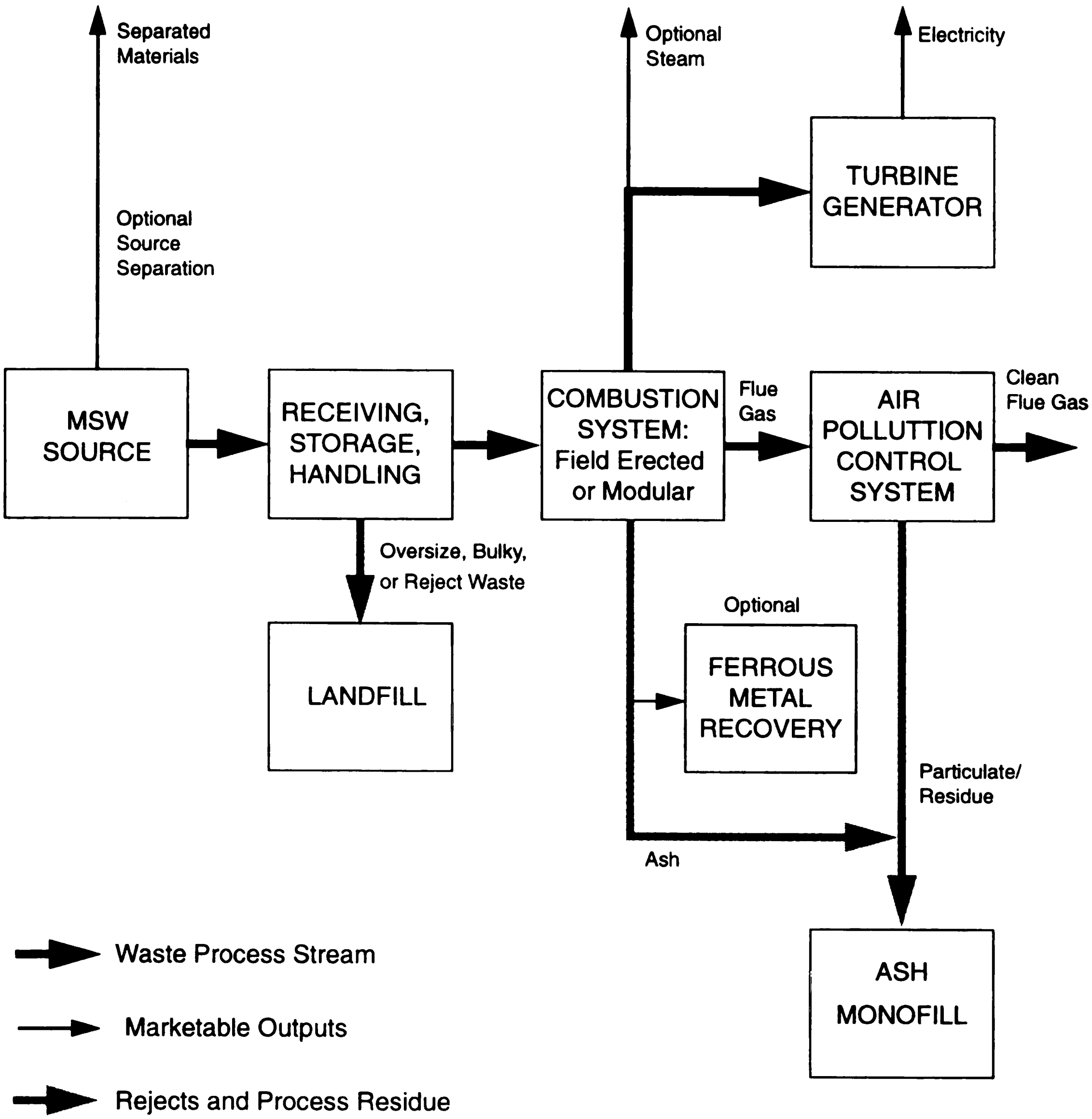
Mass burning is the predominant type of incineration technology used in the U.S. By definition, mass burning incinerates refuse without prior treatment. Only bulky and potentially hazardous materials are removed. In recent years, the trend to recycle has moved mass burning towards more preprocessing to recover valuable materials. Nowadays the distinction between mass burning and RDF technology becomes less clear.

The flow of MSW in a mass burn plant can be illustrated in Figure 6.3. As indicated in the figure, modular and field-erected combustors are the two variations of mass burn systems. Modular systems consist of small units of equipment fabricated in a shop before installation on site. The burning capacity of this type of system is small, ranging from 25 to 399 tons/day (TPD) of MSW. Field-erected systems, on the other hand, handle 200 to 3000 TPD of waste and consist of medium to large scale waterwall or refractory-lined furnaces (NREL, Vol IV, 1992).

In a field-erected mass burn plant, packer trucks unload the waste into a large pit. The waste is then picked up by a crane system and dropped to a hopper from where the waste is fed into a moving grate. The movement of the grate helps expose the refuse to enhance burning. The refuse has to pass through several sections of the grate from the drying grate, through the burning grate, and finally to the finishing grate where the bottom ash is discharged.

In a modular mass-burn plant, as-delivered waste is unloaded by packer truck onto a tipping floor. Bulky waste is removed. The remaining waste is fed into the furnace. Combustion in a modular system is performed in two chambers: the MSW is first fed to an air-deficient chamber where part of the waste is gasified. The gas is then directed to a secondary chamber where temperature is high and air is in excess amount. Here the gas undergoes additional heat recovery and organic destruction (NREL, Vol IV, 1992).

For energy recovery, the combustion area in both the field-erected and



Source: NREL, Vol I, 1992

Figure 6.3 Flow of MSW in a Mass Burn Facility modular systems consists either of a refractory-lined furnace chamber and a separate water-wall boiler located downstream, or a refractory-lined waterwall furnace and boiler system. In the latter case, the heat of combustion is transferred to steam or water in tubes surrounding the combustor, whereas the former configuration requires the heat to be transferred from the furnace to the boiler downstream. 3 to 10% efficiency is known to be lost in the latter configuration (NREL, Vol IV, 992).

Regardless of the technology used, the key to effectively and efficiently incinerate MSW is to burn the waste in an environment with optimal conditions for complete combustion. The necessary conditions involve temperature, time, turbulence and air. First, the minimum temperature for mass burn and RDF incineration is 1800^{o}F (BCEALR, 1989). Higher temperatures encourage the production of nitrogen oxides and volatilize metals. Lower temperatures increase the emissions of products of incomplete combustion (BCEALR, 1993). Second, there must be a one to two second residence time for the flue gases in the combustion zone (BCEALR, 1993). Third, the incinerator must provide adequate mixing of the waste so all parts of the waste can be exposed for even burning. Last, a sufficient amount of oxygen is needed in the combustion zone for complete combustion (Selke, 1990). Controlling the above parameters to achieve optimum MSW combustion is difficult because of the heterogeneity of the MSW composition (NREL, Vol IV, 1992). Preprocessing MSW prior to burning can reduce the variability of the waste and therefore increase the combustion efficiency. Preprocessing can also reduce furnace size and lower excess air requirements. As a result, the recovered fuel will be of higher quality.

RDF Technology

Fig 6.4 shows the layout of a RDF facility.

RDF facilities are equipped with waste processing devices similar to those in MRFs. RDF's use magnetic separators to sort out ferrous metals, air classifiers to separate the heavy from the light materials, and sometimes eddy current separators are used to retrieve aluminum. The sorted and initially shredded MSW is further shredded to meet a size requirement. The final shredded size can range from two to six inches.

RDF can be fired in suspension or cyclone boilers, partially in suspension or partly on a grate or completely on a grate. A typical RDF system converts 75-85 wt.% of the waste into RDF, and produces 10-17% ash. The Btu value of the RDF ranges from 4800-6400 Btu per pound, which is about half the Btu value of coal (NREL, Vol IV, 1992).

From the perspective of fuel recovery, RDF combustion seems to be a preferable technology to adopt. The reason that it is not as prevalent as mass burning is due to its high cost and its complex processes. Further, one problem of preparing RDF is that explosions may occur during shredding of the MSW. Methods to reduce the problem include using shredders of slower speeds, and removing explosives and hazardous materials prior to combustion. This problem has yet to be fully eliminated (NREL, 1992).

Environmental Releases

Environmental emissions from incineration plants are the major deterrent to the public's acceptance of the facilities. Heavy smoke that emerges from the plants' stacks is unpleasant enough to view, not to mention what pollutants are

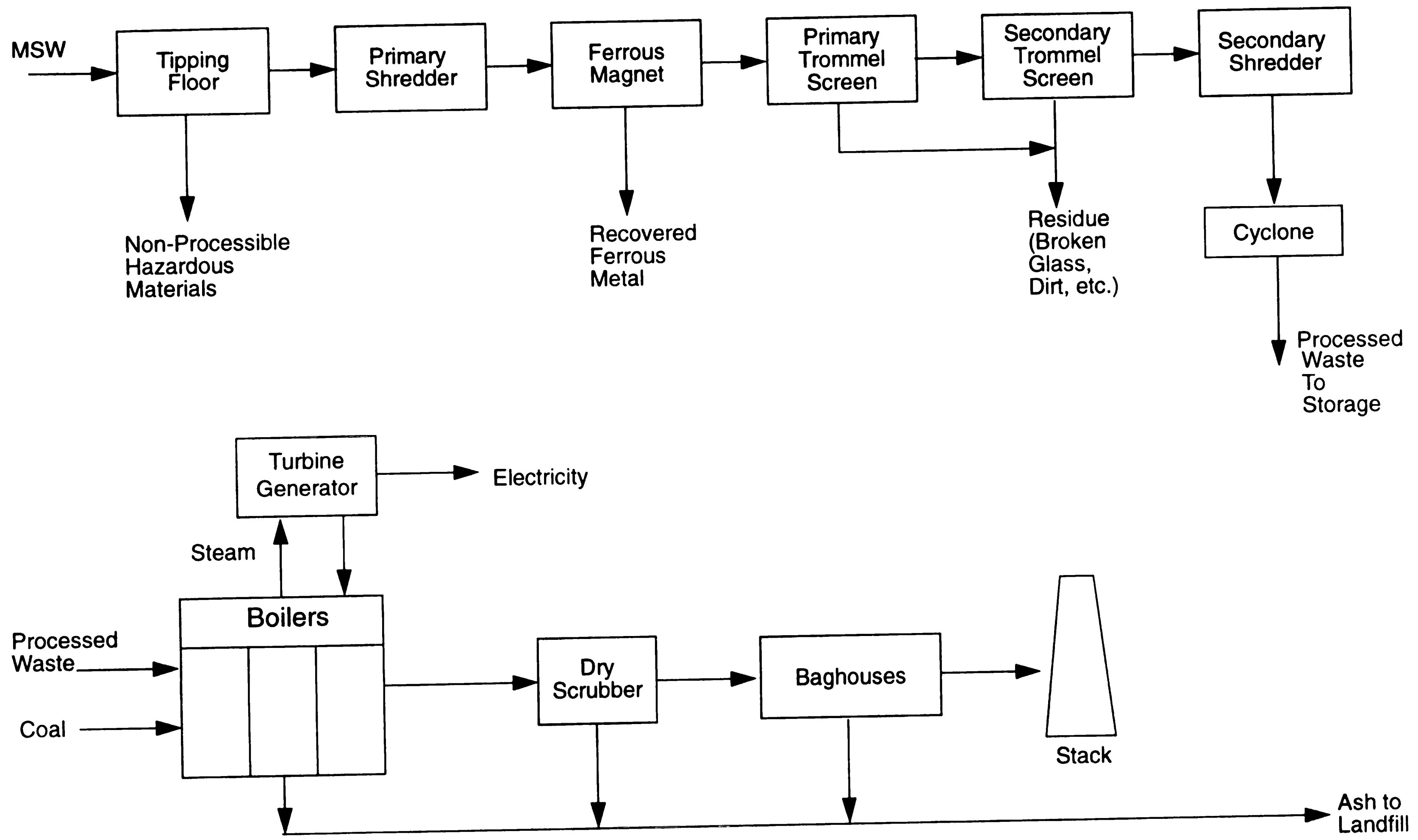


Figure 6.4 The Layout of a RDF Facility

Source: NREL, Vol. IV, 1992

actually contained in the smoke.

Air emissions include nitrogen oxides (NO_x); acid gases such as sulphur dioxide (SO_x), hydrogen fluoride (HF) and hydrogen chloride (HCl); particulate matter containing metals, dioxins, furans and condensation of acid gases as well as organic pollutants including PCBs, PAHs, hydrocarbons and VOCs. Ash residues are also of concern.

The pollutant that attracts the greatest attention is the dioxin/furan family of organic chemicals. Dioxins (PCDDs) are chemical compounds having a chemical structure $C_4H_4O_2$, and the term includes 75 chlorinated dioxins. Dioxin is also used to designate the most toxic member, 2,3,7,8-TCDD (BCEALR, 1993). The related compounds, chlorinated furans, have 135 types, also termed PCDFs in short. Sources for PCDDs and PCDFs include bleached paper, plastics, rubber, yard waste, pesticides and other chlorinated materials. Dioxins can also be formed during combustion. They can be destroyed at high temperature during combustion, but are also capable of reforming after combustion. Because dioxins condense onto fly ash, the RCRA regulation on dioxins requires a removal rate of 99.9999% by pollution control devices (BCEALR, 1993). There are studies that indicate dioxins cause cancer and some other serious diseases in test animals. The effects of human exposure to dioxins, however, have not been confirmed but remain debatable.

Particulate matters (PM) are incombustible materials such as metals, inorganic oxides and broken glass. Acid gases, volatilized metals, dioxins and organic compounds can also be condensed onto particles to form PM. Small particles (less than 10 microns) can enter the respiratory system while larger particles may transfer in food chain pathways or by direct ingestion following inhalation (BCEALR, 1993).

NO_x are the products of all conventional combustion processes, which can be formed by either fuel nitrogen oxidation or thermal nitrogen oxide formation. While the nitrogen content of MSW is about 1%, 75% to 80% NO_x emitted from burning MSW is originated from fuel nitrogen oxidation. Thermal nitrogen oxide formation refers to exposing the air inside the combustion zone to high temperatures leading to nitrogen oxide formation (Hattemer-Frey and Travis, 1991). NO_x are known contributors to smog and acid rain formation. Adjusting the combustion temperature, residence time and oxygen level in the combustion zone, and removing nitrogen containing materials such as yard and food wastes from the in-feed can reduce their formation.

Acid gases like HCl, SO_2, and HF are also produced during combustion. HCl is originated from PVC, paper products, food wastes and other chlorine-containing materials. Fluorine sources include plastic, teflon coated metals and other fluorocarbon products. These gases are capable of corroding the plant facilities. If they escape, they are producers of acid rain.

There are two types of ash residues generated by MSW incineration: bottom ash and fly ash. Their total amount equals approximately 25-35 wt% of the waste being incinerated (Tellus, 1992). Bottom ash consists of unburned materials which are mostly non-toxic, and accounts for 90% of the total ash. The remaining 10% is fly ash that contains particulates being captured by the pollution control devices (Tellus, 1992). Paul Connett, a researcher and dioxin expert contends, "The better the incinerator is at protecting the air, the more toxic the ash is going to get." (Firstman, 1989). Therefore, the contamination levels with metals, dioxins and other pollutants in fly ash are expected to be much higher than the bottom ash. When fly ash is landfilled with bottom ash, contaminants can release into leachate, which in turn has the potential to contaminate groundwater. Due to its toxicity, the U.S. Supreme Court prevents ash from municipal incinerators from being disposed in MSW landfills, and it has to be managed according to toxic waste regulations.

The first attempt to remove environmental emissions is to avoid their formation by removing potentially hazardous materials in the waste. Removing metallic wastes can reduce heavy metal emissions. Rejecting yard and food waste can lower NO_x formation. Further reduction of emissions has to rely on pollution control devices. An electrostatic precipitator uses a series of electrodes to remove particulate matters in flue gases. The same pollutants can also be removed by fabric filters, or baghouses, which consist of a series of cylindrical filtering bags. The efficiency of these devices ranges from 99.7% to 99.99% (BCEALR, 1993). Acid gases are treated by scrubbers which use alkaline reagents such as lime to react with the flue gas. Acid gases contained in the flue gas are neutralized by the alkaline and salt is formed as a residue.

External Costs of Incineration

In the Tellus Packaging Study, the pollutants that have been quantified for incineration include CO, NO_x, particulates, SO_x, VOCs, HCl, HF, PAHs and PCDDs/PCDFs. These pollutants are apportioned to the packaging waste according to their composition.

Bottom ash and fly ash are also evaluated for their effects on leachate release in ash landfill leachate. It was found that the amount of pollutants releasing to the environment from ash is minor compared to those from air emissions (Tellus, 1992).

It has to be noted that the values of the pollutants evaluated were obtained from newer mass burn incinerators, and only the controlled pollutants are quantified, in other words, pollutants that have passed through pollutant control devices. Pollutants generated within the plants are not evaluated.

Table 6.4 External Costs for Incineration of Packaging Materials

Material	Environmental Cost of Production	Conventional Cost of Disposal	Environmental Cost of Disposal		Total External Cost
			Collection	Air Emission	
Aluminum	1933	189.24	3.04	0.9	2126.18
Ferrous	230	134.67	1.20	0.9	366.77
Glass	85	106.79	0.28	1.04	218.99
Paper	269	85.86	1.10	1.63	365.59
Corrugated	214	93.62	1.21	1.63	306.46
HDPE	292	162.15	2.56	1.44	458.15
PET	854	165.76	2.61	1.44	1023.81

Source: Tellus, 1992

The total external costs associated with mass burning of packaging waste are shown in Table 6.4. The conventional costs of disposal for incineration are different from those for landfilling because in incineration, the cost for ash disposal and the revenues for energy recovery are included. The environmental cost for collecting MSW to incineration is the same as that for landfilling because it is assumed that garbage treated by the two methods was collected by the same trucks. The environmental costs due to air emissions are again minimal compared to those of production. Air emission costs for incineration are highest for paper and plastic products. The reason is because those two types of materials contain chlorinated constituents and other chemicals for the formation of acid gases. VOCs and metals are also present in these two materials, and the combustibility of these two materials makes them more reactive in the combustion process to release those pollutants.

Incineration is one effective means to reduce refuse volume. It should take a more active role in the solid waste management scheme. Incineration plants should be equipped with efficient furnaces and air pollution control devices. The types and quality of waste to be burnt should be carefully controlled. The Government should set sufficient and strict regulations on emission limits from incineration plants. There should be programs established to monitor and test air emissions from incineration facilities and operators of the plants should be professional and disciplined in following air emissions regulations. If the above are followed, incineration should not be as great a threat to the public as it is now perceived.

RECYCLING

Recycling refers to the processing of used materials which would otherwise become discards, and remanufacturing those materials into new products. When the material is made into the same product as its previous life, the process is considered primary recycling. If the material is turned into a new product with less stringent specifications than the original product, it is termed secondary recycling (Selke, 1990). Recycling does not only operate as an individual strategy for waste management, but is also integrated into waste-to-energy and landfilling operations to recover recyclable materials.

Applying the definitions, composting is categorized as one form of recycling. However, since composting employs a set of technologies different from recycling, it will be discussed and evaluated in a separate section in this chapter.

Collection Methods

There are three types of collection methods now in use: curbside collection asks for separated or commingled recyclables to be set on curbside for pickup by a collection vehicle; drop-off collection requires participants to bring their recyclables to a drop-off center; and buy-back collection is similar to drop-off except that participants are compensated for their recyclables. Drop-off centers can be attended by personnel or they can simply provide sets of containers in the centers for drop-off. Drop-off collection is more common in multi-family establishments, but the participation rate in drop-off recycling is low compared to the rapidly rising curbside collection. In buy-back recycling, since participants are paid for their recyclables, the quality of the materials is high; therefore the materials require little processing except consolidation. Sometimes, they can be sent directly to manufacturers. Statistics, however, show that both drop-off centers and buy-back centers seldom capture as much as 10% of the waste stream (NREL, Vol VII, 1992).

The recyclables gathered can be picked up either by MSW packer trucks or recycling trucks. MSW packer trucks can alternate their schedule between collecting MSW and recyclables or they can co-collect the two types if the recyclables are contained in some distinguishable bags such as blue bags.

Recycling trucks consists of several compartments (5 to 6 the maximum). Recyclables are separated by types and are put into separate compartments. To prevent damage of the collected materials, recycling trucks usually do not compact the collected materials like the conventional MSW trucks do. Figure 6.5 summarizes the various pathways for recyclables collection.

Processing the Recyclables

The place where collected recyclables are processed into marketable forms is called a MRF, materials recovery facility. Typically, three functions are carried out in a MRF:

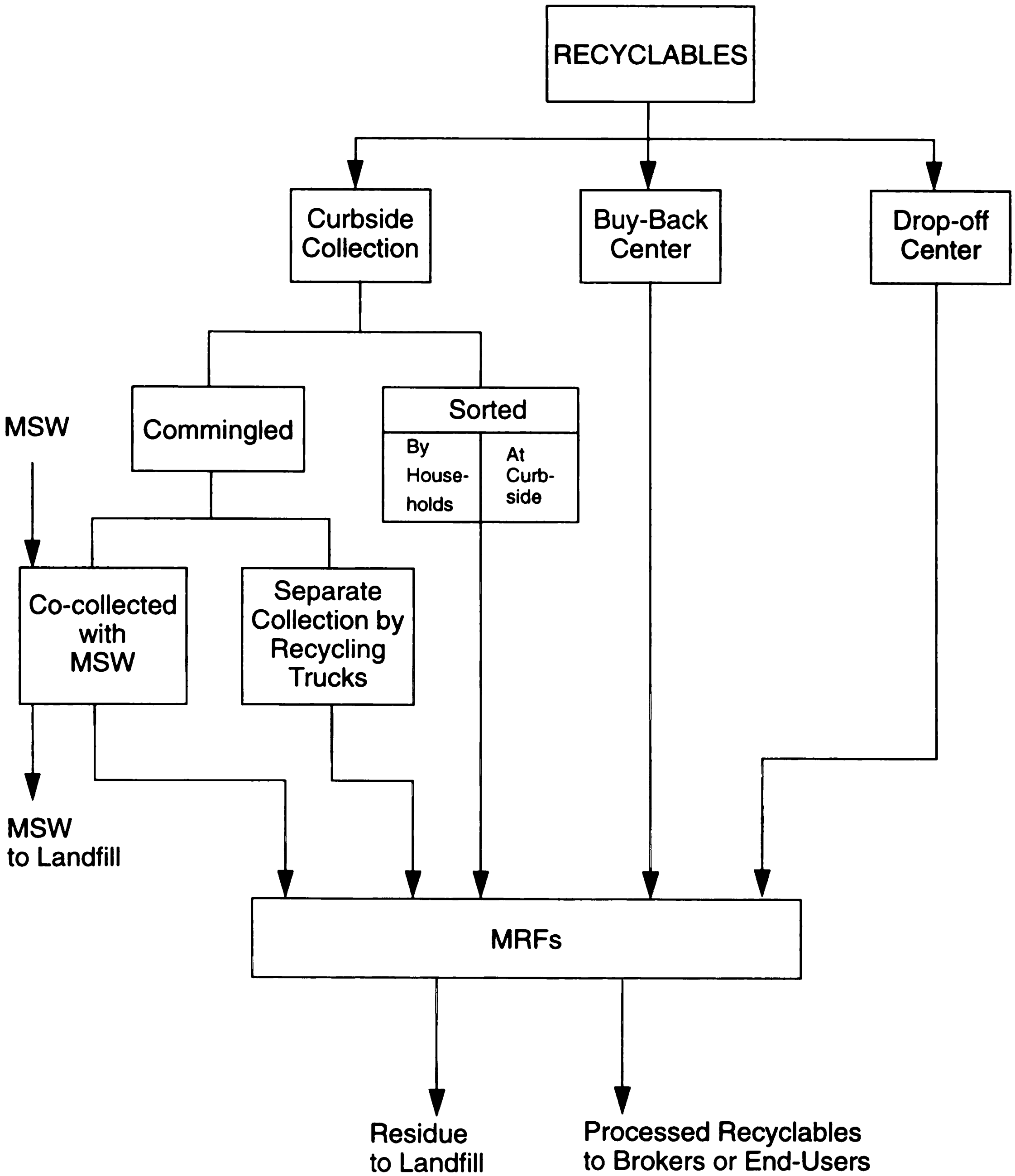
1). sort and separate the recyclables by types,

2). reduce separated recyclables to marketable sizes, and

3). bale or pack recyclables for shipment.

The degree of automation in a MRF depends on the amounts and types of materials the facility is handling. In any case, a certain degree of manual operation is inevitable. Old corrugated cardboard, paper, and plastic containers have to be sorted manually, and glass containers are separated into three colors, most commonly by manual workers as well.

In a MRF which accepts both papers and containers, there will usually be two processing lines: one for paper recyclables and the other for separating various types of containers. In the paper processing line, old corrugated cardboard will first be sorted out because it can easily be spotted. Other workers working along the sorting table will separate magazines from newspaper. Sorted papers may be baled with or without shredding, depending on the requirements from the end-users. Mechanical balers are used to bale the papers for shipment.



Source: NREL, Vol. VII, 1992

Figure 6.5 Pathways of Recyclables Collection

In a container processing line, sorting the containers by their types involves more steps than paper processing. Commingled containers are passed onto a conveyor belt. Magnetic separators will sort out steel cans and other ferrous metals from the stream of containers. The magnetic separator is made of a permanent or electromagnetic type of magnet in the form of a drum, a pulley or a belt. The ferrous materials may be flattened by a flattener or reduced to small pieces by a shredder, depending on the specifications of the end-users. Aluminum cans can be mechanically separated by an eddy-current aluminum separator. An air classifier sorts out the heavy containers from the light ones. It operates with a stream of moving air on which the lighter materials such as aluminum, plastics and paper flow while the glass containers stay at the bottom. Once aluminum and plastics are separated from the glass, they are conveyed to a sorting station and manually sorted by workers who further separate the containers into aluminum and different types of plastics. The glass containers are diverted to their sorting station where flint, amber and green glass are separated out manually. Besides using the eddy-current or air classifier methods, one can install an inclined sorter along the conveyor. The slight inclination will sort out lighter containers like aluminum and plastics by making them drop to the side.

Once each type of material is gathered with its own kind, it is densified through size reduction. Glass containers are crushed and then boxed for shipping. PET and HDPE are granulated. Aluminum cans are flattened.

The more recyclables a MRF has to handle, the more mechanized the processes are. When the operation is small, some of the mechanical sorting processes are simply replaced by manual sorting.

There are MRFs that handle mixed solid waste as well, called mixed waste MRFs. They are usually operated in conjunction with WTE facilities. In such a facility, mixed MSW arrived at the tipping floor is first sorted to remove bulky and non-processible waste. In-feed wastes are then sent to conveyors where bags are opened manually or by bag-breaking bars. They are then screened by a trommel or disc screen. Screening separates materials into two sizes: undersize and oversize. An example for the primary screen size is 5 inches. Oversize materials including corrugated cardboard, newsprint, and office paper will be manually sorted. Ferrous materials are picked up by magnets. The remainder of the oversize refuse is sent to WTE plants. Glass containers in the undersize stream are manually picked out. They are sorted by colors, then crushed and screened before loadout. Magnetic separation is used again to pick out steel materials among the undersize. The remainder of the undersize then undergoes a second screening of 2 inches. The overs are manually sorted for aluminum, PET , and HDPE. The unders are conveyed to a common refuse station. Schematics of the design of a mixed waste MRF may vary from plant to plant but similar equipment and procedures are involved as described.

Table 6.5 Contaminant Tolerance for Recyclables

Material	Contaminant Level (wt.%)
Newsprint	≤ 2%
Glass Cullet	Other 2 colors ≤ 5% each Non-glass contaminants ≤ 1%
Aluminum	Non-aluminum contaminant & moisture < 1.5%
Tin-Plated Steel Cans	< 2%
PET	< 3%
HDPE	Colored HDPE content ≤ 10% Non-HDPE, non-plastic contaminants ≤1%
Mixed Rigid Plastics	Non-plastic material < 3%

Source: NREL, Vol VII, 1992

Mixed waste MRFs usually recover only 10-20% of the wastes in marketable forms, whereas MRFs can recover 80% or more (Diaz, 1993). In either type of MRF facility, processed materials should be almost free from contaminants in order for the materials to be acceptable to end-users. Table 6.5 lists the contaminant tolerance of each material.

Environmental Contamination

Environmental contamination of recycling comes from two main sources: during collection and during processing. Collection vehicles exhaust fumes while the engines are on. Stopping and going during pick-up of recyclables or garbage cause the vehicle to emit more pollutants than if it was in constant motion. Also, the compaction cycles in a garbage truck cause more pollutants to be generated than a recycling truck which typically does not compact its collection. During processing of recyclables in the MRF, minute amount of particulates, VOCs, and metal emissions may be released due to materials' interaction with mechanical units. Movement of wastes in the plant may generate dust and odors. MRFs are typically equipped with ventilating units and fabric filters to ease the problem of air emissions inside the facilities. Noise pollution is also generated by collection vehicles and the machinery. Other environmental contamination such as vermin infestation should not be significant.

External Costs of Recycling

Tellus' data on the external costs of MSW recycling are listed in Table 6.6. The conventional cost of recycling includes transporting, collecting, processing and disposing costs of the solid waste plus revenue from the sale of the recyclables. It is therefore not surprising for aluminum to earn a negative value in this category. Paper packaging just breaks even between its cost and revenue, while other materials on the list cost to recycle.

Table 6.6 External Costs of Recycling Packaging Materials

Material	Environmental Cost of Production	Conventional Cost of Disposal	Environmental Cost of Disposal		Recycling Credit		Total External Cost
			Collection	Facility	% Recycled Content	Credit[2] (in negative $ value)	
Aluminum	1933				100%[4]	-1620.0	-228.97
Recycled Aluminum	313	-553.5	11.53	N.A.[3]	75%	-1215.0	176.03
					55%[5]	-891.0	500.03
Ferrous	230				40%[4]	-8.0	326.90
Recycled Ferrous(40%)	222	101.44	3.46	N.A.	26%	-5.2	329.70
					12%[5]	-2.4	332.50
Glass	85				100%[4]	-30.0	83.27
Recycled Glass	55	27.12	1.15	N.A.	65%	-19.5	93.77
					30%[5]	-9.0	104.27
Paper	269				100%[4]	-134.0	141.92
Recycled Paper	135	0.0	6.92	N.A.	75%	-100.5	175.42
					47%[5]	-63.0	212.92
Corrugated[1]	214				100%[4]	-64.0	275.38
Recycled Corrugated	150	122.31	3.07	N.A.	60%	-38.4	300.98
					19%[5]	-12.16	327.22
HDPE	292	366.38	19.77	N.A.	N.A.	N.A.	678.15
PET	854	357.76	23.06	N.A.	N.A.	N.A.	1234.82

[1]Corrugated cardboard data is calculated from using 69% corrugated medium and 31% linerboard (Schall, 1992)
[2]Credits are represented as negative costs
[3]N.A.: not available
[4]Technologically feasible % (Schall, 1992)
[5]Existing Recycled Content (Schall, 1992)

Source: Tellus, 1992; and Schall, 1992

The environmental cost of disposal is based on the emissions generated during the collection of the recyclables. Emissions from recycling trucks are quantified. The amount of emissions of different kinds is apportioned to different recyclables based on their in-truck volume, because recycling trucks are filled up by volume. Tellus did not include emissions from processing facilities in the environmental cost due to sampling problems in recycling facilities during their study. Those data were unavailable from other sources either.

EPA's Environmental Criteria Assessment Office has completed a study entitled, "Public Health, Occupational Safety, and Environmental Concerns in Municipal Solid Waste Recycling Operations" (EPA, 1993). The study, however, reports the relative significance of the hazards in terms of low, medium or high impacts only. Table 6.7 summaries their findings. Another study, carried out by the Solid Waste Association of North America for EPA's MITE program, has evaluated the air, water and noise pollution in six MRFs. The report will be published in spring 1995. According to the project officer of this study, the pollution does not pose much environmental concern (Frola, 1995).

When comparing the conventional cost of recyclables collection to those of garbage collection (Table 6.2), glass, aluminum and ferrous containers and paper cost more to be collected as garbage than as recyclables. The outcome is the result of a combination of factors including materials' density, materials' revenue, recycling rate and recyclability. Aluminum generates high revenue. Glass, due to its high density and large quantity, is easy to collect and process for recycling. Paper is cheaper to recycle than to dispose owing to its high recycling rates. On the other hand, PET and HDPE cost more to recycle because of their light weight and bulkiness.

The environmental costs of recyclables collection are higher than those of garbage collection because the costs were determined according to the amount of

Table 6.7 Environmental Impacts of Municipal Solid Waste Recycling Operations

	Public Health Concerns		Occupational Safety Concerns		Environmental Safety Concerns	
	Activities		Activities		Activities	
Concern Item	Collection and Sorting	Material-specific processing	Collection and Sorting	Material-specific processing	Collection and Sorting	Material-specific processing
Sharp objects	low	NMS	medium	low		
Ergonomic and lifting injuries	low	NMS	medium	low		
Fires and explosives	low	NMS	medium	low	low	low
Flying and falling debris	low	NMS	medium	medium		
Temperature and pressure extremes			low	low		
Moving equipment and heavy machinery	low	NMS	low	low		
Noise	low	NMS	medium	low		
Aesthetic impacts	low	NMS				
Traffic	low	NMS	low	NMS	low	NMS
Process chemicals and container residues	low	NMS	medium	medium		
Gaseous releases	low	NMS	low	low	low	low
Particulate releases	low	NMS	medium	medium	low	low
Waterborne releases	low	NMS			low	low
Solid waste and sludge	low	NMS			low	low
Microbiologic	low	NMS	low	low		
Pests	low	NMS	low	low		

NMS: The hazard identified is not specific to any particular material

Source: EPA, 1993

the various kinds of pollutants emitted by collection vehicles apportioned per ton of packaging waste. Since recycling trucks do not compact the recyclables, the amount of pollutants apportioned per ton of materials in a recycling becomes higher. On the whole, the total external costs of recycling turn out to be comparable to the total external costs of landfilling or incineration.

COMPOSTING

Composting, being regarded as one form of recycling in the EPA's hierarchy of solid waste management, does fit the classification since it *remanufactures* (composts) organic matter into new products (compost). The more precise definition for composting is the biological decomposition of wastes consisting of organic substances of plant or animal origin, under controlled conditions, to a state sufficiently stable for nuisance-free storage and utilization (Diaz et al., 1993). The mentioning of "under controlled condition" is important because it signifies composting is a deliberate process that inputs suitable conditions to promote the decomposition of the organic waste, instead of letting it go through a natural degradation such as that in a landfill.

Composting can be classified as aerobic or anaerobic. Composting in the presence of oxygen creates less odor than anaerobic composting. During aerobic composting, energy is generated by microbial activities, hence raising the temperature of the compost mass high enough to kill the pathogens present, making the process safer for human health. Aerobic decomposition is also faster than anaerobic decomposition. The latter requires structures to ensure total enclosure of the waste and control of offensive odors. Therefore, most of the current composting activities are aerobic.

Composting depends on microbes that are present in the waste to attack and degrade the organic matter in the waste. Three types of microorganisms are involved in the process, namely bacteria, actinomycetes and fungi. Bacteria are responsible for a large part of the degradation. Actinomycetes and fungi specialize in attacking cellulosic and lignaceous components of the waste. Paper is thus one of their targets.

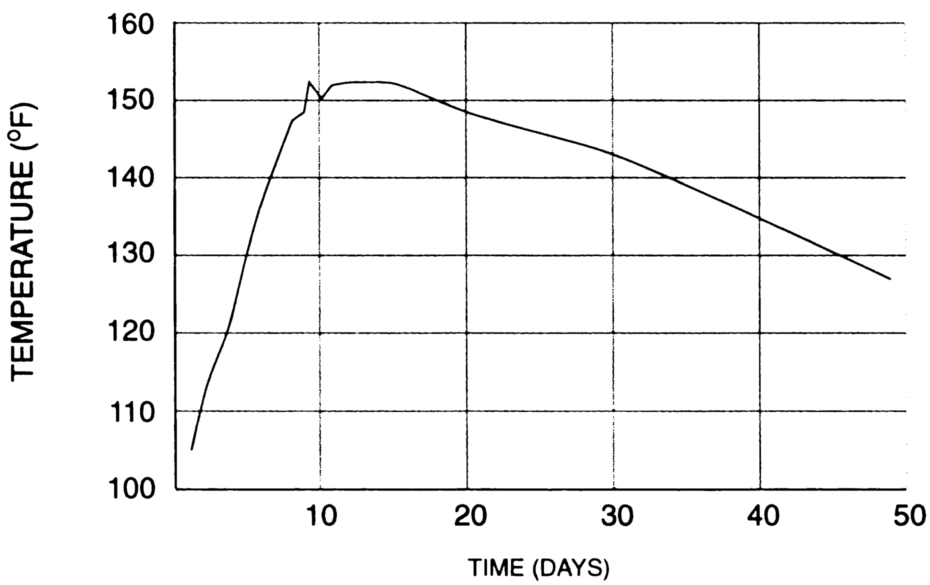
To compost, sufficient amount and varieties of microbes must be present in the waste. This prerequisite is fulfilled in most cases. Other factors affecting the rate of composting include the availability of nutrients in the waste, carbon to nitrogen ratio, particle size, temperature, aeration, moisture content and pH level.

Composting Conditions

Nutrients: microorganisms have to be supplied with nutrients of which their cellular mass is composed, in order to reproduce and perform decomposition. Certain nutrients are also necessary as their energy source or enzyme constituents. Nutrients that are needed in large amounts are called macronutrients and include carbon, nitrogen, phosphorous and potassium. Micronutrients are needed in small amounts, and include cobalt, manganese, magnesium, copper, and others. These nutrients are usually present in adequate amounts in the waste to be composted (Golueke, 1977).

C:N: During metabolic activities, carbon is oxidized to carbon dioxide by microorganisms. Additional carbon is converted into cell wall, membrane, protoplasm and other storage products. Nitrogen is required for the synthesis of protoplasm. In comparison, carbon is in higher demand than nitrogen. The optimum ratio of the two elements should be 20:1 to 25:1. If the ratio is lower than the optimum, that is excess nitrogen is present, microbes will convert the excess nitrogen to ammonia, which may eventually be volatilized. A deviation of the desired C:N ratio will slow down the composting rate. The remedy for it is to add nitrogenous waste if C:N is too high and add carboneous waste if C:N is too low (Golueke, 1977).

Particle Size: The smaller the particle size of the waste, the more the surface area is exposed to microorganisms for decomposition. The particle size, however, cannot be so small that it fills up interstices, and therefore obstructs aeration. The minimum permissible particle size depends on the type of wastes. Waste of good structural strength may acquire a particle size in the range of 0.5"- 3". Green plant waste, on the other hand, should have particle size greater than 2" (Golueke, 1977).



Source: Diaz et al., 1993

Figure 6.6 Typical Temperature Curve of Composting Waste

Temperature: Figure 6.6 is a typical temperature curve for composting waste. Heat generation is a by-product of metabolic activities of microorganisms. Once the waste is placed in a suitable condition and environment for composting, temperature starts to rise gradually. When the waste is accustomed to and settled down in the environment, temperature rise becomes exponential, due to the easy decomposition of wastes such as sugars and starch. Temperature begins to level off at around 150^{o}F and eventually starts to decline when most of the easily decomposed materials are broken down. A final drop in temperature is an indication of a mature compost. A temperature above 150^{o}F is undesirable because high temperatures favor the formation of spores which lead to a slow-down of decomposition. Microbes that are incapable of spore forming are killed at such high temperatures. Temperature of the compost should therefore be kept at around 150^{o}F (Diaz et al., 1993).

Aeration: Aeration is important to composting as it keeps the decomposition aerobic. A sufficient amount of oxygen provided to the waste maintains the expected rate of composting and keeps foul odors from forming. Depending on the structure of the composting waste, aeration can be provided by frequent turning of the waste or by pipes which feed the waste with forced air.

Moisture Content: All microbial activities cease when the moisture content is less than 8-12% (Diaz et al., 1993). On the other hand, too high a moisture content decreases the presence of interstices and in turn creates an anaerobic environment. The optimum moisture content is a function of the structural strength of the waste particles. Higher strength permits higher moisture content (75-80%).

When food waste is the major component, a bulking agent, which is any material with a high structural strength, should be added to the waste to create porosities and to absorb some moisture. In general, moisture content should always be kept above 40% (Diaz et al., 1993).

pH level: the pH level of a composting waste seldom drops below the permissible level. Should it occur, lime can be added to the waste. pH level usually drops to 5.0 due to the formation of organic acids. These acids are substrates for subsequent microbial populations, and therefore pH will rise to as high as 8.5 as these acids are consumed.

Composting Technology and Processes

1). Collection

Materials to be composted can be collected from mixed MSW or separated organic waste. The latter usually includes leaf and yard waste only. Some may include food waste. Currently, mixed MSW is the main source of feedstock for MSW composting facilities.

2). Preprocessing

The preparation procedures for waste to be composted are very similar to that of incineration. The goals are sorting and size reduction. Incoming waste is first weighed, then manually inspected for non-compostable or hazardous materials. The waste is then passed through a magnetic separator to sort out ferrous metals. Remaining waste is shredded into a particle size of 2 to 3 inches. Shredded waste may be further screened for glass, stones and other contaminants. Residues are sent to RDF facilities, to MRFs, or are landfilled. Water is added to the compostable waste until the moisture content is 50-60%.

3). Compost Process

Two types of systems are now in use: windrow (open) and mechanical (closed) systems. Windrow systems can be further classified as static and turned windrow systems.

a). Static Windrow System

In a static windrow system, materials to be composted are formed into a stationary elongated pile (windrow). Air is forced through the pile by a mechanical system.

First, the preprocessed waste is mixed with a bulking agent if needed. The mixed waste is then ready to be formed into a windrow. A windrow has to be constructed above paved surfaces to prevent possible leachate from contaminating the groundwater and for ease of material handling. To construct a static windrow, a long perforated pipe connecting to a blower is placed above a compost pad. A layer of bulking agent is covered on top of the pipe to absorb moisture and for the uniform distribution of air to the waste. The preprocessed waste is piled in the form of a windrow, roughly conical in cross-section, above the pipe with the pipe oriented at the ridge of the windrow. The conical shape facilitates the sliding of precipitation instead of accumulating and seeping into the compost pile. One end of the perforated pipe should be buried entirely into the pile to prevent short-circuiting of air. The pile is covered with a layer of matured compost to absorb odors from the compost mass (Diaz et al., 1993).

b). Turned Windrow System

The difference between turned windrow and static pile lies in their methods of aeration. A windrow of the same configuration as the static pile is constructed in a turned windrow system, but without the pipe and blower. Instead, the windrow is torn down and reconstructed periodically to expose the previously unexposed particles. Frequency of turning depends on the environmental conditions. On the average, a windrow pile is turned once every four days.
In both windrow systems, leachate collection devices should be prepared. In places where there is heavy rainfall, windrow piles should be sheltered. In desert areas, windrow should be protected from wind to reduce moisture loss (Diaz et al., 1993).

c). Mechanical Systems

In contrast to an open system in which a windrow is constructed, a mechanical system, also called a closed system, contains the waste to be composted in an enclosed “reactor” or “digester”. The mechanical reactor provides uniform aeration by stirring, rotating or tumbling motions, depending on the design of the reactor. The reactor is also equipped with air inlets and outlets for aeration. Due to the high cost of utilizing a mechanical system, composting mass is usually treated by a reactor for one to three days only. It is then taken out for windrowing for another month or more. This period is called curing. The time required for a complete composting process can be understood via Figure 6.7.

4). Postprocessing

Indicators of mature MSW compost include its change to a dark gray color, and a granular texture. When the temperature of the composting mass drops to 40-45°C, the compost is stable and can be postprocessed and stored without nuisance (Golueke, 1977).

Postprocessing activities may include size reduction, final screening and air classification.

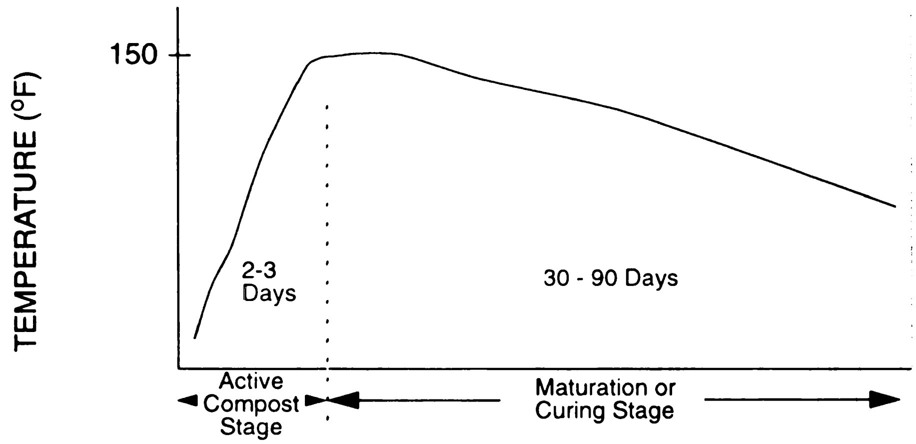


Figure 6.7 Schedule for a Complete Composting Process

Environmental Impacts

MSW composting is still at its infancy stage. There are few detailed discussions on its environmental impacts, The availability of quantitative data is also scarce. The potential environmental and human health related impacts of composting are discussed as follows:

1).Odors

The number one problem with composting is offensive odors. It is inevitable that some odors will be generated in a composting facility. Odors are composed of chemical compounds like ammonia, hydrogen sulfide, dimethyl disulfide, etc. It is stated that odors do not become a health hazard until they become particularly intense (Diaz et al., 1993). There is, however, no standard for that intensity level. Composting facilities keep the odors down to a level based on human judgment and most often of all, based on reactions from area residents. Two methods of odor control are possible, namely, chemical scrubbing and biofiltration. Chemical scrubbing can reduce the odors to 75-200 DT (Dilutions to Threshold). Biofiltration's performance limits are 10-100 DT. As references, a swimming pool has an odor level of 120-200 DT and an average room smells at a level of 20 DT (Kowalczyk, 1994).

Foul odors are generated by anaerobic decomposition. If the composting process is carefully managed and properly carried out, odor levels can be minimized. It is, however, easier to say than do. Some composting facilities are closed down due to unresolved odor problems. Researchers have to continue the search for the proper process management in order to keep the level of odors down.

2). Air Emissions

Air emissions may be generated during the collection of the waste and during the composting process. Air emissions from collection trucks should be quantified and apportioned to the waste collected to be composted. When this waste is collected in the form of mixed MSW, its environmental impact due to collection truck emissions should be less than that from recycling trucks.

During preprocessing in which shredding and mixing of waste occur, and in the course of windrow turning and compost transferring, dust will be generated in the facilities. During composting, carbon dioxide will be emitted from the pile. If the process turns anaerobic at one time or another, methane gas will be produced. Microbes might be transported out of the pile by dust particles. All these emissions are yet to be quantified.

3). Leachate and Run-Off

When the moisture content is high (>60%), leachate may form, but it can be minimized by sheltering the compost pile from precipitation and by operating on hard surfaces (Diaz et al., 1993). So far, leachate and run-off have not been reported as problems in composting facilities.

4). Vectors

Putrescible waste can attract rodents and flies. Composting facilities should take serious controls on the hygienic situations within and in the proximity of the facilities.

Table 6.8 Trace Elements in Soil and in Compost
(Unit: Parts per Million)

ELEMENT	IN SOIL	IN COMPOST	PROPOSED U.S. STANDARDS
Cadmium	0.06	3.4 (2.3 - 7)	18
Chromium	100	223 (159-828)	2000
Copper	20	285 (190-912)	1200
Lead	10	496 (348-1250)	300
Mercury	0.03	4.0 (0.6-5.9)	15
Nickel	40	77 (39-709)	500
Zinc	50	1008 (596-1370)	2700

Source: NREL, Vol. I, 1992

5). Soil contamination

When compost is applied to soil, it decomposes and releases elements into the soil. Estimates of the amount of trace elements found in soil and in compost are presented in Table 6.8. There has also been talk about the presence of dioxins in composts. The level seems to be low, but detailed characterization has not been done (NREL, Vol I, 1992).

Environmental Costs of Composting

The calculation of the environmental costs of MSW composting cannot be achieved at this stage because little has been done in quantifying the environmental impacts of composting. With the industry still trying to sort out the consistent and approved procedures for performing composting, a lot of factors vary from time to time. In the area of packaging, paper is the only material which is compostable. The goal to quantify the environmental impacts of paper composting is even more difficult because paper is not only being composted in MSW composting. It is also added into leaves and yard waste composting, and mixed paper and sludge composting (Goldstein, 1992). Experiments are still being carried out to investigate the effect of paper to sludge ratio on the physical properties of compost mass (Anderson and Smith,1994).

Tellus did not include composting in their study. When the technology of composting becomes mature and more data become available, the environmental impacts discussed in the previous section should be quantified and monetized according to Tellus' methodologies. Monetized impacts should then be apportioned to paper according to composition analysis of the compost mass. Different ratings might result from the composting of paper with MSW and with sludge, or a weighted average can be calculated from the two types of composting.

In the meantime, the rating of composting still remains somewhat arbitrary. To facilitate the calculation of the total environmental costs of composting in the future, the following equation summarizes the costs to be included (all costs are presented in per ton of packaging material):

Total Environmental Cost of Composting a Packaging Material

$= C_e^p + C_c^c + C_e^c + C_{eg}^p + C_d$

where

C_e^p =environmental cost of production, which is equal to the virgin material production since composting does not substitute any virgin material production for recyclables production.

C_c^c = conventional cost of composting, which includes the monetary costs of collecting and transporting the compostable waste, the processing of the materials into compost and the revenue from the compost.

C_{eg}^p= energy cost of production

C_d = cost of raw material depletion

C_e^c =environmental cost of composting = $C_e^o + C_e^t + C_e^f + C_e^l + C_e^v + C_e^s$

where

C_e^o = environmental cost of odors

C_e^t = emission cost of collection trucks

C_e^f = emission cost from composting facility and from compost pile

C_e^l = environmental cost of leachate from compost pile

C_e^v = environmental cost of vector attack

C_e^s = environmental cost of soil contamination

REUSE

To reuse, in the context of solid waste management, means to utilize a product or a package a second time or more in its original physical or structural form without any remanufacturing. By this definition, returnable and refillable packages are considered as two forms of reuse. Although methods to reuse are easier to conceive than those of source reduction, similar to the fate of source reduction, reuse is not getting as much emphasis as recycling or other waste management options in spite of its high ranking in the EPA's hierarchy. A number of reasons can account for this phenomenon:

1). Reuse is similar to source reduction in the sense that it requires a change of life-style and behavior from consumers. Habits are hard to change.

2). Manufacturing processes have to be modified for producing reusable or refillable packages.

3). An infrastructure for transporting and collecting returnable containers has to be developed.

4). Workers have to be retrained to handle reusable containers or adopt to new ways of packaging.

5). Start-up costs for producing or utilizing reusable packages are higher than one-way packages.

6). Research is needed to develop reusable packages to replace the less enduring one-way packages.

7). The reuse of food containers may raise hygienic concerns.

8). Transporting and cleaning of containers may add negative impacts to the environment.

Among the above listed obstacles, the need to establish an infrastructure for package return is one major difficulty for developing package reuse on a large scale. Such infrastructure, which used to be present years ago in the beverage industry, is non-existent today. To re-establish it requires vast investments and meticulous planning. Transport distances are no longer confined within a small community like that in the old days. Products being manufactured in a single plant are shipped across state lines and distributed to thousands of retailers. The foremost concern in developing a system for the return transportation is whether such a system is economically sound. The second concern is whether it is indeed environmentally beneficial. Research has yet to be carried out to answer those questions.

Even though reuse has not been developed as one popular form of waste management, some reuse efforts have been made across the country. In the government sector, five states have source reduction procurement programs in which reuse is addressed (Fishbein and Gelb, 1992). Connecticut is the only state that has legislation requiring state agencies to take steps in eliminating products that are not reusable. The act, however, does not mandate specific actions but make recommendations only. Suggestions of reuse in state agencies made by Connecticut or other states include the use of refillable ball point pens, reusing envelopes, making duplex instead of one-sided photocopies, using large reusable containers for dressings, buying certain items in bulk, and buying copiers with duplex capabilities, to name a few. All of these suggestions require behavior modification from users. In addition, higher prices have to be paid for the reusable items because they are usually more expensive than their disposable counterparts. They are, however, expected to pay off in the long run.

At institutions such as hospitals, correctional facilities and schools, examples of reuse activities include the switch to reusables in food services in the New York prisons, the replacement of disposable corrugated cardboard with reusable containers in New York City Health and Hospital Corporation, switching from disposable bed pads to reusables in the Hospital of St. Raphael, and reusing paper that has been used on one side in Roslyn High School (Fishbein and Gelb, 1992).

In the business sector, a few companies are known for reusing: Allied-Signal's Bendix Automotive Systems expects to furnish collapsible, returnable shipping containers to its suppliers; AT&T is promoting double-sided copying and has made procurement policies to improve the duplexing performance of the copying machines; Herman-Miller, Inc., a major office furniture manufacturer, replaces cardboard and plastic wraps with reusable blankets in packaging their furniture; and Boston Park Hotel has installed shampoo and lotion dispensers in hotel bathrooms (Fishbein and Gelb, 1992).

Perhaps one of the potentially largest reuse activities lies in returnable beverage bottles. The use of returnable containers is much more prevalent in Europe than in the United States. For example, Pepsi-Cola has introduced a returnable plastic bottle in Poland that will cut prices by 40% from its disposable equivalent. A factory was built near Hungary to supply refillable PET bottles to Coca-Cola. The Germans are testing a polyurethane-coated glass multi-trip bottle which is scratch and impact resistant while being 40% lighter than conventional bottles. In the U.S., GE Plastics has developed a polycarbonate (Lexan) returnable bottle for dairy and non-carbonated beverages. It launched its use at some schools in Pennsylvania, which seemed to be a success. This bottle is also given favorable reviews in Europe. The cost for an 8 oz. Lexan plastic milk bottle is 30 cents, whereas a waxed carton costs 2 to 3 cents. GE estimates that including washing and handling, 70 to 80 trips per Lexan bottle is required to justify its cost (Biocycle, Aug. 1992).

Besides being made of a higher grade material, returnable containers may be made of the same material as their disposable equivalents, but the containers have to be thicker so as to provide the strength and resistance to damage during multiple trips. In either case, the production cost for returnable bottles is likely to be higher. In addition to justifying the conventional costs of production, transportation and cleaning, the added environmental cost due to multitrip transportation and the avoided environmental cost from the production of disposable containers have to be taken into account when determining the overall benefits of returnable containers. The comparison can be made by performing life-cycle assessment on the two types of containers.

Reuse is not, and should not be, limited to the commercial and governmental levels. Fellow citizens, being triggered by some imagination and innovations, can reuse materials in everyday lives as well. The incentive is to reduce personal spending. For example, brown paper bags can be opened up and be used as mailing paper, plastic grocery bags can be reused as liners for small trash cans, glass bottles are good storage containers for dried food, old envelopes can be used to hold coupons, and the ideas go on. Governments should put more efforts on promoting and providing recommendations to citizens on materials reuse.

Environmental Costs of Reuse

Recently, an European study on returnable and non-returnable packaging was published (Pöll and Schneider, 1993). In the study, so-called "eco-prices" were established for returnable and non-returnable drink containers based on their ecological impacts. In the United States, the quantification of reuse activities is unavailable at this time as material reuse is practiced in discrete sectors in different ways in this country without any infrastructure. In the meantime, however, an equation is established to prepare for the calculation of the total environmental cost of reuse should data become available in the future. The equation is compatible with the methodology being used in calculating the total environmental costs of other waste management options in this project, and is focused on evaluating the impacts of returnable packages.

Seven types of costs are involved (Note: All costs are presented per ton of packaging material):

1). *Environmental Cost of Production* = $\frac{1}{n} C^p{}_e$

where n = total number of uses
$C^p{}_e$= environmental cost of production of the reusable material

2). *Energy Cost of Production* = $\frac{1}{n} C^p{}_{eg}$

$C^p{}_{eg}$ = energy cost of production

3). *Environmental Cost of Return Transportation* = $(n - 1)C^t{}_e$

When the package is used n times, (n-1) collections are needed for its return from consumers to manufacturers or packagers.

$C^t{}_e$ = environmental cost of transportation (emissions from collection trucks)

4). *Environmental Cost of Cleaning* = $(n - 1)\ C^{c}_{e}$

A package being used n times will be cleaned at a designated facility (n-1) times

C^{c}_{e} = environmental cost of cleaning is originated from the emissions due to energy consumption for cleaning; dust and contaminants generated in the cleaning facility; water effluent from the cleaning process; and the environmental cost of managing reject packages.

5). *Conventional Cost of Disposal* = $\dfrac{rC^{r}_{c} + iC^{i}_{c} + lC^{l}_{c}}{n}$

where
r = recycling rate of the material
i = incineration rate of the material
l = landfilling rate of the material
C^{r}_{c} = conventional cost of recycling the material
C^{l}_{c} = conventional cost of landfilling the material
C^{i}_{c} = conventional cost of incineration the material

At the end of its useful life (after n uses), the package may be disposed of through landfilling, recycling or incineration. It cannot be known for sure which method of disposal the container is destined to, therefore, a weighted average of the conventional costs of disposal among the three methods is used: $rC^{r}_{c} + iC^{i}_{c} + lC^{l}_{c}$
Since for each ton of materials, it is disposed only once after n uses, therefore, the conventional cost of disposal should be shared among the n uses. Therefore, the numerator is divided by n.

6). *Environmental Cost of Disposal* = $\dfrac{rC^{r}_{e} + iC^{i}_{e} + lC^{l}_{e}}{n}$

Similar to the rationale for #5, the numerator is the weighted average of the environmental cost of disposal among landfilling, incineration and recycling. The cost is shared among n uses.

7). Cost of Raw Material Depletion = $\dfrac{C_{d}}{n}$

where C_{d} = cost per ton of a raw material

Equation For Reuse:

Therefore, the Total Environmental Cost per ton of Reusable Material

= 1) + 2) + 3) + 4)+ 5) + 6) + 7)

$$= \frac{C_d + C_{eg}{}^p + C^p{}_e + rC^r{}_c + iC^i{}_c + lC^l{}_c + rC^r{}_e + iC^i{}_e + lC^l{}_e}{n} + (n+1)(C^t{}_e + C^c{}_e)$$

$$= \frac{C_d + C_{eg}{}^p + C^p{}_e + r(C^r{}_c + C^r{}_e) + l(C^l{}_c + C^l{}_e) + i(C^i{}_c + C^i{}_e)}{n} + (n+1)(C^t{}_e + C^c{}_e)$$

Package refill can be considered as one form of reuse. There are two types of package refill practices: one requires consumers to wash and rinse the original package and bring it to the store for refill content. This refilling activity is similar to that of returnable bottles except the consumers instead of the packager perform the cleaning, transporting and refilling. Therefore, it can be modelled with the equation for returnable packages.

Another type of refilling is done by purchasing a special refill package at the store, bring that package home, and transferring the contents from the refill package to the original container. The evaluation of the worthiness of this type of refilling activity to the environment requires an assessment in terms of the original package and the refill packages. This project evaluates the impacts of various solid waste management methods in terms of packaging materials because it is more practical to perform the evaluation in terms of material types. Assessing each single package in the market is impossible, and the results would be too distinctive to draw generalized conclusions from. Since special refillable packaging always involves two types of packages, which may be made of different materials by different technologies, it has to be assessed and described in terms of the packages instead of per ton of the materials used. It is therefore decided that the evaluation of the environmental impacts of this type of refillable packages is not compatible with the rating scheme of this project and hence it should be assessed in a separate study.

CHAPTER VII
THE RESULTS

TOTAL ENVIRONMENTAL COSTS

The itemized environmental costs of packaging material production and the conventional and environmental costs of disposal by landfilling, incineration and recycling are summarized in the first five columns in Tables 7.1(A), 7.1(B) and 7.1(C). The seven types of packaging materials listed in the tables are selected from the Tellus Packaging Study. Aluminum refers to the type that is made into soft drink cans. Steel refers to the type for making steel cans. Glass refers to the type for bottles, and HDPE and PET refer to the grades that are manufactured into containers. In each of the three tables, the environmental costs of production are extracted from Table 3.3. The conventional costs of landfilling, incineration, and recycling are taken from Table 3.2. The energy costs of production are taken from Table 5.1. Lastly, the social values of raw material depletion corresponds to the costs listed in Table 5.3.

In Table 7.1(C), the costs of recycling packaging materials are identified with the percentages of recycled content specified. Since aluminum cans, glass containers, corrugated cardboard and folding boxboard can be made with 100% recycled content, and cost data are available for the production of these 100% recycled materials, the environmental impacts of recycling these materials are

Table 7.1 Itemized Environmental Costs ($/ton material) for:

(A) LANDFILLING

Material	Environmental Cost of Production	Energy Cost of Production	Social Value of Raw Material Depletion	Conventional Cost of Landfilling	Environmental Cost of Landfilling	Total Environmental Cost
Aluminum	1933	1110.72	128.36	279.66	3.04	3454.78
Steel	230	53.35	39.29	127.86	1.20	451.70
Glass	85	72.09	29.53	34.09	0.28	220.99
Boxboard	269	159.99	72.50	106.97	1.10	609.56
Corrugated	214	133.04	61.72	119.41	1.21	529.38
HDPE	292	210.66	88.91	335.82	2.56	929.95
PET	854	290.34	101.28	339.13	2.61	1587.36
					Average:	1112.00

(B) INCINERATION

Material	Environmental Cost of Production	Energy Cost of Production	Social Value of Raw Material Depletion	Conventional Cost of Incineration	Environmental Cost of Incineration (Collection + Emissions)	Total Environmental Cost
Aluminum	1933	1110.72	128.36	189.24	3.04 + 0.9	3236.90
Steel	230	53.35	39.29	134.67	1.20 + 0.9	459.41
Glass	85	72.09	29.53	106.79	0.28 + 1.04	294.73
Boxboard	269	159.99	72.50	85.86	1.10 + 1.63	590.08
Corrugated	214	133.04	61.72	93.62	1.21 + 1.63	505.22
HDPE	292	210.66	88.91	162.15	2.56 + 1.44	757.72
PET	854	290.34	101.28	165.76	2.61 + 1.44	1415.43
					Average:	1037.00

(C) RECYCLING

Material (% Recycled Content)	Environmental Cost of Production	Energy Cost of Production	Social Value of Raw Material Depletion	Conventional Cost of Recycling (includes revenues)	Environmental Cost of Recycling	Total Environmental Cost
Al (100%)	313	44.43	0	-553.50	11.53	-184.54
Steel (40%)	222	47.53	33.15	101.44	3.46	407.58
Glass (100%)	55	52.97	0	27.12	1.15	136.24
Bxbd (100%)	135	122.23	0	0	6.92	264.15
Corugat. (100%)	168	119.81	0	122.31	3.07	413.19
HDPE (0%)	292	210.66	88.91	366.38	19.77	977.72
PET (0%)	854	290.34	101.28	357.76	23.06	1626.44
					Average:	520.00

evaluated based on this percentage. On the other hand, a maximum of 40% recycled content is allowed in the production of steel cans, therefore, the evaluation of steel can recycling is based on this maximum allowable content. Currently, little or no recycled content is included in the manufacturing of PET and HDPE containers. No data are available for the recycling of these two plastics. The recycling of HDPE and PET is therefore modelled with 0% recycled content. The collecting, processing, and selling of the recyclable HDPE and PET are, however, included in the quantification. Users of the data contained in Tables 7.1 should be aware of the differences in the amount of recycled content involved in the quantification of materials recycling.

For each material, the costs in the first five columns of Tables 7.1 (A), (B) and (C) are then summed up to a total environmental cost that is used for the rating. The total environmental costs are listed in the last column of Tables 7.1 and in Table 7.2. Since paper is the only kind of packaging material that is compostable, the spaces listed under composting in Table 7.2 for materials other than paper and corrugated cardboard are closed for entry. The total environmental costs of composting paper packaging are labeled as “not available” because no cost data has been established for MSW composting yet. The column of reuse is filled with “n.a.” as well for the same reason of the lack of data, but as was discussed in the previous chapter, the environmental costs can be calculated using the equation established in this project once the required data become available. The same is true for composting. The costs for source reduction are zeros for all materials because the implementation of source reduction requires minimal cost and source reduction programs are typically targeted at reducing the consumption of materials in general without focusing on specific materials.

TABLE 7.2 Total External Costs for Various Waste Management Options

Material	Landfilling	Incineration	Recycling		Composting	Reuse	Source Reduction
			% Recycled Content	Cost			
Aluminum	3454.78	3236.90	100%	-184.54	___	N.A.	0
Ferrous	451.70	459.41	40%	407.58	___	N.A.	0
Glass	220.99	294.73	100%	136.24	___	N.A.	0
Boxboard	609.56	590.08	100%	264.15	N.A.	N.A.	0
Corrugated	529.38	505.22	100%	413.19	N.A.	N.A.	0
HDPE	929.95	757.72	0%	977.72	___	N.A.	0
PET	1587.36	1415.43	0%	1626.44	___	N.A.	0

N.A.: not available

Therefore, when source reduction is designated a cost of zero dollars, it applies to all materials.

THE SCORING FORMULA & THE RESULTS

The methodology for scoring the solid waste management options for their environmental impacts begins with designating the score of 1 to correspond to the zero costs from source reduction. A score of 1 means that the solid waste management method is of no cost to the environment. A score of 0 is assigned to correspond to the average of the total environmental costs of landfilling from the seven types of packaging materials. The use of a landfilling cost as the baseline cost supports EPA's hierarchy of solid waste management which prescribes landfilling as the least desirable management option. The total environmental cost of landfilling varies from a low of $220.99 from glass to a high of $3454.78 from aluminum. A mid-range cost should be chosen to compromise with the large range. Furthermore, landfilling is nonetheless one method of managing solid waste, therefore some credits should be given to its function in the system. An average cost of landfilling is thus a suitable choice for the baseline score. This average turns out to be $1112.

An alternative choice of a baseline cost is the weighted average of the total environmental costs of landfilling according to the volume of landfill space each type of materials occupies. However, while the volume of plastics or paper as a category in the landfills is estimated, the volume of individual types of plastics or papers are not measured. Therefore, this choice of a baseline cost cannot be calculated for now.

The relationship between the scale for costs and that for scores is illustrated by the two scale lines in Figure 7.1 below. The scores for costs between $0 and $1112 can be calculated by proportion.

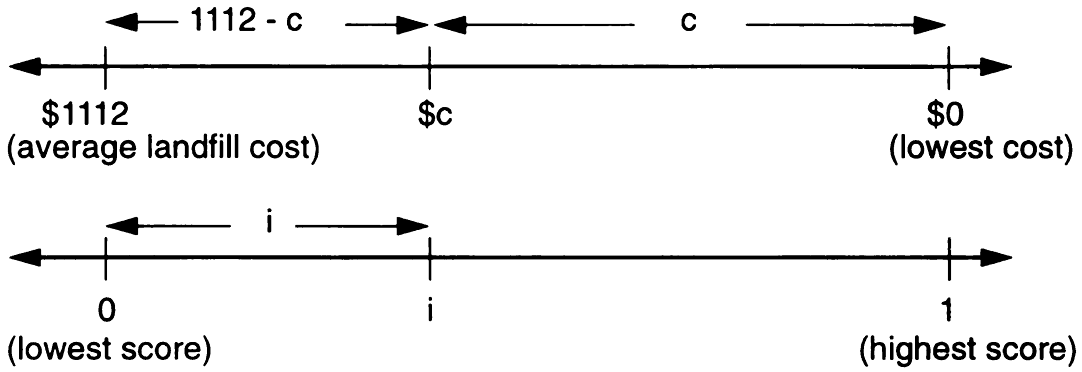


FIGURE 7.1 Scales for the Conversion from Costs to Scores

As shown in Figure 7.1, let 'c' be the total environmental cost of a waste management method for a packaging material and 'i' be its corresponding score in the scale of 0 to 1. Solving by linear proportion,

$$\frac{1112 - c}{1112} = \frac{i}{1}$$

$$i = 1 - \frac{c}{1112} \qquad \text{Equation 7.1}$$

Using equation 7.1, the total environmental costs listed in Table 7.2 are converted to scores listed in Table 7.3. One thing to note is that the recycling of aluminum into 100% recycled content costs a negative amount of $184.54. Since the scoring scale developed does not define negative costs, that dollar amount is assigned a maximum score of 1. Also, costs that are below the baseline costs of $1112 are given a score of 0.

TABLE 7.3 Ranking Scores of Solid Waste Management

Material	Landfilling	Incineration (Z)	Recycling ($X=X_1+X_2$)		Composting (Y)	Reuse (W)	Source Reduction
			% Recycled Content	(X)			
Aluminum	0	0	100%	1	—	1	1
Ferrous	0.59	0.59	40%	0.63	—	>0.63	1
Glass	0.8	0.73	100%	0.88	—	>0.88	1
Boxboard	0.45	0.47	100%	0.76	≥ 0.76	>0.76	1
Corrugated	0.52	0.55	100%	0.63	≥ 0.63	>0.63	1
HDPE	0.16	0.32	0%	0.12	—	>0.32	1
PET	0	0	0%	0	—	>0	1

Result Analysis

The scores listed in Table 7.3 clearly illustrate that when compared with landfilling and incineration, recycling is the most environmentally benign disposal option for almost all materials on the list except for HDPE and PET. The reasons for the low recycling scores of those two plastics is because they are evaluated with no recycled content. Incineration scores better than landfilling when the materials in question are capable of generating fuel values. Non-combustible materials like steel and glass give slightly higher scores in landfilling than incineration, indicating that it is better to landfill than to put them into incinerators. This result is understandable considering the fact that non-combustibles do not have any positive contributions to the incineration process but leave behind ashes that ultimately have to be landfilled. On the whole, the average environmental cost of landfilling of the seven kinds of packaging materials is $1112/ton, of incineration is $1037/ton, and of recycling is only $520/ton. All the above results generally support EPA's hierarchy that recycling is preferred to incineration, and incineration is preferred to landfilling.

Steel cans display a recycling score that is higher than landfilling and incineration even though only 40% of recycled content is technically allowed into the recycled production, and only 12% of which is post-consumer scrap. If the environmental benefits of steel cans being recycled into other steel products are also considered, the recycling score for steel cans should show an even higher value. Aluminum displays an outstanding benefit on recycling but scores poorly in landfill and incineration because aluminum is not combustible to benefit in incineration. In addition, landfilling or incinerating aluminum pays a high price by not re-utilizing such a valuable material but burying it. Even though both aluminum and steel are metals, steel scores better than aluminum in landfilling and incineration because the production of steel is less energy intensive than that of aluminum.

PET earns zero scores because its environmental costs are higher than the baseline cost. When examining the environmental costs of incinerating, HDPE and PET display slightly lower costs than the costs of landfilling and recycling. This is the effect of their high fuel values and low ash content that make burning the materials less expensive than landfilling or recycling. In addition, the light weight and bulkiness of plastics cause landfilling or recycling the materials to be more expensive because they take up more space during collection and final disposal and both the conventional and environmental costs are apportioned to the materials based on the volume they occupy. The overall environmental costs for HDPE and PET are almost the highest compared with other materials except aluminum. Both being plastics, the scores of PET are lower than those of HDPE because, while their conventional and environmental costs of disposal are very similar, the environmental cost of producing PET is over $500 more than HDPE. The fact that no recycled content is incorporated into HDPE or PET bottles also causes their recycling scores to be the lowest.

The ratings are constructed by basing on packaging materials rather than packages because it is assumed that by studying the impacts of the materials and by knowing the amount of the materials consumed, one can always determine their environmental impacts, regardless of the physical forms that they are in. Even though HDPE and PET do not seem to score well in the per ton material basis at this stage, they show a different picture when compared with other materials on the per package basis:

Table 7.4 Environmental Costs of Beverage Containers

I). 16 oz. Juice Containers

	wt. of material (oz) per container*	wt. of material for comparison	# of containers for comparison	Landfilling Cost	Incineration Cost	Recycling Cost
Glass	9.95	1 ton	3216	$221	$295	$136
HDPE	0.80	0.08 ton	3216	$74	$61	$78

II). Half Gallon Juice Containers

	wt. of material (oz) per container*	wt. of material for comparison	# of containers for comparison	Landfilling Cost	Incineration Cost	Recycling Cost
Glass	24.1	1 ton	1328	$221	$295	$136
PET	3.15	0.13 ton	1328	$207	$184	$212
HDPE	1.60	0.07 ton	1328	$65	$53	$68

III). 16 oz. Soft Drink Containers

	wt. of material (oz) per container*	wt. of material for comparison	# of containers for comparison	Landfilling Cost	Incineration Cost	Recycling Cost
Glass	8.45	1 ton	3787	$221	$295	$136
PET	1.10	0.13 ton	3787	$207	$184	$212

IV). 12 oz. Soft Drink Containers

	wt. of material (oz) per container*	wt. of material for comparison	# of containers for comparison	Landfilling Cost	Incineration Cost	Recycling Cost
PET	0.9	0.13 ton	4629	$207	$184	$212
Aluminum	0.6	0.09 ton	4629	$300	$281	- $16

* Source: Tellus, 1992

Table 7.4 compares soft drink and juice containers of various sizes made of different packaging materials. For each type of container listed, the environmental costs of different containers are compared by using the same number of containers for a given beverage content. The amount of material required to produce that given number of containers is multiplied by the per ton total environmental cost of the material. The results clearly show that plastics, PET or HDPE, use much less material for the containment of a given volume of a product, and hence impose less environmental impacts in landfill and in incineration. Recycling costs may be higher for plastic containers in some cases because no recycled content is incorporated in those containers. Nonetheless, PET and HDPE generally are more environmentally advantageous than other materials when they are compared on the per package basis.

Applications in the SWIPES Equation

For composting, since there is insufficient data for the quantification of its environmental impacts, the user of the SWIPES equation may choose to enter a score that is between 1 and the rate of recycling for the material in question. For example, if a composting score Y for corrugated cardboard is asked for, the user may enter a score that is at least equal to or greater than 0.63, which is the score for recycling corrugated cardboard. This judgment assumes that the user feels composting this material is at least as beneficial to the environment as recycling.

Similarly, the scores for reuse can be entered according to the user's judgment. If it is believed that reuse is superior to recycling, as it was prescribed on the EPA's hierarchy, one should input a value for W that is greater than the score for recycling. For HDPE, whose recycling score is lower than its incineration score at this time, the user may enter a reuse score that is higher than the incineration score. In the future, when data are available for package reuse, the user may use the equation established in Chapter VI to calculate its score.

One note has to be made on the scoring method: this scoring scheme is developed by using the average of the total environmental costs of landfilling as a reference and the scores of the packaging material for other disposal options are compared to this average cost. The scale of measurement is therefore confined entirely to those packaging materials evaluated in this thesis. No comparison is made between packaging materials and other non-packaging materials.

CHAPTER VIII
DISCUSSION AND CONCLUSIONS

Originated from the objective of determining the values of the coefficients W, X, Y, and Z in the SWIPES equation, this project has expanded into evaluating various environmental costs incurred during the life-cycle of different packaging materials. The scores assigned to the four coefficients are derived from those environmental costs. It turns out that those scores are not the only useful results; the environmental costs can also be used in making packaging and solid waste management evaluations. Making the costs of the relevant environmental impacts explicit allows us to identify the individual impacts that are more harmful to the environment, and hence seek improvements in those areas. The environmental costs of landfilling, incineration and recycling facilitate the comparison of integrated solid waste management schemes that implement a different mix of the three options. A scheme that is found to charge the least total environmental costs would be the least harmful to our environment. Moreover, packaging professionals may use the per ton environmental cost for each material of a package, as is done in Table 7.4., to evaluate the per package environmental impact.

It has to be emphasized that the scores generated in this project are determined by evaluating operations that meet regulatory standards. In other words, the scores are meant to be based on how various material production and solid waste management operations should perform, and not how the public perceives them to perform. The public's opinions on solid waste management methods may suggest a very different set of ratings, but those ratings are not objectively based on technological factors, nor appropriate operating conditions.

Accordingly, the environmental costs and the scores reflect the current status of the solid waste management in the technical perspective for packaging materials: they reveal that it is environmentally beneficial to incorporate more recycled content in material production. The results also show that some technical difficulties have to be overcome by plastics as a material before their wastes can be managed with environmental costs that are comparable to other materials. These include the problem of collecting bulky plastic containers for disposal and the technical feasibility of incorporating an appreciable amount of recycled content in making new containers. There have been suggestions of shredding the containers upon collection to reduce their in-truck volume. As mentioned before, a newly developed PET container being made of 100% recycled content has been favorably reviewed by the FDA. Technical advances in different aspects of solid waste management are emerging with time. The present environmental costs and scores become an imprint for the status of current solid waste management technologies. New costs and scores will be generated in the future. By then, data from different eras can be compared to reveal the evolution of solid waste management in our society.

FUTURE RESEARCH

A large amount of data from different aspects of packaging production and disposal is incorporated into the ratings. However, some of the required data are unavailable, therefore, leaving room for future improvements in the ratings. Data that should be included are the environmental discharges from MRFs, the environmental impacts from reuse and that of composting. Research should be conducted to obtain those data for packaging materials.

In this project, the evaluation of recycling is based on closed-loop recycling, which means aluminum cans are recycled to produce new aluminum cans. It is this guideline that makes steel can recycling and plastic container recycling score low. The former can only allow up to 40% recycled content and the latter does not have commercially available containers that contain any substantial amount of recycled content due to technical difficulties. If the environmental effects of diverting the recycled content of steel cans and plastics containers to products other than their own types are quantified, their recycling scores should be more comparable to those of other materials on the list.

Unlike glass, aluminum or steel, which can be recycled indefinitely, repeated recycling of paper continuously shortens the fibers, and repeated plastics recycling degrades the polymers. Eventually, these materials reach a point when they can no longer be recyclable and have to be disposed of by other means.The difference in the environmental effects of recycling indefinitely recyclable materials and the recycling of degradable materials should be studied.

During the course of this project, some of the costs involved have been changed. Market prices of some of the recyclable materials rose dramatically in 1994. Old corrugated boxboard was modelled at $25-30/ton in the Tellus

Packaging Study. Starting out with that same price range at the beginning of 1994, this boxboard soared to \$150/ton in July and lowered to \$70/ton in October of the same year (Steuteville, Dec. 1994). Aluminum marketed at nearly \$1400/ton in 1994, \$500/ton more than the price used in Tellus' study (Steuteville, Dec. 1994). Recyclable HDPE was sold for \$320/ton in 1994, compared to a sale of \$135-140/ton for all plastics in 1992 (Tellus, 1992). These new prices will significantly reduce the conventional cost of recycling, therefore producing higher ratings for recycling. The effects of these price changes are not accounted for in the current ratings because the revenues of recyclable materials cannot be easily singled out from the conventional costs of disposal that were calculated by the WastePlan model from the Tellus Institute. The WastePlan computer program asks for the input of a variety of data including the revenues from marketed recyclables. The program then adds an incremental volume equivalent to 15% of the entire waste stream to each material for calculating the marginal cost of managing each material. The program also adjusts the amount of investments required for the disposal system due to this large change in waste volume to be handled. The role that the material revenues plays in this calculation is therefore unclear without a careful study of the algorithm of the WastePlan program. Moreover, the relatively volatile markets for recyclables nowadays may produce fluctuating market prices, making it difficult to come up with a stable rating for recycling. An update on the ratings should be performed when the market prices for recyclables become more stable. Another cost that is known to have changed is the tipping fees for landfilling. The opening of some mega landfills as well as a lot of private landfills increased the landfill volume in some states and stirred up competition for disposal contracts. As a result, the tipping fees have decreased in some parts of the country (Breen, 1993). According to Tellus' methodology, the tipping fee of landfilling is apportioned to different materials based on their densities in the landfills. The quantitative effect of this change in tipping fees on each material is therefore unknown without re-calculating those apportioned costs.

In conclusion, the environmental costs and their respective scores presented in this thesis are only the foundation of a database for packaging waste management. This database is expected to be refined when more research is conducted to update the costs that build up this database. The list of scores is expected to expand in the future when more materials are evaluated and when data become available for solid waste management options such as reuse and composting.

APPENDIX

APPENDIX

CRUDE OIL PRICE CONVERSION

From *Monthly Energy Review (*EIA, 1992*)*,

Crude oil price: $14.78/barrel
(1 barrel equals 42 gallons for crude oil)

From *Concise Encyclopedia of Science and Technology* (Parker,1994),

Specific gravity of crude oils: 0.82 - 0.95

From *ASTM-IP Petroleum Measurement Tables* (ASTM,1953),

Specific Gravity 60/60°F	Pounds/US Gallon
0.82	6.8274
0.885 (average of 0.82 and 0.95)	7.3694
0.95	7.9113

Therefore, Crude Oil Price $14.78/42 gallons
= $14.78/(42 gallons x 7.3694 lb/gallon)
= $0.048/lb

DERIVATION OF PULPWOOD PRICES

i). From Tellus' quoting from the American Paper Institute, *Paper Paperboard Wood Pulp Capacity 1988-91:*

Origin of Wood Fiber:

Roundwood and roundwood chips	60.7%
Forest residues (logging residues)	4.7%
Manufacturing residues (sawdust, planar shavings and mill broke)	34.6%

ii). From Tellus:

a). Fiber input in Kraft pulp production:

Fiber Source	lb/BDT pulp	% of Total Source
Roundwood (48% water)	7329	75.6%
Forest Residue (48% water)	571	5.9%
Manufacturing Residue (Bone Dry)	925	
estimated (with water)	1800	18.6%

b). Fiber input in NSSC pulp production:

Fiber Source	lb/BDT pulp	% of Total Source
Pulpwood and chips (48% water)	4283	75.6%
Forest Residue (48% water)	334	5.9%
Manufacturing Residue (Bone Dry)	544	
estimated (with water)	1050	18.5%

iii). From *Pulpwood Prices in the Southeast,* (Howell, 1992):
Average pulpwood prices for roundwood and roundwood chips in 1992 Dollars/green ton):

Roundwood:	Softwood	$22.79
	Hardwood	$18.88
Roundwood Chips:	Softwood	$26.99
	Hardwood	$23.12

Average: $22.95/green ton

iv). Assumptions:

a). Since softwoods and hardwoods are usually combined in making paper but the exact mix proportion is not known, an average of softwood and hardwood prices is used.

b). No data is found on the prices of forest and manufacturing residues. No assumptions are made on those prices. Since the percentages of pulpwood used in paper making described in the Tellus study are higher than API's percentages, only pulpwood and pulpwood chips are used in estimating the raw material prices for paper making. No costs from forest or manufacturing residues are included.

ITEMIZED COSTS OF MAJOR MATERIAL INPUTS

ALUMINUM

Raw Material	lb/ton aluminum	Cost
Crude Oil	1020 lb	$48.96
(From Tellus: 1 ton Petroleum coke comes from 1.2 ton crude oil)		
Anthracite	40 lb	$0.68
Fluorspar	6 lb	$0.58
Bauxite	8999.5 lb	$76.5
Limestone	411.32 lb	$0.88
(From Tellus: 2.88 tons limestone for 1 ton of lime)		
Caustic Soda (from Salt)	245.9 lb	$0.49
Soda Ash	6.1 lb	$0.27

		Total: $128.36

RECYCLED ALUMINUM: No inputs from exhaustible natural resources. Therefore, zero cost.

STEEL

Raw Material	lb/ton steel	Cost
Iron Ore	172.64 lb	$2.69
Limestone	860.23 lb	$1.85
Coal (for Coke)	1516.41 lb	$15.95
(From Tellus: 2900 lb coal/ton coke)		
Feldspar	16 lb	$1.54
Iron Ore Concentrates	1718.54 lb	$17.26

		Total: $39.29

RECYCLED STEEL

Raw Material	lb/ton steel	Cost
Iron Ore	143.52 lb	$2.24
Limestone	789.60 lb	$1.70
Coal (for Coke)	1260.63 lb	$13.26
Feldspar	16 lb	$1.54
Iron Ore Concentrates	1429.67 lb	$14.36
Caustic Soda	26.88 lb	$0.05

		Total: $33.15

GLASS

Raw Material	lb/ ton glass containers	Cost
Sand	1184 lb	$8.08
Limestone	363 lb	$0.78
Feldspar	168 lb	$3.10
Soda Ash	394 lb	$17.57

		Total: $29.53

RECYCLED GLASS: No inputs from exhaustible natural resources. Therefore, zero costs.

UNBLEACHED COATED FOLDING BOXBOARD

Raw Material	lb/ ton board	Cost
Roundwood and chips	6302.94 lb	$72.33
Limestone (for lime)	76.78 lb	$0.17

		Total: $72.50

RECYCLED FOLDING BOXBOARD: Inputs for rosin, alum, starch and coating only. No pulpwood needed. Therefore, zero costs.

LINERBOARD

Raw Material	lb/ ton board	Cost
Roundwood and chips	6533.80 lb	$74.98
Limestone (for lime)	79.60 lb	$0.17

		Total: $75.15

RECYCLED LINERBOARD: Inputs for rosin, alum, starch and salt cake only. No pulpwood needed. Therefore, zero costs.

CORRUGATING MEDIUM

Raw Material	lb/ ton corrugating m.	Cost
Roundwood and chips	3814.01 lb	$43.77
Soda Ash	267.15 lb	$11.92

		Total: $55.69

RECYCLED CORRUGATING MEDIUM: Inputs for rosin, alum, starch and sodium sulfite only. No pulpwood needed. Therefore, zero costs.

HDPE

Raw Material	lb/ ton HDPE.	Cost
Crude Oil	1340 lb	$64.32
Natural Gas	16,500 cu. ft	$24.59

		Total: $88.91

PET

Raw Material	lb/ ton PET.	Cost
Crude Oil	1876 lb	$90.05
Natural Gas	7540 cu. ft	$11.23

		Total: $101.28

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