89-024/NJ

Assessment of Impacts of Production and Disposal of Consumer Packaging on the Environment

NJ DEPE Contract P31152

VOLUME I

prepared for:

New Jersey Department of Environmental Protection and Energy

prepared by:

Tellus Institute
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May 1992

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CONTENTS AND COPYRIGHT NOTICE

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The five Tellus Institute Packaging Study reports are printed in two volumes. Due to its length, Report #2 is printed separately in Volume 2. The other four reports are printed together in Volume 1, beginning with the Executive Summary (Report #5).

The Executive Summary is also available as a separate volume.

89-024/5NJ

Assessment of Impacts of Production and Disposal of Consumer Packaging on the Environment

NJ DEPE Contract P31152

Report #5:
Executive Summary

prepared for:

New Jersey Department of Environmental Protection and Energy

prepared by:

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Dozens of people have worked hard over the past three years to make this study possible. At Tellus Institute, John Schall first conceived of the idea for the study, wrote the original proposals, and created and directed the Tellus side of the Council of State Governments project structure, through which we received the majority of the research support. Dr. Frank Ackerman directed the separate New Jersey and E.P.A. projects, and served as principal investigator and technical director for the entire research effort. Karen Shapiro played a major role throughout the study, including coordination of the production process analyses, and creation and application of the health effects ranking system, among other contributions. Monica Becker provided detailed knowledge of paper industry products, technologies, and emissions. Mark Rossi explored countless intricacies of plastics industry production processes. Todd Schatzki turned masses of disparate data into a coherent model of the New Jersey solid waste system. Irene Peters reviewed the economics literature and current European research, and helped develop the approach to monetary valuation of pollutants. Gretchen McClain analyzed the packaging products in the case studies, and did much of the final revisions of the reports. Other present and former Tellus staff members who worked on the study include Marc Breslow, Jeannette Herrmann, Elizabeth Titus, and Anne Weaver.

Among the sponsors of the study, our colleagues at New Jersey DEPE persevered and offered detailed, thoughtful criticism throughout a process that became much larger and longer than they had anticipated. While we could not incorporate all of their advice into the final products, we benefitted greatly from their active involvement. Athena Sarafides put in an astonishing amount of effort from beginning to end of the study, accompanied at different times by Mike Winka, Mary Sheil, Jeanne Herb, Michelle Crew, and other NJ DEPE staff.

Current and former E.P.A. staff including Terry Dinan, Bill O'Neill, Paul Kaldjian, and Mike Flynn provided much-needed support and encouragement. Without the two E.P.A. grants we received, this study would have been impossible. At the Council for State Governments, Steve Brown remained unflappably calm and good-natured as his once-simple coordinating task grew to include more confusion and controversy than he had ever dreamed of. Individual sponsors and reviewers, too numerous to mention here, provided helpful suggestions and criticisms at a series of Advisory Committee meetings. Our critics will find some of their comments on earlier drafts have been incorporated in the final reports.

As valuable as all these inputs have been, the findings and conclusions expressed in this study are those of Tellus Institute alone. None of the sponsors are responsible for the judgments and calculations expressed in the final reports. And, as explained in the text, these reports are only the end of this project, not the end of the research agenda it suggests. We hope to be involved in refining and extending this area of research for years to come.

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1. INTRODUCTION: TOWARD A SCIENTIFIC BASIS FOR PACKAGING POLICY

Until recently the United States economic environment encouraged the proliferation of packaging, and ignored the social costs of packaging production and disposal. Raw materials, energy, and waste disposal were cheap, and pollution prevention and resource conservation were not on the public agenda. Under these conditions, manufacturers and retailers emphasized consumer "convenience," based in part on ever-more elaborate packaging. After a single, frequently brief use, the final user could simply discard packaging at little or no cost, and without concern for the impact on the environment.

Today the situation is changing. Waste disposal is no longer cheap, resource depletion is widely recognized as a major problem, and pollution prevention has moved to the top of environmental agendas. In this context, the social costs of producing and discarding packaging are becoming increasingly apparent, and interest in packaging policy is spreading rapidly.

Packaging accounts for approximately 30% by weight and 50% by volume of the solid waste stream. Its useful lifetime is often very short: it travels rapidly from factory to market to garbage can. Moreover, it appears to be shifting away from readily recycled materials toward ones which are difficult or impossible to recycle. Thus many states and communities are considering policy options to reduce the level of packaging or alter the mix of packaging materials.

The discussion of packaging policy, however, is frequently lacking in firm scientific foundation. There are important differences in approach and emphasis, even among those who seek to reduce or modify packaging use for environmental reasons. Is it more important to eliminate the use of the most harmful materials, or to achieve overall reduction in packaging volume? Is one material always better than another -- is glass, for example, always environmentally preferable to plastic -- or does it depend on the end use and local conditions? Is there a preferred material, or ranking of materials, for all uses?

Two popular policy approaches implicitly embody opposite conclusions about the true costs of packaging materials. On the one hand, proposals to ban a particular substance such as polyvinyl chloride, polystyrene, or even all nonrecyclable plastics, seem to assume that specific materials are the problem. That is, they assume that the true social or environmental costs of packaging made from these materials are much higher than the costs of other packaging alternatives. In this case, the goal of public policy is to reduce use of the most harmful packaging materials.

On the other hand, proposals to place a per-unit (per-pound or per-container) fee on all packaging, regardless of material, seem to assume that the sheer bulk of packaging is the problem. Implicitly, this approach assumes that the true costs of packaging are similar for all materials. In this case, the goal of public policy is source reduction and recycling of all materials, across the board.

These and related questions motivated the research on packaging materials carried on by Tellus Institute over the past several years. In June 1989, Tellus Institute (formerly Energy Systems Research Group) began a study for the New Jersey Department of Environmental Protection and Energy, entitled "Assessment of Impacts of Production and Disposal of Consumer Packaging on the Environment." The goal of the study was to assess the lifecycle environmental impacts of different packaging materials, in order to help provide a scientific basis for formulation of packaging policy.

In view of the immense scope of the questions raised in that study, additional research was required. Tellus also obtained support for related packaging research from U.S. Environmental Protection Agency, and from the Council of State Governments (representing a number of states and industry groups). The entire round of studies is now complete, and we are reporting on its final results.

1.1 Our Approach to Packaging Analysis

Many other studies of packaging problems have been conducted. Both industry and government agencies continue to sponsor additional research on packaging impacts. Why, then, was the Tellus research needed? Our research embodies a unique approach, one that we believe should be more widely adopted, refined, and elaborated in future studies. Specifically, our approach differs from other published research in four principal respects.

First, we aimed to provide a **comprehensive comparison** of the full range of packaging options. Rather than focusing on a few specific materials, we analyzed all major materials used in consumer packaging today: aluminum, glass, steel, five types of paper, and six types of plastic. Where data was available, we contrasted virgin and recycled production of the same material. The goal was to provide a framework and a database to support the analysis of any packaging choice or policy proposal, allowing evaluation of all options on a comparable basis.

Our commitment to comparability meant that we did not use data sources that were available for only one or a few materials. Although this was at times frustrating, the alternative would have been far worse from a methodological standpoint. Selective inclusion of noncomparable data could have created the appearance of "rewarding" or "penalizing" some materials due to differential data availability.

Second, while other studies have analyzed specific packages, such as soft drink containers, in great detail, our focus was **at the level of materials rather than packages.** We made this choice for two reasons, one a research hypothesis and the other an observation about policy formation. It is our hypothesis that the predominant lifecycle impacts of packaging are due to the choice and quantity of major materials used, rather than to the specific package being made; we hope to test this view in future research. If this hypothesis is correct, then it is appropriate for life-cycle analysis to focus, as we have done, on materials rather than specific packages.

Independent of this hypothesis, the focus on materials is also a matter of practicality in policy formulation. Development of comprehensive packaging policies on a product-by-product basis is impossible: there are too many distinct products and packages in use today to carry out detailed individual life-cycle analyses of every individual packaging choice. An attempt to perform a full-scale study of every package in the marketplace would be neither affordable, for research sponsors with limited funding, nor useful if completed, for policymakers with limited time to read and respond to environmental research. In short, the public sector needs to address packaging problems at the level of broad categories such as major material usage, rather than at the level of detailed dissection of single-product options.

Third, we made a fundamental decision to **rely solely on public sources of information.** We used government databases, and other information available in the public domain. To be included in our study, data had to be accessible to all researchers, and had to provide comparable coverage of all the materials and industries in our study.

This innocuous-sounding principle is actually at odds with common practice in the field. Many "life cycle assessment" studies of packaging and other products rely heavily on proprietary data, often supplied at the discretion of the company that sponsors the research and makes the product being studied. Such studies fail to pass a basic test of scientific method: their results are not reproducible by independent observers, since the underlying data are not available. The problem is compounded when the results of the study favor the products or plans of the sponsoring company; in such cases the line between scientific research and marketing literature begins to blur.

One clear conclusion from our study is that the public data on environmental impacts of manufacturing is far from ideal. Problems of time lags before publication and gaps in data coverage appeared to worsen as government statistical agencies suffered budget cutbacks in the 1980's. The data limitations encountered in our study strongly suggest the need for improvements in the quality and timeliness of environmental data collection.

Fourth, we attempted to **evaluate the relative importance of different environmental impacts**, producing one of the most difficult and controversial portions of our study. We believe that quantitative evaluation of different impacts is crucial to the goals of our research. If (as frequently happens) two packaging options cause dozens of pollutant emissions, with each option causing more of some emissions and less of others, how should the resulting emissions listings be evaluated? If policymakers read such a study and come to a conclusion about which option is better, then some implicit standard of comparison is being applied. Our goal was to make that standard explicit, and to encourage discussion of the appropriate bases for comparison of environmental impacts.

Other studies have appeared to avoid this difficult step, often simply adding up the pounds of pollutants resulting from a product lifecycle. However, this approach does not escape the need for comparative evaluation of environmental effects. Rather, it imposes one simplistic standard of comparison, namely "a pound is a pound is a pound." If the total weight of emissions is all that matters in the end (if, for instance, it is the only quotable summary statistic emerging from a study), then the study has assumed that a pound of any one pollutant is exactly as bad as a pound of any other.

We are sure that, pound for pound, certain pollutants are worse for health and the environment than others, in some cases by many orders of magnitude. So we concluded that another approach was needed. Drawing on techniques that have been widely used and debated in energy regulation, we "priced" many pollutants at the cost of currently required control measures. This approach assumes that the social cost of pollution must be at least as great as the costs society is willing to impose for pollution control. For criteria air pollutants and greenhouse gases, we simply adopted control cost values that have been developed in the process of energy regulation and analysis.

For many other major pollutants, however, no such off-the-shelf valuations were available. To address a broader range of emissions, we ranked a wide range of toxic and carcinogenic substances emerging from packaging lifecycles on the basis of their impacts on human health. This allowed us to calculate the quantity of total "health hazard" resulting from production and disposal of each packaging material.

We then placed a monetary "price" on health hazards, based on our conclusion that society has demonstrated a willingness to pay about $1,600 per pound of lead removed through air pollution controls. This allowed us to express the health hazards in dollar terms; the price for each toxic or carcinogenic substance is based upon its relative health impact, compared to the impact of lead. Note that a change in the $1,600 figure would change the absolute levels of all hazard "prices", but would not affect the *relative* ranking of the various health hazards.

Many aspects of the study, particularly the health effects ranking and related pricing issues, are still experimental. During the course of this study, U.S. EPA and the Society of Environmental Toxicology and Chemistry (SETAC) have been developing a technical framework for life-cycle assessment (LCA). That framework is still evolving; at the current state of development, LCA encompasses three components:

- *Life-cycle inventory*, the quantification of the energy and raw material inputs and environmental releases associated with each stage of production

- *Impact analysis*, or assessment of the impacts on human health and the environment associated with energy and raw material inputs and environmental releases quantified by the inventory

- *Improvement analysis*, the evaluation of opportunities to reduce energy and material inputs or environmental impacts.

Our inventory report on packaging material production processes coincides with the first step of LCA. The health effects ranking and related pricing is a first attempt at carrying out an impact analysis since, at this time, no single methodology exists for performing an impact analysis.

We believe that we have advanced the state of knowledge on packaging materials, raised a number of previously hidden issues that require discussion, and given the best answers possible based on the available data and resources. We are well aware that we are offering the first word, not the last, on these critical topics. We hope that our current study will be followed by further refinement and development of research tools and publicly available databases in this area. We look forward to further discussion of both the policy implications and the technical questions flowing from our study.

1.2 Assumptions and Limitations of the Study

Despite the broad scope of this study, there are a number of important assumptions and limitations which the reader should be aware of. The results presented in later sections, and in the detailed reports, should be understood in the context of the issues raised in this section.

We intentionally set out to analyze the **marginal, rather than average, impacts** of production and disposal of packaging materials. That is, our goal was to identify the change in social costs resulting from an increase or decrease in the use of materials. For policymaking purposes, the average impact of a material is of limited importance. Most policies will result in a marginal change in the quantity of one or more packaging materials; thus the significant impacts to study are the marginal impacts of quantity changes. In many cases, we were not able to identify the difference between average and marginal impacts, and used average impacts for lack of better information. In two very important cases, however, we used calculations of marginal costs and impacts.

One case is the analysis of disposal impacts of packaging materials. Our study of disposal impacts is based on New Jersey data; we used actual population, waste stream composition and quantity, and many waste management system characteristics in our modelling. However, **our disposal analysis departs from current average New Jersey conditions in two regards.** First, it is based on information available from state agencies and published sources in mid-1990. New Jersey solid waste planning assumptions have changed since that time, so that the assumed level of incineration and various other parameters in our report are no longer up to date. This point is elaborated in the preface to the disposal cost report, written by New Jersey DEPE.

Second, independent of recent changes in data inputs, our disposal impacts were never based on *average* New Jersey waste management practices. Rather, they reflected our estimates of the expected marginal impacts of additional packaging materials. That is, the disposal impacts are the incremental burdens resulting from an assumed increase in collection, recycling, and disposal to handle an assumed increase in the state's packaging waste stream. More detail on these calculations is presented in Section 4 of this report, and in the full report on our disposal impact calculations.

In another area of analysis, our inventory of production impacts makes frequent use of data on energy production. All materials manufacturing processes use significant amounts of energy, and the impacts of energy production are an important part of the lifecycle impacts. Identification of the precise sources of energy -- the types of oil wells or power plants, for example -- for each manufacturing process would be impossibly complex, and of limited value: since imported and domestic crude oil, for example, are perfect substitutes for each other, manufacturers can switch effortlessly between them based on changing market conditions, with no change in production technique. The same applies to the choice between hydroelectric, nuclear, and coal-fired electrical power.

Therefore, we **assumed uniform marginal energy sources** for all production processes. For oil and natural gas extraction we assumed that the marginal source is onshore domestic U.S. production (in part due to lack of data for offshore and foreign production). For electricity production we assumed that the marginal source is a coal-fired power plant.

In the case of electricity, the distinction between marginal and average impacts is particularly significant. The aluminum industry, for example, uses a substantial amount of hydroelectric power. Indeed, its location was historically determined, in part, by the availability of cheap hydropower. Thus the *average* impacts of aluminum production as it exists today include the impacts of hydropower. However, the supply of hydropower is quite limited -- and because it is so cheap, it will be fully utilized regardless of fluctuations in the aluminum industry. The *marginal* source of electricity, which will increase or decrease in response to changes in aluminum production, is coal-fired power. The same applies to other industries and regions of the country: while many sources of power are used, coal-fired plants are the marginal source of additional baseload generation in most of the country most of the time.

It would be a mistake to try to "customize" the results of the study for a particular state by using information on that state's electric utilities in place of the marginal coal plant impacts. Materials production is coordinated and integrated on a nationwide, if not multinational, basis, so state-specific energy impacts would not be appropriate. Moreover, coal is often the dominant marginal power source in a state, even when other fuels appear important on an average basis.

The **health effects ranking**, introduced in the "Methods and Case Studies" report, **is a new, and still experimental, approach** to evaluation of multiple pollutants. We have not yet tried the many modifications and sensitivity analyses that could be performed on this ranking system, nor have we developed rankings based on other measures of environmental damage; we hope to pursue this area of research in future studies. However, this does not mean that it is prudent to proceed without any pollutant ranking. When the emissions from production and disposal range from dioxin to carbon dioxide, from particulates to vinyl chloride, it is absurd to add unweighted pounds of pollutants. Our ranking of a broad class of pollutants based on human health effects, and other pollutants based on values adopted by energy regulators, captures one enormously important dimension of variation in environmental risk; it is a much more reasonable basis for policy analysis than no ranking at all.

Two assumptions closely related to the health effects ranking should be noted. We based our ranking on controlled emissions of air and water pollutants; when no controlled emission figures were reported, we assumed no emissions occurred (or, equivalently, that controls are 100% effective). And when emissions of a pollutant were reported but the health effects were unknown, we made no assumption about its effects. That is, we assumed no toxic or carcinogenic effects from pollutants unless we found published information on their effects. Therefore, the impacts associated with packaging material production and disposal may actually be underestimated.

The **monetization of health effects, based on the $1,600 per pound price for lead,** is a controversial approach, although drawing on techniques that have frequently been adopted in analyses of energy production. Based on the pollution controls currently required for air emissions of heavy metals, it appears that society is willing to impose control costs equivalent to $1,600 per pound of avoided lead emissions. We then extended this price to other pollutants in proportion to their health hazards. If control costs were proportional to the hazards being controlled, then society should be equally willing to spend $3,200 per pound to reduce emissions of a substance twice as dangerous as lead. Our calculated health effects costs show what happens when this logic is extended to the full range of hazardous emissions.

As with the health effects ranking, it would be valuable to test and compare other approaches to monetization of environmental impacts. However, some approach is needed, and in fact is implicit in any use of environmental research for policy purposes. If, based on a list of pollutant emissions, policymakers decide that one type of pollution prevention or control measures is acceptable while another more expensive type is not, then they have placed an implicit price on the emissions. Our approach makes this pricing process explicit, and more accessible for debate.

Our "lifecycle" analyses omit the stages of package forming, filling, and transportation. Similarly, we do not include the requirements for shipping containers (cartons, pallets, wrappings, etc.) used for wholesale transportation of products. Our focus on materials, rather than individual packages, necessitates these limitations. It would be a valuable extension of our method to explore "generic" impacts in these areas, e.g. for blow-molding plastic containers, printing and forming paperboard boxes, etc., and to calculate impacts of production and disposal of major shipping containers. Transportation impacts per ton-mile via truck, rail, and barge could also be calculated; a difficulty in this area is that the distance and mode of transportation for a product may be much more changeable than the underlying production technologies.

Our analysis of emissions is constrained by data availability. This problem, which appears in a number of areas, turned out to be most critical in two respects. First, **the nearly complete absence of systematic data on industrial solid waste** forced us to drop this category from our analysis. Unlike air and water emission, industrial solid waste is not regulated when it is produced. Rather, its disposal is regulated (and therefore its quantity is reported) only if it is classified as hazardous, or disposed of off-site. The great bulk of industrial solid waste, however, is not classified as hazardous, and is disposed of on-site. The monetary costs of industrial solid waste disposal are already included in product prices; pollutant emissions or even simple tonnage data for industrial waste disposal are almost entirely unreported. Therefore, despite our original intention to study industrial solid waste, we could not include it in our quantitative analysis.

Second, we used the latest publicly available emissions data in all cases. But due to the limited public resources devoted to environmental data collection, this means that **some of our emissions data are several years old.** Many industry representatives state that they are making progress in reducing pollutant emissions. When it becomes possible to verify these reductions from public data sources, then our methodology would automatically assign lower environmental costs to the materials involved. The pollutant costs calculated in this study can be read in two different ways: both as an estimate of environmental costs imposed by current production processes, and as an agenda identifying priorities for pollution reduction in the future. The highest-cost materials in our study are the ones that pose the greatest hazards *if produced with existing technologies* -- and are by the same token the highest priorities for technological change and pollution prevention.

Data limitations constrained the scope of the study in yet another way. **We were unable to model recycled plastics production**, due to the lack of data on this newly emerging technology. As the plastics recycling industry matures and data becomes available, it will be important to extend our analysis to encompass this industry and contrast it to virgin plastics production.

Finally, one frequently discussed factor related to recycling is intentionally absent from our calculations. We analyzed product lifecycles, including the environmental impacts of production and disposal; however, we **did not include any charges to virgin production processes or credits to recycling processes based on virgin materials depletion.** For example, our analysis of oil-based plastics production includes the actual environmental impacts of oil refineries and other production stages, but does not include any charges to reflect the fact that a nonrenewable resource is being depleted by this production process. Similarly, our analysis of virgin and recycled paper production reflects pulp and paper mill impacts, but does not include any recognition of virgin paper production impacts on habitat destruction, old-growth forest cutting, or other environmental issues raised in connection with logging (other than direct emissions from logging equipment). Inclusion of such factors, important as they may be, would have been beyond the scope of an already very broad study. Readers who are concerned about virgin materials depletion should consider that factor in addition to the analysis presented here, when evaluating materials recycling options.

1.3 Summary of Principal Results

Our calculations of production and disposal impacts, summarized in more detail in Chapter 5 below, can be presented on two bases: impacts per ton of packaging material, or per unit of delivered final product.

1.3.1 Per-ton Impacts

Per-ton impacts are presented in Tables 3.1, 3.2, and 3.3 of our "Methods and Case Studies" report, reproduced below at the end of this summary. The following discussion is based on those results.

On a per-ton basis, the highest-cost material is polyvinyl chloride (PVC) at $5,288 per ton, followed by virgin aluminum at $1,963 and polyethylene terephthalate (PET) at $1,108. All other materials have per-ton impacts valued between $127 and $620. Plastics other than PVC and PET fall between $537 and $620; all paper products, steel, and recycled aluminum range from $247 to $443; and virgin and recycled glass have the lowest impacts, at $157 and $127 per ton, respectively.

These impacts combine three separate categories: environmental impacts of production, conventional (monetary) costs of disposal, and environmental costs of disposal. Of the three categories, the last is clearly least. The environmental impacts of disposal in new, state-of-the-art disposal facilities, using our valuation techniques, range from $1 to $5 per ton, generally 1% or less of the total impacts. Both monetary costs of disposal and environmental impacts of production dwarf the environmental impacts of disposal. The widespread perception of waste management facilities as significant polluters may be based in part on older, poorly designed or controlled facilities; and it may reflect the fact that waste management is necessarily geographically dispersed, and causes noticeable land use, aesthetic, odor, and other impacts which were not included in this study. Production impacts, in contrast, are concentrated in a smaller number of industrial locations, and are often less visible to casual observation.

Environmental impacts of production are larger than (combined monetary and environmental) disposal impacts for all except the lowest-impact materials -- recycled and virgin glass, and virgin corrugating medium. For PVC and aluminum, disposal impacts are less than 10% of the total; for all remaining materials, disposal impacts are at least 23% of the total. Disposal impacts vary less by material than production impacts; in particular, all paper, steel, and plastic disposal impacts are between $112 and $255 per ton, a range of just over 2 to 1. Plastics are at the upper end of this range, with higher disposal impacts due to their low density and resulting collection costs. Aluminum has the lowest net disposal impacts due to high recycling rates and revenues.

The highest production costs generally reflect one or a few key pollutants. For PVC, the high impacts are due to the emissions of vinyl chloride monomer, a known carcinogen; for PET, the elevation in price over other plastics is due to antimony emissions in the final stages of production. For virgin aluminum the high impacts reflect particulate, nitrogen oxides, and sulfur dioxide emissions, the latter two due to the consumption of electricity (and consequent power plant emissions) in aluminum production.

In several cases, our results allow comparison of virgin and recycled material impacts. Since disposal costs are the same for virgin and recycled materials, it may be more useful to compare production impacts alone. For aluminum, recycled production impacts are only 16% of virgin, since recycled production avoids the high energy and environmental costs of virgin aluminum production. For both boxboard and linerboard, recycled production impacts are 49-50% of virgin; for glass the ratio is 65%. In the case of steel cans, the difference appears insignificant, with recycled impacts at 97% of virgin; this occurs because the production requirements for steel *cans* allows very limited scope for use of recycled materials. (In other uses, recycled steel is more important, and more distinguishable from virgin production.)

Finally, for corrugating medium the seemingly anomalous result is that recycled material impacts are 220% of the corresponding virgin material impacts. The available data on recycled production showed substantial impacts from deinking old paper products, due primarily to the heavy metal content of the inks. This was more hazardous than the relatively clean virgin production process -- virgin corrugating medium is the lowest-impact paper product in our study. In this case, the anomaly of higher recycled impacts may naturally disappear as less hazardous inks are adopted.

Additionally, it might be noted that linerboard and corrugating medium are combined, in proportions of about 2 to 1, to make corrugated cardboard. For the two-material combination, our figures show recycled cardboard having lower production impacts than virgin.

1.3.2 Per-package impacts

Results on a per-package or per-delivered product basis follow the per-ton results in some but not all cases. We weighed and analyzed packages used to deliver five products: soft drinks, juice, fast food hamburgers, microwave dinners, and hardware (screws and nails). Numerical results are presented in Table 4.13 of our "Methods and Case Studies" report, reproduced below at the end of this summary. The following discussion is based on those results.

Hardware packaging made from PVC was a clear loser, reflecting the extremely high per-ton cost of this material. For non-PVC hardware packages, fast food, and microwave dinners, the materials used are relatively similar in total impacts (ranging only from $247 per ton for recycled boxboard to $620 for polystyrene, a ratio of 2.5 to 1). Therefore, in most cases, the lighter the package, the lower the impacts. In our food and hardware categories, we found that different items on the market differ quite substantially in package design and packaging weight per unit of final product. The variation in packaging weight is often more important than the choice of materials.

For juice and soft drinks, package weights remain of central importance, but the results are more complex. The materials used for beverage containers range from recycled glass, with impacts valued at $127 per ton, to PET, with impacts of $1,108 -- a range of almost 9 to 1. There are two factors that complicate the analysis. First, larger containers use less material per ounce of beverage, simply based on geometry. Thus impacts per ounce of beverage are lower for larger containers, independent of material type. Second, packages made from materials with lower impacts per ton, such as glass bottles, are heavier than competing packages. If a bottle made purely from PET weighed one-ninth as much as a recycled glass bottle of the same size, then the bottle weight differential would exactly offset the differential impacts per ton of material; as a result, the two bottles would have the same impacts per ounce of beverage.

To control for the size factor, and focus on the choice of materials alone, it may be helpful to compare each container to a similar-sized glass bottle. Glass is a useful standard since it is the material used in the widest range of beverage size categories. In all categories, virgin glass impacts are roughly 120% of recycled glass impacts; we will use recycled glass as the standard for comparisons with other materials. Our results allow comparison of the impacts per ounce of beverage for five categories of containers: two sizes for soda (16 ounce and 10-12 ounce) and three for juice (46-64 ounce, 16 ounce, and smaller than 16 ounce). For larger soda containers, PET is the only material in use, so no comparisons are possible.

Several materials have impacts between 80% and 120% of the recycled glass levels, on a per ounce of beverage basis. This group includes PET bottles and steel cans in all categories, and virgin aluminum soda cans. Only one package we examined, small virgin aluminum juice cans, was significantly above 120% of the recycled glass impact level. On the other hand, several materials had impacts below 40% of the recycled glass level; these included 100% recycled aluminum, aseptic packages ("juice boxes"), paperboard, and HDPE juice containers. Since aluminum cans are not typically made from 100% virgin or recycled metal, weighted averages of the two impact levels should be considered: 50%

recycled aluminum, for instance, has impacts of 56% of the recycled glass level for soda cans, and 78% of the recycled glass level for small juice cans.

Since our study did not include package forming processes, we did not include the impacts of the adhesives or bonding processes used in making aseptic packages; our impact calculation is based solely on the bulk materials -- paper, aluminum, and plastic -- used in this multi-layered package. Given the complexity of the aseptic package, it seems reasonable to suspect that package-forming impacts are of relatively greater importance in this case than in the alternatives. Thus we have likely understated the environmental impacts of aseptic packages relative to the other beverage packages in our study. However, even if the omitted package-specific impacts are equal to the calculated bulk material impacts included in our study, aseptic packages will have less than 80% of the recycled glass impacts (as do paperboard, HDPE, and recycled aluminum).

One potentially important beverage container option, *refillable* bottles, was not included in our study. The glass bottles we considered were single-use containers, whether made from virgin or recycled glass; the plastic bottles were likewise single-use containers made from virgin plastic. There is very little recent U.S. experience with individual consumer use of refillable bottles (they are largely confined to bar and restaurant use); thus it is difficult to project a realistic number for average return trips and repeated uses before loss or breakage, an essential parameter in the analysis of refillables.

While it is not appropriate to draw sweeping or definitive policy conclusions from this limited set of case studies, we can suggest the tentative implications emerging from our results. For the non-beverage packaging options we examined, PVC packages have much higher impacts than alternative materials; among other materials, the lightest-weight package, per unit of delivered end product, is generally the lowest-impact product. Technologies and designs that allow lower package weights will achieve lower total impacts; weight reduction will often be more important than changes in types of materials.

For the beverage options, a number of common materials, including glass, steel, virgin aluminum, and PET, had broadly similar impacts per ounce of beverage. Another set of materials had lower impacts per ounce, including recycled aluminum, HDPE, paperboard cartons, and aseptic packages. Here the choice of materials appears significant, as well as the potential for weight reduction.

1.4 Overview of Earlier Reports

This summary report is the fifth and final report on the Tellus packaging research. The detailed results of that research are presented in the four previous reports:

- Literature and Public Policy Review

- Inventory of Material and Energy Use, and Air and Water Emissions from the Production of Packaging Materials

- Marginal Cost of Handling Packaging Materials in the New Jersey Solid Waste System

- Impacts of Production and Disposal of Packaging Materials - Methods and Case Studies.

Each of the four reports is summarized in turn in the following pages.

2. LITERATURE AND PUBLIC POLICY REVIEW

As the first task in our study of packaging, we conducted a thorough literature review to assess the existing trends in packaging legislation, and the state of knowledge of environmental impacts of packaging and the economics of packaging taxes and charges.

2.1 Packaging Legislation

With solid waste disposal costs skyrocketing, states and communities are considering and enacting legislation to limit the amount of packaging in municipal solid waste. In 1988, 2,000 solid waste bills were introduced in state legislatures, with 300 of them specifically addressing packaging issues: 206 on plastics; 77 on biodegradability; and 11 on paper packaging. Since that time the pace of solid waste legislation has, if anything, continued to increase.

Types of legislative initiatives which address the environmental impacts of a package include:

- source reduction measures
- measures encouraging or mandating recyclability of products recyclable
- deposit fees
- variable garbage collection fees.

Two important aspects of each packaging initiative, which affect political feasibility as well as environmental and economic efficiency, are: the point(s) in the product's lifecycle targeted by the measure and the scope (number of products affected) of the initiative. In general we found that voluntary initiatives targeted consumers, while financial incentives targeted both consumers and manufacturers, and regulatory initiatives targeted manufacturers.

In terms of political feasibility, voluntary initiatives were most palatable to industry, and regulatory initiatives were least, with financial initiatives in the middle. However, state and local governments have found it easier to pass regulatory initiatives -- primarily conditional and outright bans -- than to pass financial initiatives.

Packaging legislation was examined at four levels, state and local initiatives, regional initiatives, federal initiatives, and international initiatives. States often introduce voluntary measures as a part of broader solid waste management initiatives (as in Florida and Maine). Of the financial initiatives examined, deposit laws are in effect in 10 states (California, Connecticut, Delaware, Iowa, Maine, Massachusetts, Michigan, New York, Oregon, and Vermont).

Variable fees, which concentrate on changing consumer habits by connecting increasing disposal costs to the amount of garbage residents place at the curb, are implemented on the local level by municipal governments. Seattle, Washington was the first major city to adopt such a system, although many smaller communities have instituted variable fees as well.

Tax credits and exemptions were typically designed to reduce a manufacturer's operating costs, in order to encourage a socially beneficial action. One instance of a tax credit targeted at packaging was Rhode Island's "Act Promoting the Use of Paper Bags" which exempted all biodegradable or returnable bags, boxes, wraps or containers from state tax. However, the more prevalent type of measure is the investment tax credit, which offers manufacturers a state or federal tax deduction for investing in a specific production process, such as manufacturing paper from waste paper.

Packaging fees target all packaging products with the specific aims of reducing packaging waste, increasing recycling, and raising revenue. State legislatures considering this measure include Vermont, Oregon, New York, Connecticut, California, Minnesota, and Massachusetts.

Regulatory source reduction initiatives include bans, mandatory reusable bottles, product specifications, and minimum warranty requirements. Materials and packages which have been targets for bans include polystyrene, polyvinyl chloride, the plastic/aluminum can, non-degradable plastic retail bags, multi-material food and beverage containers, and non-degradable six-pack connectors. An example of a wide-ranging conditional ban is a Minneapolis law banning all packaging sold in retail establishments which is not degradable (degradable plastics are not exempted), returnable or recyclable.

Regional initiatives are discussed, for example, in reports from the Council of North Eastern Governors (CONEG) Source Reduction Task Force. The CONEG Task Force recommends measures such as establishment of a Northeast Source Reduction Council, and adoption of preferred packaging guidelines. On the federal level, much of the action on packaging issues has centered around efforts to reauthorize the Resource, Conservation, and Recovery Act of 1976 (RCRA). On the international level, many European nations have already enacted national regulations governing the use of packaging materials, especially non-returnable/non-reusable beverage containers.

2.2 Packaging and the Environment

We reviewed six major studies of the environmental impact and energy requirements of various packaging materials. They were selected for their comprehensive coverage, generally dealing with many materials and/or packaging types. Two studies dealt with the energy requirements for packaging production; the remainder examined the environmental impacts of beverage and non-beverage packaging.

The first study evaluated the energy required for manufacturing many types of plastic materials, including packaging. It then compared this requirement to the energy required for manufacture of substitute materials. It found that replacing plastic products with non-plastic alternatives would increase U.S. energy consumption by 834.2 trillion Btu annually. Several examples depict how the use of plastics reduces energy consumption. For instance, the use of plastics in automobiles leads to lighter, more fuel efficient vehicles.

The objective of the second study was to determine the amount of energy consumed in the production of paper, glass, steel, aluminum, and plastic packaging. Comparing energy requirements to make a unit of final product, plastic bags, milk containers, and meat trays were found to use less energy than their paper alternatives, while paper cups used less energy than plastic. Many other specific comparisons were examined as well. Recycling of glass, aluminum, and steel was found to provide energy savings ranging from 25% in the glass industry to 84% in the aluminum industry.

The third study examined the environmental impacts of non-fluid foods packaging. The study was based on 16-year time series data for 14 non-fluid food packaging types and for 10 non-fluid food groups. The authors found an increase in the total weight of food packaging per capita during the study time period, but a slight decrease in the weight of packaging per dollar of food expenditure or per pound of food delivered. There was also a decrease in the amount of minerals industry output used in these types of packaging.

The remaining studies evaluated the impacts of production of beverage containers, the most intensively studied type of packaging. Two reports examined soft drink packaging and one looked at milk packaging.

The two soda studies, both done by Franklin Associates, arrived at differing conclusions. The earlier study found that plastic (PET) bottles produce lower environmental impacts than other packages in all impact categories including energy. However, refillable glass bottles became environmentally preferable if they could achieve a return trip rate (average number of uses before loss or breakage) of 4 to 6. The later Franklin study found that the refillable glass bottle used the least energy and had the lowest air and water emissions, while PET bottles minimized solid waste.

The milk study presented the most detailed look at production processes and life cycles, raising many issues addressed in our study. In an examination of refillable containers, the milk study found that 50-trip plastic (HDPE) bottles had lower environmental impacts than 20-trip glass bottles, and that the largest containers had the lowest impacts per gallon of milk delivered.

A problem with many of the studies was that proprietary databases were used, and without access to these data it was not possible to verify or reanalyze results. Another problem was that all but one were completed in the 1970s or early 1980s, when research funding in this area was more readily available. Rapidly changing technologies and patterns of packaging use have rendered many of the detailed findings in these studies obsolete. However, the methods of analysis were helpful in formulating our approach to the impacts of packaging production.

2.3 The Solid Waste Product Charge

Finally, our literature review examined one of the most extensive past debates in packaging policy, the discussion of a solid waste fee or product charge. This idea was proposed in Congress and researched in some depth in EPA-sponsored studies in the 1970's, but then received much less attention as policy discussion focused elsewhere in the 1980's. (While there now is an emerging literature of more recent discussion of the same topics, little or none of it was available when we began our study and did the literature review in 1989-90.)

The rationale behind a solid waste product charge is straightforward. Every material product enters the solid waste stream at one time or another, and when it does it contributes to the monetary and environmental costs of managing this waste stream. The cost is imposed on society at large; it is what economists call an external cost (the agents whose activity gives rise to the cost do not have to pay it). If these agents were forced to pay the true costs of their activities, economic theory holds that they might find it profitable to alter their behavior. Thus, a policy imposing taxes or fees on those who produce externalities should alter their behavior in a socially optimal way. The solid waste externality arises with the production of materials that will eventually require disposal. So every material product should bear a disposal charge reflecting its contribution to the monetary and environmental costs of solid waste management.

Following a brief description of the early Congressional discussion, we reviewed two shorter studies. One emphasized the importance of accurate cost estimates of the magnitude of the solid waste externality. The other presented simplified calculations suggesting that the externality might be a very small fraction of product prices.

We then turned to the two major studies (Bingham et al. and Miedema et al.) which examined the effects of policies concerning packaging of consumer products. Both studies employed elaborate econometric models of production processes, designed to estimate factor substitution in response to price changes. Both also developed extensive databases on use of materials by industry, to allow estimation of their econometric models. While we have a number of criticisms of these models, they are both far more ambitious and theoretically developed than other studies in the literature.

The Bingham study modeled the effects of four policies concerning consumer product packaging: regulation requiring the use of recycled materials in packaging; a tax on the total weight of packaging, a tax on the weight of virgin material in packaging, and a per-unit tax on rigid containers. The study found that the tax on virgin materials used in packaging produced the greatest reduction in waste disposal, followed by recycled content regulation. Both reduce disposal primarily through boosting recycling rather than through source reduction. The per-container tax caused the greatest source reduction, possibly because the modeling of the per-container tax involved higher tax rates than other alternatives. The tax on total weight of packaging would likely produce similar levels of source reduction, if it were boosted to an equivalently high tax rate.

The Miedema study considered the effects of a tax levied on the weight of virgin material in packaging. It used somewhat newer data than the Bingham study, and a number of changes in model specification designed to reflect the potential for material substitution more accurately. However, in retrospect it is remained severely constrained by the limited knowledge and experience with secondary materials available in the late 1970's. The study found that a tax on virgin materials in packaging, at the equivalent of $80 per ton in 1990 dollars, would cause about 3% source reduction and 4% additional recycling of packaging materials.

In summary, the two major studies of the 1970's found taxes to be preferable to regulations, and estimated that taxes could cause modest amounts of source reduction and recycling. However, the models are both dated, and many of their detailed assumptions are now obsolete. Far more information is now available, for example, about costs and performance of secondary materials.

3. INVENTORY OF MATERIAL AND ENERGY USE & AIR AND WATER EMISSIONS FROM THE PRODUCTION OF PACKAGING MATERIALS

In order to evaluate the environmental impacts of producing and using packaging materials, we developed models of the production process for each material. We determined the energy and material inputs and the environmental emissions associated with each production process; wherever possible, we developed separate data for production processes using virgin and secondary (recycled) materials.

This summary begins with a description of our methodology, and then briefly presents each of the production processes examined in our study.

3.1 Methodology

For the production of packaging materials, we examined a number of steps in the production process, including:

1. raw material extraction;
2. raw material processing; and
3. packaging material manufacture.

In each step, energy and raw material inputs into the production process were calculated, as were air and water emissions. Several environmental effects associated with the production of these different packaging materials were not considered in this study. For instance, occupational health and safety impacts were not quantified, nor were other effects such as habitat loss and deforestation. Data on solid waste generation, as explained in Section 1, was usually not available. The exclusion of such impacts does not imply any judgement that such impacts are trivial or nonexistent, but rather that the scope of the study had to be limited.

A fourth step in the production model is the transportation of raw materials used in packaging manufacture. This step has not been included because transportation data were available for some packaging materials -- glass, aluminum, and steel -- but were sparse or totally lacking for paper and plastics.

Some packaging materials and intermediate products may be produced in more than one way. In this study, we tried to determine trends in production, and have used data on the predominant production process for each material.

Materials production may yield outputs other than the one of interest in this study. For example, production of coke (a material used in steelmaking) from coal results in byproducts such as coke oven gas, tar, and light oils, along with the coke. The byproducts can either be used at the facility or sold. The energy value of these byproducts was credited to the production process and the amount of energy used in the coking process, along with pollutants generated by the process, were apportioned to coke and the by-products.

An important step in the study was defining the boundary conditions for each stage in production. We applied a "one step back" rule: inputs and emissions were counted for processes one step back from the major stages of packaging production, but not for two processes back. For example, the inputs and emissions associated with mining the major raw materials and ores were documented in the study; the inputs and emissions associated with producing the mining equipment were not included.

The production of packaging materials requires the use of many additives, as well as the major raw materials. In applying the "one step back" rule to additives, we calculated the inputs and emissions for production of the additives, but did not go back any farther. That is, we did not model the production of inputs to the process of producing additives.

Furthermore, many minor additives are used in the production of packaging materials. Studies conducted in the wake of the 1970's energy crisis detailed the energy and material inputs associated with production processes, including the production of additives. As a result, there is an abundance of energy data, while in many cases environmental data is lacking. In this study, while we have listed the energy requirements for such additives, we included environmental emissions only if the additive comprised at least 5% of the total material inputs into production.

The goal of this study was to produce a model that calculates the per unit impacts associated with changes in packaging usage. Thus, the inputs into the model are the pollutant factors associated with a unit of packaging, expressed as pounds of pollutant per unit weight of packaging. To compile the data necessary for this study, we drew on the knowledge of government, industry, environmental, and academic experts. Data sources included government, academic, and industry reports and papers on raw materials extraction, manufacturing processes and pollutants, and energy inputs for each packaging material examined. The government data sources included reports from the U.S. Department of Commerce, U.S. Department of Energy, and the U.S. Environmental Protection Agency (EPA). These data sources were used to determine the pounds of pollutants per unit weight of material produced.

We have presented two emission factors associated with each production step -- uncontrolled emission factors (i.e., emissions measured upstream from a pollutant control device) and controlled emissions (i.e., emissions measured downstream, or after, a pollution control device). While the latter emission factor determines the quantity of pollutant released into the environment, during the course of our research it became apparent that much more data are available for uncontrolled than for controlled emissions. Additionally, it is important to remember that the wastes from emission control processes, such as scrubber sludges, must still be disposed; hence the true environmental burden is greater than the controlled emissions would indicate.

3.2 Production Processes

3.2.1 Paper and Paperboard

Paper and paperboard can be produced by a number of processes using both virgin and secondary raw material. We examined eight production processes for different types of paper and paperboard, five from virgin and three from secondary materials. For each process we identified their material and energy requirements, and air and water emissions. The categories of paper and paperboard covered in this study are as follows:

A. Virgin Paper and Paperboard

Bleached kraft paperboard
Unbleached, coated folding boxboard
Linerboard (for corrugated cardboard)
Corrugating medium (for corrugated cardboard)
Unbleached kraft paper

B. Recycled Paper and Paperboard

Folding boxboard
Linerboard
Corrugating medium

The production of paper and paperboard from virgin raw materials involves a number of steps. For all virgin paper products, the first few steps are essentially the same: wood harvesting, transport to the mill, storage and conveyance within the mill, washing, debarking, and chipping. The subsequent steps, pulping, bleaching, and final paper or paperboard manufacture, depend on the desired final product.

Pulping separates the wood into fibers; paper and paperboard are formed when these fibers are reunited in a mat formation. The quality of the pulp and subsequently, the quality of the paper or board required, determines the pulping process used. Pulp is naturally brown, so for some end uses the pulp is bleached to obtain a whiter product. Paper and paperboard is then formed from the pulp.

Several different pulping processes are used in the paper industry, with significantly different environmental impacts. Most paper packaging materials are produced with the kraft pulping process, which is used to make three-quarters of the pulp produced in the U.S. The process relies on sulfates and other chemicals to break down the wood into fiber; the characteristic odor of a kraft mill is caused by the emission of sulfur compounds from many points within the mill.

Bleached kraft paperboard, one packaging product made from kraft pulp, is converted into cartons for milk, frozen foods, cosmetics, blister packs, and many other packaging items. Another product, unbleached coated folding boxboard finds applications in a number of food and non-food product categories, such as cereal and cracker cartons, beverage carriers, dry soap containers, hardware, and toys. A third product made from kraft pulp, unbleached kraft paper, is used in bags, shipping sacks, wrapping paper, and other packaging and non-packaging products.

Virgin linerboard is usually made from kraft pulp and is used as the facing material of a corrugated container or in solid fibre boxes. Corrugated cardboard is a "sandwich" made from a fluted corrugating medium placed between two layers of linerboard.

Corrugating medium is produced primarily from neutral sulfite semi-chemical (NSSC) pulp. The NSSC process relies on a combination of chemical and mechanical methods of separating the wood into its constituent fibers.

There are several sources of air and water pollutants emitted from paper and paperboard production. The paper industry is an energy intensive industry, requiring large quantities of steam for pulping and drying of paper and paperboard, and electricity for running pumps, refiners and paper machines. The industry gets its energy from a number of different sources, including oil, gas, coal, and combustion of wastes from pulping and wood processing. The combustion of these materials, either on-site or at an off-site utility, results in air emissions such as particulates, SO_x, NO_x, and lead.

Both pulping processes examined in this study require the use of sulfur compounds, and give rise to air emissions of such compounds. Pulping is also a source of water pollutants although water and chemicals may be recovered, thereby reducing water pollution. Perhaps the most publicized source of pollution associated with paper and paperboard production is from the bleaching process. Chlorine, a key ingredient in the most common bleaching processes, reacts with the pulp to form a variety of chlorinated organic compounds. These compounds, including small quantities of highly toxic dioxins and furans, have been found in water effluents from bleached kraft mills. However, other stages of manufacturing, including the final manufacturing of paper and paperboard, produce important effluents as well.

3.2.2 Recycled Paper and Paperboard

Recycled wastepaper is used in the production of many categories of paper-based packaging. Within the broad category of paperboard, wastepaper is used in the production of the two components of corrugated shipping containers (linerboard and corrugating medium), and recycled folding boxboard is used to package food, soaps and other products.

The production of paper and paperboard from secondary materials requires two principal steps, repulping the wastepaper, and final paper and paperboard production.

The separation of wastepaper and paperboard fibers for reuse can generate large volumes of pollutants in the waste water. When secondary fibers are reused, they become shorter, weaker and more difficult to capture, so that more fibers escape to the effluent stream. Many mills have succeeded in reducing the volume and solids content of their discharges by recycling their water in a closed water circulation system.

One major source of water pollution from secondary paper production are the inks and coatings found on wastepaper. Toxic metals such as cadmium, zinc, aluminum and chromium are used in some inks in trace amounts. These materials are liberated in the pulping process and are either incorporated in the final product or become part of the mill's effluent stream. Efforts are being made in the U.S. and in other countries to promote the use of non-toxic inks. As inks and coatings on paper products become less toxic, the environmental hazards of secondary paper production will be diminished.

Paperboard materials manufactured from recycled fibers requires less energy than from virgin fibers. For unbleached coated boxboard and linerboard, a 44% energy savings is realized; for corrugating medium there is a 27% energy savings.

3.2.3 Aluminum

Major packaging uses of aluminum include foil and cans. The production of virgin aluminum involves three major processes:

Bauxite ore mining and processing
Alumina production
Aluminum production

Bauxite, the only commercially valuable aluminum ore, is mined by the open-pit method. The overburden is removed and draglines strip the bauxite deposit, which is then transported to processing plants. Overburden is replaced and the site is restored. Next the ore is crushed, washed, and screened at a processing plant to eliminate clay minerals and dirt. The ore is then kiln-dried in a rotary kiln. The majority of energy required for mining and processing is consumed by this step.

Following the bauxite processing, alumina, an intermediate product, is produced by the Bayer process. This process recovers the pure alumina from bauxite ore by removing impurities. Pressure and steam heat along with caustic soda are used to dissolve the alumina present in the ore.

The third major stage of production involves the electrolytic reduction of alumina into aluminum and oxygen. This yields molten aluminum, to which alloying materials such as iron, silicon, magnesium, and manganese can be added. The aluminum is cast into ingots, which are shipped to users who will then melt them for reuse.

Most of the direct pollutants from this three-stage process are emitted during the production of alumina and aluminum. The entire process is also very energy-intensive, so energy use (including electricity generation) is an important indirect source of air pollutants. Water pollution arises in part from the use of wet scrubbers for controlling air pollutants.

New aluminum containers can be made using up to 100% used beverage containers. When aluminum is produced from secondary materials, the bauxite mining and processing, alumina production, and aluminum production stages are all eliminated. Instead, new production stages are required: shredding, delaquering (heating the metal to remove outer coatings), and melting of aluminum containers. The energy savings from recycled production are substantial; it requires 8.32 MMBtu to produce one ton of new can sheet stock from used beverage cans, whereas it requires 208 MMBtu to produce one ton of molten aluminum from virgin materials. Due to the large energy savings, air pollution associated with energy production is decreased. However, some air pollution is generated from melting the used aluminum.

3.2.4 Glass Containers

The manufacturing of glass containers involves four major steps:

Raw material mining and processing
Mixing of raw materials
Melting and refining of raw materials
Forming of molten glass and post-forming of glass containers

Glass production as described in this study includes the production of containers. Unlike other packaging materials, glass container manufacturing is an integral part of glass production. Once molten glass is produced, it is immediately formed into a container.

The major raw materials for virgin glass production include silica, limestone, soda ash, and feldspar. Mining of these minerals includes excavating the ores, crushing to reduce size, and various operations to remove impurities. In addition to these minerals, some cullet (very small pieces of used glass) is used in the production of virgin glass.

Upon delivery to the glass plant, raw materials are crushed and prepared into batches. The mixed batches of raw materials are continuously fed into furnaces which melt the raw materials at extremely high temperatures. This stage consumes a large percentage of the total energy used in the production of glass, and generates much of the air and water pollution associated with virgin glass production.

The molten glass must then undergo a refining process where the glass is freed of any crystalline materials and gas bubbles. Following the refining process, the glass is cooled to approximately 2000° F in order to be ready for forming. The molten glass is sliced into lumps, which are called gobs. Each gob will be formed into a single glass container.

Once a gob is formed, it is cooled and shaped. After formation, a tin oxide coating can be added to enhance the adhesion of other coatings and to strengthen the container. Finally, the glass containers are transferred to an annealing oven to relieve internal stress in the glass caused by uneven cooling of the glass during the forming process.

Glass is 100% recyclable, and can be repeatedly recycled. Glass plants typically use cullet -- broken or crushed glass -- as a primary raw material in the manufacture of glass containers. Using cullet saves an estimated 23% of the energy required for virgin glassmaking; this reflects the fact that cullet melts at a lower temperature than the virgin raw materials. In addition, glass recycling eliminates the air pollution associated with raw material mining and processing.

3.2.5 Steel

While steel's primary uses are in heavy industrial products and consumer durables, 3% of U.S. steel production in 1989 was used to make containers for food and beverages. The production of steel involves several processes. These include:

1. Iron ore mining and processing
2. Limestone quarrying and lime formation
3. Coal mining and processing
4. Coke formation from the coal
5. Sintering
6. Blast furnace iron making
7. Steel making
8. Steel forming

The first three processes are carried out at the mines where the various ores originate; the remainder of the processes occur at integrated steel plants. While there are two types of furnaces used for steel making -- the basic oxygen furnace (BOF) and the electric arc furnace -- only the BOF can be used to manufacture the quality of steel required for steel can production. Therefore, our analysis focused solely on the BOF furnace type.

BOF steelmaking, starting from coke formation (step 4 in the above list), proceeds as follows.

Coal is converted to coke in the byproduct coke oven plant, so named because the useful byproducts of coking are recovered and either used at the facility or sold. The coke is then used in the blast furnace where it serves both as a fuel and as an oxygen-reducing agent.

Next, sintering, an optional step in the steel making process, may occur. Sintering is a form of recycling of steelmaking process wastes. It uses iron ore fines (particles too small for ordinary use), fine coke known as coke breeze, limestone, mill scale and recovered blast furnace flue dust; these are converted into an agglomerated product, sinter, which can be charged to a blast furnace. The sinter plant is typically located near the blast furnace.

The processed iron ore is charged into the top of the blast furnace, a large cylindrical tower, along with the coke and lime. As these materials travel down the furnace, their temperature increases due to the burning coke gases. In the top part of the furnace oxygen is removed from the ore by the coke gases. Lower down in the furnace, lime reacts with the coke and ore impurities to form slag. At the bottom of the furnace the molten iron, known as pig iron, forms a pool. Periodically the molten iron and slag are removed.

Steel scrap is loaded into the BOF along with the molten iron. After an initial heating period, lime and fluorspar are added through a chute to form a slag layer with the impurities. After this, molten steel is removed; it is at this stage that alloying materials may be added. The molten steel is then poured into ingot molds or into a continuous casting machine.

The production of coke and the production of pig iron in the blast furnace are the stages that cause most of the pollution associated with steel production. During the coking operation, approximately one-quarter of the weight of coal is liberated as gases and vapors; over 2000 different chemicals are found in these gases. Fugitive emissions from coke ovens are difficult to control.

Post-consumer steel food and beverage cans are 100% recyclable and may be recycled back into packaging cans. Once steel cans are collected, post-consumer steel cans are sold to detinning companies or to steel mills. The cans may be detinned, to remove the fine coating of tin that covers steel food and beverage cans. Or, depending upon their intended use, the steel cans may be used directly without detinning. A number of steel mills and minimills (mills equipped with electric arc furnaces to produce steel from scrap) buy steel can scrap directly. The material is then blended with other steel scrap so that residual levels of tin are kept within specification limits.

Recycled steel has traditionally been an essential raw material of the steel making process. In the BOF, 20%-35% scrap is technically required to be added to the hot metal from the blast furnace. While much of this scrap is in-house scrap generated at the steel mill, it can be supplemented with steel can scrap. No more than 40% scrap can be used in a BOF due to technical limitations.

Reflecting these limitations and patterns of scrap usage, we modelled "virgin" steel production as using 72% pig iron and 28% in-house steel scrap, since these were typical values for BOF operation. We then modelled "recycled" steel production as using 60% pig iron, 28% in-house steel scrap, and 12% detinned steel cans, since this was the maximum achievable level of scrap usage. These modelling assumptions explain the relatively small difference in our calculated energy use and air and water emissions for virgin vs. recycled steel. In our model of recycled steelmaking, can scrap displaced only one-sixth of the pig iron used in our virgin model (i.e., reduction from 72% to 60% pig iron), due to the limits imposed by the BOF technology.

Other parts of the industry, particularly minimills using electric arc furnaces, can use up to 100% steel scrap. These mills make other steel products such as reinforcing bars for construction, but cannot make the steel sheet needed for new cans. Thus our recycled model reflects what is possible for the specific purpose of can manufacturing, not what is possible for the steel industry as a whole.

3.2.6 Plastics

Plastic packages are produced from many different types of plastics. Therefore, our analysis distinguishes several leading plastic types, and identifies their separate uses in packaging. On the broadest level, plastics are divided into two major categories: thermosetting plastics and thermoplastics. Thermosetting plastics (thermosets) are cured, set, or hardened into a permanent shape. Once cured, thermoset plastics cannot be remelted or restored into a flowable state. Thermosets are mainly used in durable goods and account for 17% of U.S. plastics sales. Thermoplastics, on the other hand, do not cure or set; they soften to a flowable state when reheated and therefore can be remelted and rehardened. Thermoplastics account for 83% of all U.S. plastics sales, including almost all packaging uses.

The two largest end users of the 58 billion pounds of plastics sold in 1988 were the packaging and construction industries. The packaging industry was the largest single consumer, accounting for 25% of all plastics sales, more than 14 billion pounds, in 1988.

Plastic packaging products are made from at least 15 polymers. However, seven principal polymers account for 94% of all packaging products, or more than 13 billion pounds in 1988. These polymers, which were analyzed in our study, are:

High-density polyethylene (HDPE)
Linear low-density polyethylene (LLDPE)
Low-density polyethylene (LDPE)
Polypropylene (PP)
Polyethylene terephthalate (PET)
Polystyrene (PS)
Polyvinyl chloride (PVC)

Other polymers utilized by the packaging industry (but not examined in our study) include ethylene vinyl acetate copolymer, epoxy, polycarbonate, styrene acrylonitrile, phenolic, urea, cellulosics, and acrylonitrile-butadiene-styrene.

The major stages in the lifecycle of a plastic package are:

1. Raw materials extraction (natural gas and crude oil),
2. Natural gas processing and petroleum refining,
3. Organic chemicals production,
4. Plastics polymerization,
5. Plastics molding and forming,
6. Package filling by user,
7. Product purchase by consumer, and
8. Disposal.

The multi-stage process of plastics production is examined in some detail in our study, because important air and water emissions occur at many stages of the process. The early steps, which are common to many or all plastics, are of great importance; emissions in the polymerization and later stages (when each type of plastic is handled separately) are far from the whole story of the environmental impacts of plastics production.

The life of a typical plastic package begins when crude oil and natural gas are extracted from the earth. Coal has historically been used as a raw material in plastics production, but currently the primary raw materials are natural gas and crude oil. Because natural gas and crude oil are a soup of hydrocarbons, they are first separated and then broken into their constituent parts before being used to produce plastic polymers. The principal products from natural gas processing that are used in plastics are ethane and propane. The principal products from crude oil refining used in plastics are liquified petroleum gases (LPG--propane and butane), naphtha, and gas oil.

Once ethane, propane, LPG, naphtha, and gas oil are isolated from natural gas and crude oil, they can be processed into the organic chemical feedstocks that are used by the plastics industry: ethylene, propylene, benzene, and paraxylene. Further processing of the feedstocks into intermediate or derivative products is required for the production of some plastics. PET, PS, and PVC are all manufactured from plastic feedstock derivatives. A feedstock or intermediate chemical is used to produce "monomers," the building blocks in the formation of plastic resins.

Plastics are formed by linking individual molecules together, a process called polymerization. The single molecules used in this process are called monomers; they are formed into long chains (of the same molecule) known as plastic polymers, sometimes with the aid of catalysts, carrier fluids, or emulsifiers. Before, during, and after polymerization, additives are mixed into the plastic to impart specific qualities such as flexibility, color, or resistance to ultraviolet light degradation to the final product. When the additives are blended with the product after polymerization, the process is called "compounding." Finally, after compounding, the plastic resin is molded and formed into a final product. Additives may still be added as the resin is formed into packaging products through various methods, including extrusion, injection molding, blow molding, and others.

There are four general types of polymerization processes:

bulk:	monomers and the necessary reactants are combined in a reactor where the chemical reaction occurs.
emulsion:	takes place in water with the help of an emulsifier (soap) and a water-soluble initiator.
solution:	a chemically inert solvent provides a carrier fluid for the chemical reaction.
suspension:	uses a liquid non-solvent as a carrier fluid, producing bead-like polymer particles that are "suspended" in the fluid.

We first analyzed the common processes of oil and gas extraction, processing and refining, and feedstock production. We then turned to analysis of intermediate chemical, monomer, and polymer production for each of the major resins.

3.2.7 High Density Polyethylene (HDPE) and Linear Low Density Polyethylene (LLDPE)

HDPE, LLDPE, and LDPE are all polyethylenes. The density, molecular structure, and other properties of each polyethylene determines its packaging end use. LDPE and LLDPE are used primarily in film applications, while HDPE is used primarily to make containers. As a group, polyethylenes accounted for 8.5 billion pounds, or about 60% of all plastic sales used for packaging in 1988.

The leading packaging plastic is HDPE, with 1988 packaging sales of 4.2 billion pounds, of which containers accounted for 84%. Containers made with HDPE include milk and water jugs and liquid clothes detergent bottles. The other large packaging use for HDPE, accounting for 13% of 1988 packaging usage, is plastic film; HDPE is used to make plastic grocery sacks, for example.

Packaging film is the principal market for LLDPE, accounting for 93% (or 1.4 billion pounds) of 1988 packaging sales. HDPE and LLDPE compete against each other in some film markets such as grocery sacks. However, the principal polymer which LLDPE competes against in film markets is low density polyethylene (LDPE).

Because HDPE and LLDPE have very similar molecular structures, they can be manufactured at the same facility using the same processes. Both polymers are produced in low pressure reactors through three different processes:

PET manufacture is a complex process involving a number of intermediate chemicals. PET is formed by polymerization of the monomer bis-hydroxyethyl terephthalate (BHET), which in turn is formed by mixing ethylene glycol with either dimethyl terephthalate (DMT) or terephthalic acid (TPA). Lacking data on the less common TPA process, we analyzed the impacts of PET manufacture in the more common DMT process.

The process begins with the production of the two intermediate chemicals: ethylene is transformed into ethylene glycol, while paraxylene and methanol are used to make DMT. (An alternative process makes DMT from TPA; we found little available data on this process.) In monomer production, ethylene glycol and DMT are purified, and then combined in a reactor, in the presence of a catalyst, yielding BHET and methanol. Polymerization of BHET into PET occurs in a different reactor.

The principal air pollutants from PET production include volatile organic compounds (VOC), sulfur oxides, and nitrogen oxides. Antimony emissions from the final stages of PET production are also important.

3.2.11 Polystyrene

In 1988, sales of polystyrene ranked fifth among all plastics in the U.S. In sales for packaging products, polystyrene ranked third behind HDPE and LDPE. When one thinks of polystyrene, images of Styrofoam (a registered trademark of Dow Chemical) coffee cups and fast food hamburger packages (clamshells) often come to mind. Polystyrene, however, is used to produce many products besides those made from foam. Examples of polystyrene packaging include cottage cheese and yogurt containers and clear plastic lids on take-out salad containers.

Styrene is polymerized, in the presence of heat and a catalyst, into polystyrene. Styrene can be polymerized by bulk (mass), solution, suspension, or emulsion techniques. Bulk and suspension techniques are the ones most commonly used, and the ones examined in this study. Different grades of polystyrene are produced from these processes, depending on the desired end use. The three principal grades are general purpose (also called crystal), high impact, and expandable bead.

General purpose polystyrene is a clear and brittle polymer. For increased strength, flexibility and durability, another compound -- either polybutadiene (a flexible polymer) or styrene-butadiene rubber (synthetic rubber) -- is added to styrene, to create high impact polystyrene. Expandable polystyrene bead is formed by expanding general purpose polystyrene with a blowing agent, producing a polymer with good insulation and impact resistance qualities.

The principal set of pollutants from polystyrene production are volatile organic compounds (VOC). In mass polymerization, the VOC emitted is styrene.

3.2.12 Polyvinyl Chloride (PVC)

PVC has many applications in construction and manufacturing; only about 9% of all PVC is used in packaging. Of the packaging usage, about half is in the form of bottles and containers; most of the remainder is film, with smaller amounts used for coatings and closures.

There are four methods for producing PVC; one of them, the suspension method, accounts for 83% of all PVC output. Therefore, our data refer solely to the suspension method. In the multi-stage production process, ethylene and chlorine are initially combined to yield ethylene dichloride. This chemical is then combined with additional ethylene and oxygen to produce vinyl chloride (VC) monomer and hydrochloric acid. Finally, the VC monomer is polymerized into PVC.

The environmental impacts of PVC production are widespread. Air emissions from the suspension production of PVC result from each step in the production process; the VC monomer is by far the most important and harmful pollutant involved. Of the total VC emissions, 38% are fugitive emissions, which are of great concern from a worker exposure perspective. Sources of fugitive emissions include agitator seals, pump seals, valves, leaks in polymerization equipment, dryer vents, and bagging.

Wastewater emissions result from many production steps, including the stripping process, the centrifuge, the dryer, and reactor washing. Due to the volume of wastewater generated by the suspension method, water emissions are an important factor in PVC production.

3.2.13 Plastics Recycling

Post-consumer plastics recycling is in its infancy in the U.S. Of the approximately 14 billion pounds of plastics consumed by the packaging industry in 1988, roughly 250 million pounds, or 1.8%, were recycled. The collection infrastructure, separation technology, and end-use markets are all in the early stages of development. Because the plastics recycling effort is so new, publicly available data on the energy requirements, material requirements, and environmental impacts of plastics recycling operations are virtually non-existent. The only data we found were for water effluent from polystyrene and HDPE recycling plants. We found it impossible to create a complete set of data on recycled plastics process emissions comparable to those for virgin plastics production.

4. THE MARGINAL COST OF HANDLING PACKAGING MATERIALS IN THE NEW JERSEY SOLID WASTE SYSTEM

Reduction of the amount of packaging materials in the waste stream has been difficult in part because neither the producer nor the consumer is aware of the disposal cost of the product. Quantifying the disposal impacts of packaging is a first step toward developing policy measures affecting packaging in the waste stream. In our third report, we addressed this need by investigating the economic impacts of various packaging materials on the New Jersey solid waste system.* That system includes collection programs and facilities to process recyclables and to dispose of waste through incineration, landfilling or transfer to out-of-state facilities.

To measure disposal impacts we estimated the marginal cost to the solid waste system of handling an additional ton of each material. The marginal cost is the change in the cost of the solid waste system when the amount of a particular material in the waste stream is changed. The variation in the costs of managing different packaging materials results from differences in their characteristics, such as density, recycling rate, recycled material revenue, energy content, and ash content.

4.1. Modelling Methodology

In brief, our methodology for producing marginal costs involved modelling the New Jersey solid waste system, the pathway of each packaging material through that system, and the additional costs imposed on the system by a hypothetical increase in each material.

First, we developed a model of the cost structure of the New Jersey solid waste system, based on existing and planned new facilities and programs as of mid-1990. Data was collected on the types of collection programs and recycling and disposal facilities used in New Jersey for handling residential waste, along with the costs of these systems. The Tellus Institute's WastePlan model, a computer-based solid waste planning tool, was used to organize and analyze much of this data.

The data inputs to the model were collected from New Jersey state agencies, counties, and municipalities, as well as other sources. Because many county disposal methods were in transition in mid-1990 (when the data was collected), we determined what primary disposal method each county planned to use in the next three to five years, and

* We have modeled the New Jersey solid waste management system for two reasons. First, New Jersey Department of Environmental Protection was the original sponsor of this study. Second, the New Jersey solid waste management cost structure, as modelled here, may be indicative of the increasing solid waste costs in store for other regions in the future.

modelled the county's waste management system accordingly[**]. In our model, we collected data for seven different regions. Four counties were modelled individually: Atlantic, Camden, Ocean, and Somerset. The remaining 17 counties were placed into three groups, defined by the primary disposal method that each county planned to implement in the early 1990s -- either in-state incinerators, landfills or out-of-state transfer. These four individual counties and three groups were then treated as seven independent entities in our modelling efforts.

The highest per ton disposal costs, \$154 per ton, were borne by those counties relying heavily on out-of-state transfer; their costs were clearly dominated by the high cost of waste export. The lowest per ton disposal cost, \$113, was enjoyed by those counties grouped under the landfill category, with no one cost component clearly dominating.

Then, for each packaging material we determined the percent recycled, buried, burned, or transferred out-of-state, based on our model of the New Jersey solid waste system. Note that our calculations rely on existing and planned capacity as of mid-1990, and do not always agree completely with more recent data.

Finally, we estimated the marginal costs for each packaging material by increasing the quantity of the material in the waste stream (holding all other waste quantities and conditions constant), and recalculating the total system cost. The resulting cost increase, per ton of additional material, is the marginal cost. Thus the marginal cost includes cost increases to each of the solid waste system components, based on the percentage of the additional material that is recycled, buried, burned, or transferred. For example, in modelling the marginal costs of the recycling and garbage collection programs, the additional material in the waste stream implies that additional trucks and labor are required to collect all recyclables or garbage. Marginal costs were produced for the entire solid waste system and for individual collection programs and facilities.

The various types of packaging in the waste stream were broken down into eight categories: aluminum, glass, ferrous, corrugated cardboard, paper packaging (paperboard), PET plastic, HDPE plastic, and non-recyclable plastic containers. These categories broadly covered all of the types of packaging found in the residential waste stream.

4.2 Results of Marginal Cost Analysis

A number of patterns can be seen in the results of the marginal cost analysis. Costs per ton, the cost measure we report most often, is inversely related to density (with a few exceptions), since many waste management costs are based on volume. Collection costs of materials are primarily a function of how quickly the trucks become full, how frequently

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[**] New Jersey solid waste plans have changed since mid-1990, so our model no longer corresponds precisely to current county plans.

containers are set out, and how efficiently crews can collect materials. All of these characteristics are affected almost solely by the volume of the material being set out or recycled. The same is true for landfilling costs, and for the same reason: landfills, like trucks, fill up by volume.

However, not all solid waste system costs are volume-based. Incinerators and transfer stations charge for disposal based on the weight of the material. (Incinerator operations depend on energy and ash content as well as weight of materials received.) Another waste management option where costs are not directly tied to weight or volume is a recycling facility, where the highly differentiated processing technologies and material revenues play a large role in determining net costs for individual materials.

For example, consider the pattern of costs for incineration of various packaging materials. Plastics, which have a low ash content (about 8%) and a very high energy content (14,500 Btu/lb), actually have net revenues from incineration; revenues from electrical generation are larger than the costs of incinerator operation and residue disposal. On the other hand, aluminum, glass, and steel have high costs when incinerated because they generate virtually no electricity but leave a large amount of ash requiring disposal. Paper packaging is at an intermediate level of energy content, ash content, and net incineration costs.

In recycling, per ton revenues and the technologies employed to recycle each material are major factors affecting marginal costs. Aluminum, which receives about $900 per ton in revenues, has processing costs of about $166 per ton; thus the recycling facility has a marginal *revenue* of $734 per ton of plastics processed. Plastics, in contrast, have revenues at $140 per ton, and processing costs also equal to $140 per ton, resulting in a marginal cost of approximately $0 (break-even).

4.3 Summary of Final Results

The final results of this report are summarized in the table below. It presents the marginal costs, by weight and by volume, of handling each packaging material in our model of the New Jersey solid waste system. Aluminum has the lowest marginal costs whether measured by weight or volume, due to its high recycling rate and revenue. Other materials are ordered roughly according to their density. Denser materials, particularly glass, have lower costs per ton. But less dense materials, particularly plastics, have lower costs per cubic yard. However, all materials do have a significant cost impact upon the solid waste system. Clearly, the lowest cost option -- not considered in this report -- is packaging reduction, at zero disposal cost.

Summary of Packaging Material Marginal Costs

Material Density	Marginal Costs Per Ton	 Per Cubic Yard	Curbside Density (lbs/cu yd)
Aluminum	$ 23.73	$ 0.71	60
Steel	$134.30	$10.07	150
Glass	$ 70.98	$23.07	650
Corrugated Cardboard	$118.18	$ 8.86	150
Other Paper	$110.47	$ 9.67	175
PET	$249.80	$ 4.37	35
HDPE	$243.45	$ 4.26	35
Other Plastics	$235.06	$ 4.11	35

5. IMPACTS OF PRODUCTION AND DISPOSAL OF PACKAGING MATERIALS - METHODS AND CASE STUDIES

Our "Methods and Case Studies" report synthesizes and develops further the information presented in the earlier volumes, in order to yield a single, overall calculation of the impacts of production and disposal for each material. The results are first presented on a weight basis, showing the impacts of producing a ton of each major packaging material. Then, for selected case studies, the per-ton results are converted into per-package impacts, illustrating the use of our results for policy analysis.

5.1 Packaging Production Costs

Monetary costs of packaging production are already reflected in the prices of packaged products on the market. The price paid by a beverage bottler for cans or bottles, for example, is passed along to the final consumer. Environmental costs of production, however, are not incorporated. In Chapter 1 of the "Methods and Case Studies" report, we present a method for evaluating these environmental costs.

To make a comparison between economic and environmental costs, some explicit or implicit monetary valuation of the environmental costs is required. Refusal to place an explicit price on pollution, for research purposes, simply means that policymakers who use the results will apply their own implicit prices -- as they decide, for example, how much pollution is "enough" to justify a more costly but environmentally preferable technology.

Economists have proposed several methods, all of them problematical, for monetary valuation of environmental effects. We concluded that the least problematical (though certainly not problem-free) for our purposes was the control cost method, valuing pollutants at the price society is willing to pay for pollution controls. This method has been applied extensively in studies of energy generation, yielding price estimates for most EPA criteria air pollutants, and for greenhouse gases. We used these price estimates, although for all of our materials except glass and recycled aluminum, they turned out to be of little importance to the final results (see Table 3.1, which is reproduced at the end of this summary).

Far more important to our results is the evaluation of the hazardous substances emitted in production of packaging materials. These hazards were identified in detail, and linked to production processes, in our inventory of packaging production impacts. The dozens of hazardous substances we studied vary widely in their degree of hazardousness; a central challenge was the development of a system to rank and compare these effects.

After a review of the diverse possible approaches to hazard ranking suggested by the scientific literature, we adopted several simplifying assumptions and procedures. Although we recognize the broad, multi-faceted nature of environmental impacts associated with hazardous substances, we limited the scope of our study to the relatively well-documented area of human health effects. Moreover, we used laboratory analyses of the health effects of pollutants, ignoring differential impacts resulting from pollutant transport from source to receptor.

We grouped hazardous substances into two categories, carcinogens (cancer-causing pollutants) and noncarcinogens (pollutants that cause toxic health effects other than cancer). For carcinogens our ranking is proportional to the published cancer potency factors, or measures of the cancer risk, of each substance. For noncarcinogens we used the oral reference dose (RfD), an estimate of the maximum daily exposure which will not cause harm; since a higher RfD means less serious health effects per unit of the pollutant, our ranking is proportional to 1/RfD. To establish the "exchange rate" between the carcinogen and noncarcinogen rankings, we used Occupational Safety and Health Administration (OSHA) permissible exposure levels for substances at the bottom of each ranking (i.e., for the least hazardous carcinogen and noncarcinogen in our study) to derive an overall ranking for each pollutant. Alternatives to these methods which we considered and rejected are discussed in the text.

The result of this calculation is a single numerical health effects ranking per pound of each pollutant in our study. That ranking allows, for example, summation of the total health effects of pollutants emitted in the production of a ton of glass, or any other pollutant. It also means that, if a price can be established for any one pollutant, it can be extended to the others in proportion to their health effects.

To establish a price for one pollutant, we reviewed the costs of compliance with regulations on airborne lead emissions. Currently it appears that our society is willing to impose control costs equivalent to about $1,600 per pound of lead removed from an industry's air emissions. We took this value as our benchmark, and assigned proportional prices for the remaining hazardous pollutants. A pollutant which is one-tenth as hazardous as lead would be assigned a cost of $160 per pound; a pollutant three times as hazardous as lead would be assigned a cost of $4,800 per pound. (A change in the $1,600 lead cost estimate would scale all health hazard "prices" up or down uniformly, but would not change their relative sizes.)

Finally, the prices are used to estimate the total health "cost" of production for each packaging material. The price for each pollutant is multiplied by the amount of that pollutant caused by the production of one ton of glass, for example; the results are summed to yield the overall health cost of glass production.

5.2 Environmental Impacts of Packaging Disposal

Our disposal cost report modelled the conventional cost of handling packaging materials in New Jersey's solid waste system. In Chapter 2 of the "Methods and Case Studies" report we examined the environmental impacts of the solid waste collection systems and facilities which handle packaging materials.

For landfills, we estimated the leachate pollutants per ton of landfilled waste, in a modern, controlled landfill. We then allocated these pollutants to the waste stream components, based on the quantities and chemical composition of wastes. Since packaging materials are not major contributors to landfill gas production, we omitted any calculation of landfill gas impacts.

For incinerators we estimated the air pollutants and ash produced by burning a ton of waste in a new mass-burn incinerator with state-of-the-art air pollution controls. As with landfills, we then allocated these pollutants to the materials in the waste stream based on the quantities and chemical composition of the wastes. A review of leachate data from ashfills revealed that the quantity of pollutants released by this route is minor compared to incinerator air emissions.

For recycling facilities, very little information is available. We reviewed one published study that measured ambient air pollution levels (which are different from facility emissions) at a fairly small recycling facility. These pollution levels are very low compared to other solid waste facility emissions. Moreover, the tested recycling facility's pollution levels are difficult to associate with particular materials, or to scale up to larger facility operations. Lacking any other information, we did not calculate environmental impacts of recycling facility operations.

For garbage truck and recycling truck operations we used published air emission factors for the relevant truck sizes, and our estimates of the amount of driving required to collect packaging materials.

The result of Chapter 2 is a full set of environmental emissions per ton of each packaging material handled in the New Jersey solid waste system.

5.3 Full Costs of Packaging Production and Disposal

In Chapter 3 of the "Methods and Case Studies" report, we bring together the results of the preceding chapters and studies to calculate the full cost of packaging production and disposal. The environmental emissions per ton of packaging material production, as calculated in our production inventory report, are multiplied by the pollutant prices developed in Chapter 1. Conventional solid waste system costs, as calculated in our disposal cost report, are added to the environmental costs of the same system (using pollutant prices from Chapter 1 and waste system impacts from Chapter 2), yielding total solid waste system costs. The sum of these categories is the full cost (excluding the conventional cost of production, which is included in the price, and therefore not included in our study) of each material.

Full costs appear in Table 3.3, which is reproduced at the end of this summary. The most striking number in the table is the huge cost for PVC, far above any other material. Next in line (next highest on a per-ton basis) is virgin aluminum, followed by plastics, steel, and most virgin paper products. Recycled paper and corrugating medium made from virgin paper are even lower, while recycled aluminum, virgin glass, and recycled glass are the lowest-priced.

In most cases the majority of the cost is due to one or a few pollutants emitted in production. The high PVC price largely reflects the health hazard of emissions of vinyl chloride (VC), a known human carcinogen, during monomer production. Other plastics prices are largely based on the environmental costs of naphthalene emissions at several stages in production.

Virgin paper products that include kraft pulping (all but corrugating medium) have costs dominated by particulate emissions from pulping. For virgin corrugating medium and for recycled paper, sulfur oxide and nitrogen oxide emissions (both from the process itself and from the required electricity production) are the most important costs. The relatively low costs for glass are likewise largely due to sulfur oxide, nitrogen oxide, and VOCs.

Virgin aluminum costs are primarily based on nitrogen oxide, particulates, and sulfur oxide emissions. The much lower costs for recycled aluminum include particulates released in recycling, and sulfur oxides and nitrogen oxides from process energy. Steel costs are largely due to lead, coke oven emissions, particulates, and sulfur oxides.

Two important conclusions can be drawn from these results. First, the fact that a few pollutants dominate the cost (and environmental impact) from each packaging material suggests a focus for further research. If the key emissions from a production process could be reduced or eliminated, the environmental cost of using that material could be sharply reduced. The calculated cost levels, based on existing technology, are important for policymaking purposes, but so is an assessment of the relative potential for industrial pollution prevention in the material-processing industries. A relatively more "costly" material could be preferred over a "cheaper" one, if the more costly one can be more readily cleaned up.

Second, the total cost of disposal for many packaging materials is small when compared to the environmental costs of packaging material production. Within the category of disposal costs, environmental costs are small when compared to conventional (monetary) disposal costs. Our results imply that many emissions from materials processing industries are more damaging to health and the environment than are the emissions from solid waste facilities.

For the purposes of our analysis, we have placed monetary "prices" on health and environmental effects which are usually thought of as intrinsically non-monetary values. Our prices may be interpreted as follows: **IF** society consistently valued the range of pollutants we studied, in proportion to their human health effects, at prices consistent with $1,600 for lead, **THEN** our cost calculations would apply. Those calculations would suggest that the "expensive" (health-threatening) emissions from industry are a greater priority for control than the relatively smaller remaining emissions from new state-of-the-art solid waste facilities. This may be taken as setting an agenda for further efforts in pollution prevention research and policy.

5.4 Case Studies

In our final chapter we converted the per-ton costs of Chapter 3 into per-package costs for five specific consumer products. The distinction is important because different packages for the same product often have very different weights. For example, glass bottles are much heavier than plastic bottles of the same size. So even though glass looks much better per ton in Chapter 3, it loses much or all of that advantage on a per-bottle basis.

We selected five products: soft drinks, juice, fast-food hamburgers, microwave dinners, and hardware. For each product, we purchased, cleaned and weighed the packaging used for the product, and assigned it the appropriate costs as calculated in Chapter 3. The case study results are summarized in Table 4.13, which is reproduced at the end of this summary.

For soft drinks the full cost of packaging materials ranges from 0.05¢ to over 0.39¢, with recycled aluminum having the lowest price and virgin glass the highest. At the 10-12 ounce size, PET, virgin aluminum, and recycled glass are next lowest after recycled aluminum, while virgin glass is highest. The 2-liter PET bottle (the only material available in this size) has the lowest per-ounce impacts but becomes more expensive per ounce as the package size decrease.

For juice we examined several size categories. Among the largest containers (46 or 64 ounces), PET and virgin glass have the highest environmental impacts; HDPE and paperboard are lowest while virgin and recycled steel and recycled glass are intermediate. In one-pint containers, paperboard and HDPE again have the lowest costs while virgin and recycled glass have higher costs. Among the smallest, single-serving containers, virgin aluminum and glass are the highest-cost options, while recycled aluminum is the lowest. The much-debated aseptic, composite package is somewhat higher than recycled aluminum, although it should be noted that we did not calculate the emissions from the complex package fabrication stage.

In the case of fast-food hamburgers we studied the three packaging options used at the two leading chains, McDonald's and Burger King: bleached coated folding boxboard "clamshell" containers, polystyrene clamshells, and paper wrappers. The paper wrapper is clearly preferable; the polystyrene clamshell has a full cost 1.6 times as great as the wrapper, while the boxboard clamshell cost is over three times that of the wrapper.

We examined five microwave meal packaging options, for meals weighing roughly 9 ounces. The complexity of the package, rather than the choice of materials, appears to be the dominant factor: the two packages in which the outer box is the meal tray were clearly preferable. Since the materials used are relatively similar in cost per ounce, packaging impacts were roughly proportional to package weight. The weight of packaging used to deliver a 9 ounce microwaveable meal varied from 1 ounce to almost 3 ounces, with corresponding variation in environmental impacts.

Our study of hardware packaging options focused on alternatives for the delivery of 100 all-purpose screws. Not surprisingly, alternatives involving PVC were vastly more expensive than those using other materials, reflecting the uniquely high per-ounce environmental cost of PVC. Among other materials, plastic or recycled paper containers were somewhat lower in cost than virgin paper boxes or bags.

These case studies are not meant to be the only, nor necessarily the most important, applications of our results. Rather, they are meant to illustrate a method whereby the analysis of materials, as developed in our study, can be applied to a broad range of specific packages.

Table 3.1 Environmental Cost of Packaging Material Production

Materials	Criteria Air Pollutants, Methane, Chlorine, and Hydrogen Chloride ($/ton material)	Toxic and Carcinogenic Pollutants ($/ton material)	TOTAL Environmental Production ($/ton material)	% of Total Env. Production from Toxics/ Carcinogens
PLASTIC				
HDPE	$170	$122	$292	42%
LDPE	$210	$134	$344	39%
PET	$261	$593	$854	69%
PP	$157	$210	$367	57%
PS	$189	$196	$385	51%
PVC	$188	$4,864	$5,053	96%
PAPER				
Bleached Kraft Paperboard	$229	$101	$330	31%
Unbleached Coated Folding Boxboard	$187	$82	$269	30%
Linerboard	$193	$80	$273	29%
Corrugating Medium	$77	$6	$83	8%
Unbleached Kraft Paper	$193	$84	$277	30%
Folding Boxboard from wastepaper	$120	$14	$135	11%
Linerboard from wastepaper	$121	$15	$135	11%
Corrugating Medium from wastepaper	$162	$21	$183	12%
Virgin Glass	$83	$3	$85	3%
Recycled Glass	$54	$0	$55	0%
Virgin Aluminum	$1,511	$423	$1,933	22%
Recycled Aluminum	$312	$1	$313	0%
Virgin Steel	$74	$156	$230	68%
Recycled Steel	$74	$147	$222	66%

Table 3.2 Full Costs of Packaging Material Disposal

Materials	Conventional Disposal ($/ton material)	Environmental Disposal ($/ton material)	TOTAL DISPOSAL ($/ton material)
PLASTIC			
HDPE	$242	$4	$245
LDPE	$232	$4	$236
PET	$250	$5	$255
PP	$232	$4	$236
PS	$232	$4	$236
PVC	$232	$4	$236
PAPER			
Bleached Kraft Paperboard	$110	$2	$112
Unbleached Coated Folding Boxboard	$110	$2	$112
Linerboard	$118	$2	$120
Corrugating Medium	$118	$2	$120
Unbleached Kraft Paper	$110	$2	$112
Folding Boxboard from wastepaper	$110	$2	$112
Linerboard from wastepaper	$118	$2	$120
Corrugating Medium from wastepaper	$118	$2	$120
Virgin Glass	$71	$1	$72
Recycled Glass	$71	$1	$72
Virgin Aluminum	$24	$5	$29
Recycled Aluminum	$24	$5	$29
Virgin Steel	$134	$2	$136
Recycled Steel	$134	$2	$136

Table 3.3 Full Costs of Packaging Material Production and Disposal

Materials	FULL COST ($/ton material)	FULL COST ($/ounce material)
PLASTIC		
HDPE	$537	$0.017
LDPE	$580	$0.018
PET	$1,108	$0.035
PP	$602	$0.019
PS	$620	$0.019
PVC	$5,288	$0.165
PAPER		
Bleached Kraft Paperboard	$443	$0.014
Unbleached Coated Folding Boxboard	$382	$0.012
Linerboard	$394	$0.012
Corrugating Medium	$204	$0.006
Unbleached Kraft Paper	$390	$0.012
Folding Boxboard from wastepaper	$247	$0.008
Linerboard from wastepaper	$256	$0.008
Corrugating Medium from wastepaper	$303	$0.009
Virgin Glass	$157	$0.005
Recycled Glass	$127	$0.004
Virgin Aluminum	$1,963	$0.061
Recycled Aluminum	$342	$0.011
Virgin Steel	$366	$0.011
Recycled Steel	$358	$0.011

Table 4.13 Packaging Case Studies - Summary Cost Comparisons

Product	Size	Material	Environmental cost cents/unit	Unit
SOFT DRINK	2 liter	PET	0.12	fluid
	1 liter	PET	0.17	ounce
	1 pint	Virgin glass	0.31	
	1 pint	Recyclad glass	0.26	
	1 pint	PET	0.25	
	12 ounce	virgin aluminum	0.31	
	12 ounce	recycled aluminum	0.05	
	12 ounce	PET	0.28	
	10 ounce	virgin glass	0.39	
	10 ounce	recycled glass	0.32	
JUICE	1/2 gallon	virgin glass	0.19	fluid
	1/2 gallon	recycled glass	0.15	ounce
	1/2 gallon	PET	0.18	
	1/2 gallon	paperboard carton	0.06	
	1/2 gallon	HDPE	0.04	
	46 ounces	virgin steel	0.15	
	46 ounces	recycled steel	0.14	
	1 pint	virgin glass	0.32	
	1 pint	recycled glass	0.26	
	1 pint	paperboard carton	0.08	
	1 pint	HDPE	0.09	
	11.5 ounce	virgin aluminum	0.32	
	11.5 ounce	recycled aluminum	0.06	
	10 ounce	virgin glass	0.29	
	10 ounce	recycled glass	0.24	
	8.5 ounce	aseptic packaging	0.10	
	6 ounce	virgin steel	0.22	
	6 ounce	recycled steel	0.22	
FAST FOOD	clamshell	boxboard	0.86	quarter-pound
BURGERS	clamshell	polystyrene	0.40	hamburger
	wrapper	paper	0.25	
MICROWAVE	no-tray dinner	paperboard	0.17	ounce of food
DINNERS	no-tray dinner	paperboard, pouches	0.19	
	light tray	boxboard, paperboard	0.35	
	light tray	boxboard, PET	0.43	
	heavy tray	paperboard, HDPE	0.79	
HARDWARE	box	virgin boxboard	0.30	100 screws/nails
	box	recycled boxboard	0.19	
	plastio container	PVC	7.44	
	blisterpack	PVC, paperboard	1.77	16 screws
	blisterpack	PVC, rec. paperboard	1.73	
	plastic bag	LDPE	0.09	
	paper bag	paperboard	0.30	

89-024/1NJ

Assessment of Impacts of Production and Disposal of Consumer Packaging on the Environment

NJ DEPE Contract P31152

Report #1:
Literature and Public Policy Review

prepared for:

New Jersey Department of Environmental Protection and Energy

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Introduction

This is the first quarterly report developed under the CSG/Tellus Institute[1] Packaging Research Study, titled "Assessing the Impacts of Production and Disposal of Packaging and of Public Policy Measures to Alter its Mix." The Packaging Research Study is intended to provide analysis and information of use to policymakers interested in packaging issues and solid waste source reduction strategies. The study has four broad objectives:

- to describe the environmental impacts of packaging production, separately for each major packaging type and for virgin vs. secondary raw materials

- to calculate the solid waste system costs and impacts resulting from disposal of each packaging type, using Tellus Institute's WastePlan model and data on the New Jersey solid waste management system as a case study

- to analyze the likely economic responses to packaging taxes or bans by packaging manufacturers, end-product manufacturers, and final consumers

- to derive the macroeconomic impacts of packaging policy measures on state industry, employment, and income levels

As a first step in this research, we have surveyed the available literature and policy proposals in three general areas: packaging legislation and policy measures, environmental impacts of packaging, and economic analysis of policy measures. The three sections of this report examine these areas in depth.

Section 1 surveys a wide range of current and proposed legislation and policy measures. Most U.S. initiatives have been at the state and local levels in recent years, particularly in states facing high solid waste disposal costs and diminishing disposal capacity. Existing (already enacted) legislation addresses packaging waste primarily by encouraging recycling. Several states have adopted "bottle bills", a large and growing number of areas have mandatory recycling programs, and a variety of measures have been designed to stimulate the market demand for recycled and recyclable commodities.

Another group of proposals aims at source reduction, i.e. at eliminating the production of packaging waste. Source reduction initiatives (most of which have not yet adopted) include packaging taxes or fees, bans of specific materials, consumer and producer education programs, and toxicity reduction measures.

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modelling of both supply and demand for secondary materials. Since secondary materials markets were (by today's standards) comparatively underdeveloped in the 1970s, it is not surprising that both studies' assumptions in this area are obsolete by now.

The CSG/Tellus Institute Packaging Research Study is funded by the New Jersey Department of Environmental Protection, the federal Environmental Protection Agency, and by many states, agencies and industry groups acting through the Council of State Governments. This report is a collective product of the Tellus research staff; primary authors of the three sections are Mark Rossi (legislative review), Karen Shapiro (environmental impacts), and Irene Peters (economic analysis).

SECTION 1 PACKAGING LEGISLATION

1.1 INTRODUCTION

With solid waste disposal costs skyrocketing, states and communities are both considering and enacting legislation that will limit packaging products in municipal solid waste. In April, 1988, Suffolk County, New York, banned polystyrene and polyvinyl chloride packaging materials. Their rationale being that these materials are non-biodegradable, are difficult to recycle, and are potential sources of toxins when incinerated. One year later, in April of 1989, Minneapolis, Minnesota passed an ordinance banning non-recyclable, non-degradable, or non-reusable packaging from store shelves and fast-food restaurants.[1] Throughout the country, the winds of solid waste source reduction are blowing with growing intensity. Packaging products are increasingly becoming an object of legislative initiatives because packaging represents 30 percent of all municipal solid waste by weight. In 1988, approximately 2,000 solid waste bills were introduced in state legislatures, with 300 of them specifically addressing packaging issues: 206 on plastics; 77 on biodegradability; and 11 on paper packaging.[2]

This review and evaluation of packaging legislation examines initiatives and laws which account for the environmental impacts of packaging products and the materials they use. Since a comprehensive discussion of each law is beyond the scope of this study, representative packaging legislation initiatives and laws are described and evaluated. The evaluation of each initiative or law includes a discussion of the four questions listed below:

1. What are the goals of the proposed policy?
2. What is the target of the proposed policy?
3. How is it supposed to impact the targeted population?
4. What is the desired response from the targeted population?

Currently the different packaging source reduction initiatives enacted by governments range from none (allowing the market to operate without any public interference) to banning all packages manufactured with a specific material. Table 1 lists various policy options employed or considered by public agencies to reduce packaging produced and landfilled.

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[1]William E. Schmidt, "Local Laws Take Aim at Indestructible Trash," *The New York Times*, April 23, 1989, Section IV, p. 5.

[2]Ibid.

Table 1. Packaging Reduction Measures

Voluntary

- establishing research institutions
- joint industry/government research projects
- seminars for designers
- positive labels/logos
- media campaigns
- school curricula
- public education

Financial Incentives/Disincentives

- tax credits
- fees
- container deposit fees
- grants
- low-interest loans
- variable garbage collection fees

Regulatory

- outright bans
- conditional bans (product design requirements)
- minimum warranty requirements
- mandatory disclosure of environmental impacts
- mandatory recycling programs

Through voluntary initiatives, governments hope that with the right *information*, manufacturers will voluntarily reduce packaging production and consumers will voluntarily reduce packaging consumption. Financial initiatives are promulgated by the state to counteract the inability of the free market to account for a product's environmental impacts. This is done by using either a carrot -- tax credits, container deposits, grants, or low-interest loans -- or a stick -- taxes on products or fees on garbage. Regulatory initiatives are commands from the state to manufacture a product or provide a service such that consumers are educated, products eliminated, or externalities otherwise accounted for.

Solid Waste Jargon

The types of legislative initiatives which address the environmental impacts of a package are: 1) source reduction measures; 2) measures encouraging/mandating that products be recyclable; 3) deposit fees; and 4) variable garbage collection fees. Packaging source reduction is the elimination or minimization of the quantity and/or toxicity of packaging in the design, manufacturing, and consumption stages as opposed to after the product is already in the home or office. In Figure 1, which displays the packaging product lifecycle, it is seen that packaging source reduction decisions are made during "package design" and "manufacturing process," and by the "consumer." Also in Figure 1, it is seen that consumer source reduction measures include decisions to "reuse" a package or to "backyard" compost; another consumer source reduction measure is to purchase a product with less or no packaging.

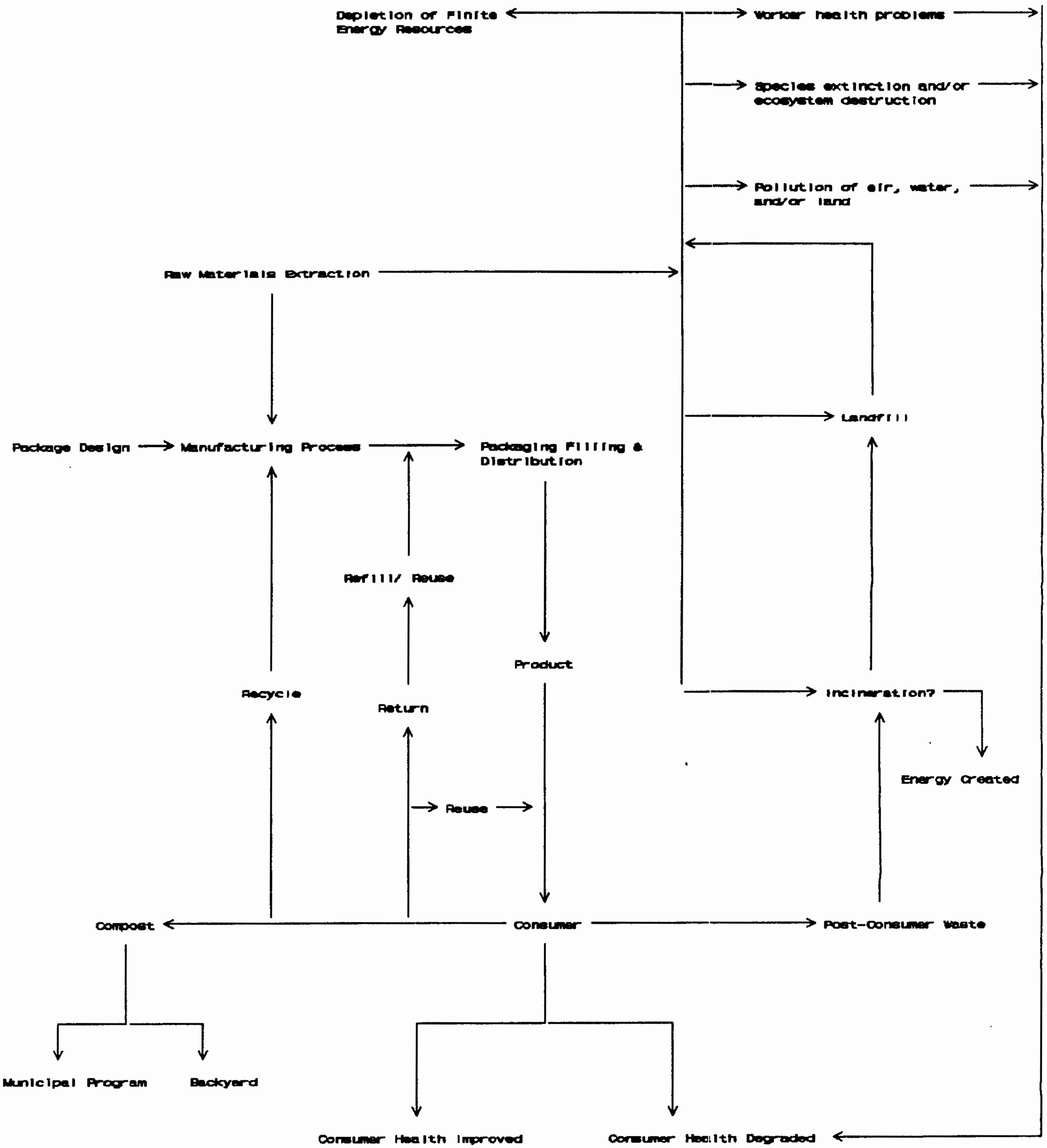
As a solid waste management strategy, source reduction is the most environmentally benign measure, according to the U.S. Environmental Protection Agency's (EPA) solid waste hierarchy. From most to least environmentally benign, the solid waste hierarchy adopted by the EPA is: source reduction, recycling, incineration, and landfilling.[3]

Source reduction measures top the hierarchy because they solve packaging waste problems by removing solid and toxic wastes from the waste stream, and encouraging and mandating the production of recyclable, reusable, and durable products. The source reduction definition outlined above, however, is not universal. In the debates and discussions surrounding efforts to decrease solid waste production, source reduction frequently becomes muddled with other solid waste terms such as waste reduction, waste minimization, and waste management.

For example, Elliott Zimmermann in *Solid Waste Management Alternatives*, and Minnesota's Select Committee on Recycling and the Environment (SCORE) refer to source reduction as waste reduction: waste reduction is the "reduction, avoidance, or elimination of the generation of pollutants or wastes." Karen Hurst and Paul Relis in their report, *The Next Frontier: Solid Waste Source Reduction*, specifically differentiate between source reduction and waste reduction: "waste reduction is an umbrella term for all waste management methods resulting in a reduction of waste reaching the landfill" whereas source reduction "focuses on the production of waste ... with the aim of reducing toxicity and volume of waste being generated and consumed." For clarity and simplicity, this survey adheres to the source and waste reduction definitions outlined by Hurst and Relis,

[3]U.S. EPA, *Solid Waste Dilemma: An Agenda for Action*, 1989.

Figure 1. Packaging Product Lifecycle Flowchart



which are also consistent with EPA definitions, which state that source reduction is "the reduction of waste at the point of production, as opposed to waste reduction, which implies reducing the amount of waste to be disposed, without specific reference to reducing the amount of waste generated."[4] What is important to note about these definitions is that waste reduction implies managing a waste that has already been produced (back-end management), whereas source reduction implies action taken by the manufacturer to either eliminate packaging materials, or design them so they can be easily managed so as to not appear at a landfill (front-end management). In Figure 1, waste reduction decisions occur when the "consumer" decides to "recycle," "return," or compost in a "municipal program;" or when the solid waste is incinerated ("incineration?").

Thus bans, depending on how they are defined (see Regulatory Initiatives), result in material substitution and/or source reduction. Initiatives that require products to be recyclable are a combination of both source reduction and waste reduction measures. The source reduction part requires manufacturers to produce a package which is capable of being recycled (front-end management). The waste reduction part hopes (unless the community has mandatory recycling) that the package is recycled (back-end management). The Office of Technology Assessment (OTA) in their report, *Facing America's Trash: What Next for Municipal Solid Waste?*, developed a classification system different than the EPA's solid waste hierarchy. The OTA divides waste management activities into two areas: prevention (i.e., source reduction) and materials management. They categorize recyclable design initiatives under materials management because design initiatives require back-end management. However, in this study recyclable design initiatives are classified as source reduction because they may require changes in production processes. Recyclable design initiatives require no changes in collection processes. Another important note to recyclable design requirements is that when a manufacturer switches from a recyclable package to a package which uses less material and is non-recyclable, the new source reduced package may produce greater environmental impacts.

Deposit fees, popularly known as bottle bills, are included in this report because they account for the environmental impacts of disposal by placing a fee on the bottle; thereby encouraging its return and recycling. Bottle bills are waste reduction measures. Variable collection fees also encourage recycling as well as residential source reduction measures by increasing the costs of garbage disposal.

1.2 STATE and LOCAL GOVERNMENT INITIATIVES

Packaging reduction initiatives by state and local governments are far more prevalent than initiatives on the federal level. Reasons for this include the local nature of

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[4]U. S. EPA, "EPA Municipal Solid Waste Source Reduction: Design for Disposal Policy Dialogue (meeting summary)" (Washington, D.C., June 14, 1988), p. 1.

solid waste problems and the curtailing of the Environmental Protection Agency's (EPA) role in solid waste management during the Reagan administration. Certainly the federal government has a role to play in packaging reduction, but to date, state and local governments have taken the lead.

For each initiative discussed in this section a description of and rationale for the initiative is provided. The rationale provided is a statement of fact, it is not an indication of support for the initiative. For example, the statement "many communities are banning packaging materials which are non-biodegradable" is a statement of why communities decide to ban a package, not why, or whether it should be banned

Voluntary Initiatives

Voluntary packaging source reduction initiatives aim at altering the product cycle by changing design, manufacturing processes, and/or consumer behavior. Those measures which target manufacturers include: establishing research institutions, joint industry/government research projects, and seminars for designers. Voluntary initiatives which aim at consumers include: positive labels/logos, media campaigns, school curricula, and public education. Most voluntary initiatives target consumers rather than manufacturers.

States frequently implement voluntary measures as part of broader solid waste management initiatives. For example, included in Florida's "1988 Solid Waste Act"[5] are three public education measures: 1) to inform users (e.g., homeowners) of garbage disposal costs; 2) to develop public awareness programs; and 3) to develop school curricula. Included in Maine's "Act to Promote Reduction, Recycling and Integrated Management of Solid Waste and Sound Environmental Regulation"[6] are sections on school curricula and a media campaign.

The goals of the Florida and Maine sections mentioned above are to promote public awareness of 1) recycling, 2) source reduction, and -- specifically in the Florida law, 3) disposal costs. These policies target solid waste as a whole, rather than targeting a specific product or material in the solid waste stream (e.g., diapers, plastics packaging, newspapers, etc.). The impacts of such policies can range from public apathy (no changes) to an increase in recycling rates and decrease in trash production because of recycling and source reduction.

[5]"The 1988 Solid Waste Act," S-1192, signed into law on June 24, 1988,

[6]An "Act to Promote Reduction, Recycling and Integrated Management of Solid Waste and Sound Environmental Regulation," LD 1431, signed into law on July 12, 1989.

Financial Initiatives

The financial initiatives examined in this section include: source reduction measures (i.e., tax credits, product fees, grants, and low-interest loans), container deposit fees, and variable fees for municipal solid waste collection. Since low-interest loans, grants, and investment tax credits are tangential to packaging reduction they are discussed in brief.

Financial source reduction initiatives encourage manufacturers to alter product design and production processes to reduce the quantity and toxicity of the final package. Of the financial source reduction initiatives, a packaging fee is the only initiative to target all packaging products. Occasionally tax credits are directed at packaging materials, but their primary purpose is to promote investments in manufacturing plants which make products from secondary materials (i.e., investment tax credits). Container deposits reduce beverage packaging waste by adding value to the package. This additional value, in part, addresses the negative externalities of disposal costs and litter. A fourth measure, variable fees, encourage both source reduction and recycling by increasing disposal costs.

<u>Deposit Fees</u>

The only packaging legislation passed into law which are similar to a packaging tax, in that they employ financial incentives or disincentives, are container deposit laws. Deposit laws are in effect in ten states: California, Connecticut, Delaware, Iowa, Maine, Massachusetts, Michigan, New York, Oregon, and Vermont. Florida will become the eleventh state when their fee becomes effective in October 1992. In all the states except Michigan and California a deposit of five-cents is placed on carbonated beverage containers; the fee in Michigan is 10 cents. Technically, the California and Florida bills are not deposit laws. The Florida law excludes those containers with recycling rates above a specified percentage. The California law does not place a fee on any container; it simply offers consumers a redemption for returned containers. This is dissimilar to the nine other states where no containers covered by the deposit are exempt from the fee for any reason.

Traditionally container deposit laws are limited to carbonated beverage containers -- soft drinks and beer. But as part of the comprehensive solid waste law passed in 1989, Maine expanded its container deposit law to include all containers of one gallon or less of any beverage, except dairy products packaged in glass, metal, or plastic containers. The bill becomes effective for all containers on September 1, 1990.

In California, consumers can redeem soft drink and beer containers for two cents (or five cents for two containers) if the recycling rate for the container is 65 percent or above; below 65 percent the redemption rate is five cents. Containers with a capacity of 24 fluid ounces or more are equivalent to two containers.

Florida's "advance disposal fee" (which is part of the Florida 1988 Solid Waste Act) places a one-cent fee on all containers made of glass, plastic, plastic-coated paper, aluminum, or other metals being recycled at rates of less than 50 percent by October 1, 1992. Containers are redeemable at registered recycling centers. If the 50 percent goal is not attained by October 1, 1995, the fee increases to two cents per container. The advance disposal fee is similar in design to the California fee, but covers more containers.

Variable Fees

Variable fees concentrate on changing consumer habits by connecting increasing disposal costs to the increasing amount of garbage residents place at the curb. Variable fee programs differ in specifics, but are based on the same concept: charge residents by the bag for the trash on the curb. Thus, such programs encourage residents to reduce trash primarily by participating in recycling and/or composting programs, and secondarily, by altering buying habits (e.g., purchasing products with less packaging) and by encouraging backyard composting. Reactions to a variable fee can range from benign actions -- such as compacting wastes -- to malign actions, like dumping trash on the sides of roads. However, community managers find that resistance decreases over time as residents become accustomed to the program.[7]

Variable fees are implemented on the local level by municipal governments. Communities that have instituted variable fees include High Bridge, New Jersey; Portland, Oregon; Perkasie, Pennsylvania; Seattle, Washington; and Woodstock, Illinois.

Tax Credits/Exemptions

Tax credits and exemptions typically are designed to reduce a manufacturer's operating costs, in order to encourage a socially beneficial action. In the past this included resource development (e.g., mining, drilling, or forestry) and job development; today this may include manufacturing products made with recycled materials. One instance of a tax credit being targeted at packaging -- in this case a tax exemption -- is Rhode Island's "Act Promoting the Use of Paper Bags ...,"[8] which "exempts from the state sales and use taxes all biodegradable bags, boxes and wrapping materials; and all returnable containers."[9] The law also requires all stores offering plastic bags to also offer paper bags (this part is discussed under regulatory measures). The intention of this law is to encourage retailers

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[7]David Riggle, "Only Pay for What You Throw Away," *BioCycle*, February 1989, p. 39.

[8]"An Act Promoting the Use of Paper Bags in Order to Reduce the Cost and Difficulty of Waste Disposal," H. 9163, signed into law on June 9, 1988.

[9]Environmental Action Foundation, "Legislative Summary."

to use paper products (which are biodegradable) over plastic products (which are non-biodegradable). The law assumes that plastic bags, boxes, and wrapping materials pose more environmental problems than their paper counterparts. Thus, it promotes neither recycling nor source reduction, but the substitution of one material with another. Rhode Island passed this measure without full knowledge of the environmental impacts of promoting this material substitution. Whether or not the measure is effective depends on the environmental merits of encouraging the use of paper over plastic.

The Rhode Island law, however, is atypical of most tax credits/exemptions; much more prevalent is the use of an investment tax credit. Investment tax credits provide manufacturers with a deduction against their state or federal taxes for investing in a specific production process: for example, manufacturing paper from recycled paper, or as has historically been the case, in raw materials extraction. Low-interest loans and grants are similar to investment tax credits because they can be designed to encourage the construction or relocation of secondary material manufacturing plants into a state. Grants, low-interest loans, and investment tax credits typically target specific materials (e.g., plastics) instead of specific product types (e.g., packaging). Tax credits, low-interest loans, and grants would be source reduction measures if they were offered to manufacturers for decreasing their use of packaging.

Packaging Fees

A packaging fee, unlike the other financial initiatives, targets all packaging products with the specific aims of reducing packaging waste, increasing recycling, and raising revenue. Economically, a fee is designed to internalize the environmental costs of disposal (and increasingly of production as well) into the price of the product, thereby producing an incentive for manufacturers to produce environmentally benign products. The decision on where to place a fee in a product's lifecycle (at the site of manufacture, distribution, or consumption) will determine the relative effectiveness of a fee in source reducing packaging. State legislatures currently considering packaging fee bills include Vermont, Oregon, New York, Connecticut, California, Minnesota, and Massachusetts.[10]

Rather than dwelling on the mechanics of each fee initiative, this section briefly discusses the Oregon, New York, Connecticut, and Minnesota bills, and focuses on the Massachusetts[11] and Vermont[12] bills. All the bills, except Oregon's, are similar in that they

[10]In 1989, none of these bills were passed with their tax provisions.

[11]H. 5654, "An Act to Protect the Environment by Encouraging a Reduction, Reuse, Recycling and Return of Packaging and Other Consumer Products," introduced by Representative Mark Roosevelt.

exempt packages which meet reuse, recyclability, and/or recycled content requirements. The New York, Connecticut, Minnesota, and Massachusetts bills all place a per unit fee on packages, whereas Oregon and Vermont tax gross sales (Oregon) and gross receipts (Vermont).

New York's "Waste Reduction and Recycling Act of 1988" places a three-cent fee on non-recyclable rigid and semi-rigid containers for non-food retail packaging and fast-food packaging. All grocery foods are exempt from the fee, and packages are also exempted if they are recyclable. In Connecticut, H. 6641 places a fee on fast-food packaging, containers, single-use items, disposables, and newspapers. Exemptions are given for recycled material content or if the material is recyclable.[13] Included in Minnesota's "Comprehensive Waste Reduction and Recycling Act of 1988" is a fee similar to Massachusetts', except the fee is one-cent instead of three-cents.

The Massachusetts initiative includes many solid waste measures, but the central feature of H. 5654 is a three-cent fee on packaging products recycled at rates less than 30 percent annually, up until 1993 (and at rates less than 50 percent thereafter), or manufactured with less than 50 percent secondary material. Products which meet one of these criteria are exempt from the fee. In an effort to address objections made to an earlier version of this bill, many exemptions are listed in the bill's most recent version; these include: 1) packages of two cubic inches or less; 2) packages covered under the container deposit law; 3) packages used in prescribed medicine; 4) tamperproof seals; 5) packages not intended for retail sale; 6) packages exempted by the FDA; and 7) reusable packaging. H. 5654 is applied to the package's first sale in the state and is based on a per unit basis; in theory the fee can also be based upon weight, composition, value, or any combination of these methods.

The Oregon bill (HB 2959, "The Plastic Container Tax Act") applies to suppliers making the first sale of plastic containers in the state. The tax is based on a percentage of gross sales and is perceived primarily as a revenue generating bill.[14] Similar to the Oregon bill is Vermont H. 404, which taxes gross receipts at a rate of 0.05 percent for all pre-packaged materials sold in Vermont. A fee of one cent is charged on empty packaging used on-site (e.g., fast-food restaurants). A 50 percent exemption to the fee is provided to packaging made 100 percent from a recyclable material and another 50 percent exemption is available if at least half of the package is made from recycled materials.

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[12]The Vermont bill was introduced in 1989 as H. 404.

[13]"The Latest Rage: Packaging Taxes," *Plastics Recycling Update*, March 1989, p. 2.

[14]Environmental Action Foundation, "Legislative Summary: Significant Packaging Initiatives Passed or Considered in 1988," 1988.

Another fee being considered in California is a four cents per pound user fee on all non-recyclable or non-degradable plastic products (A.B. 1796). The fee is to be paid by manufacturers or distributors.

The effects of a packaging tax depend on where in the product's lifecycle it is assessed. By assessing the fee as close to the manufacturer as possible, states are encouraging manufacturers to alter their production processes so that they account for the environmental impacts of a package. In contrast, a fee assessed at the retail level encourages consumers to change their buying habits. The potential effects of a packaging tax are to: 1) reduce packages manufactured, 2) reduce packages bought, 3) raise revenues, 4) stimulate demand for recyclables, and 5) promote the production of products made with recycled materials. The effectiveness of a packaging tax depends on a variety of factors, including where the tax is imposed, and how well it is enforced and obeyed. The only outcome on which states seem to agree is that a packaging tax will raise revenues; but how much revenue depends on the extent to which manufacturers: reduce their packaging, increase their production of recyclable products, and increase their use of recycled materials. For further discussion of packaging taxes, see Section 3, Solid Waste Product Charge.

Regulatory Initiatives

Standing in direct contrast to the subtle persuasion methods of economic incentives is the command and control approach of regulatory measures. Regulatory source reduction initiatives include bans, mandatory reusable bottles, product specifications, and minimum warranty requirements. Most regulatory measures force manufacturers or retailers to account for the social costs of their products. Unlike taxes -- which allow manufacturers, retailers, and consumers a choice -- bans offer one choice: obey the law or risk prosecution.

Most regulatory measures fit into one of two regulatory control categories: quantity control and quality control. Quantity controls aim to decrease the amount of packaging in the waste stream. A law requiring all grains to be sold in bulk is an example of quantity control. Quality controls aim to regulate the characteristics of the products that do enter the waste stream. For instance, the elimination of cadmium in all packages is an example of quality control.

A ban, depending on how it is defined, is either a quantity or quality control measure. Bans fall into two categories: outright or conditional. An outright ban prohibits the use of a material or package under all circumstances. A conditional ban requires a material or package to meet specific criteria, such as being recyclable or degradable.

Quantity Control

Quantity controls are typified by outright bans on specific materials or products. Outright bans are placed on products with social (environmental and human health) costs so high, they outweigh any social benefits. With respect to packaging, the intent of bans is to remove the environmental impacts of a product by eliminating the product. Whether or not an outright ban results in quantity reduction depends on the replacement product/material. Also, outright bans often result in toxic reduction, a quality control.

The recent wave of outright packaging bans started around the time Berkeley, California banned the use or sale of take-out food packaging manufactured with polystyrene blown with chlorofluorocarbons (CFC).[15] Polystyrene continues to be the object of many ban initiatives, as is polyvinyl chloride (PVC). Suffolk County, New York and Newark, New Jersey both banned all eating utensils and food containers made of polystyrene and polyvinyl chloride.[16] The Suffolk ordinance also bans the wrapping or packing of fast-food in non-biodegradable materials. Exemptions to Suffolk's ordinance include: 1) flexible packaging used to cover raw meat, poultry, and fish; cold cuts, fruits, vegetables, baked goods, and bread; 2) packaging used in hospitals and nursing homes; 3) plastic-coated paper packaging; and 4) any plastic covering food containers, eating utensils, and straws that are not made of PVC or polystyrene.

Other banned materials include: the plastic/aluminum can (Connecticut, Maine, and Minnesota); non-degradable plastic retail bags (Florida); packaging manufactured with CFCs (Florida, Maine, and Rhode Island); non-degradable polystyrene foam and plastic-coated paper (Florida); disposable polystyrene foam food and drink containers from public cafeterias (Maine; and Massachusetts and Vermont by Executive Orders); multi-material aseptic containers (Maine); multi-material food and beverage containers (Rhode Island); and non-degradable six-pack connectors (16 states).[17]

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[15]The first packaging ban was enacted in 1977 when Minnesota passed a law prohibiting the sale of milk in non-returnable/non-refillable plastic containers. The law was contested by the plastics industry, but was upheld by the Supreme Court. However, by the time of the Supreme Court ruling the dairy industry had already converted to plastics, and the law was never enforced.

[16]Resolution No. 1869-87, "A Local Law to Simplify Solid Waste Management by Requiring Certain Uniform Packaging Practices within the County of Suffolk." This bill was signed into law on March 30, 1988.

[17]Environmental Action Foundation, "Legislative Summary," 1988.

The typical reasons for state and local governments to ban a packaging material or product are because the product is not being recycled in that community or that the product has egregious environmental impacts (as defined by the initiator of the measure). Thus, the rationale behind the PVC and polystyrene bans are that these materials are difficult to recycle, are non-biodegradable, and have high environmental costs. The most frequently cited reason for banning polystyrene foam products was their use of chlorofluorocarbons (which are depleting the ozone layer). One of the reasons cited by Suffolk County for banning PVC packaging is that it produces toxic emissions when incinerated.

The effectiveness of bans in meeting their goal of source reduction is the subject of this and other research projects. The key issues raised by banning initiatives are: how industry responds (e.g., acceptance or court challenges), and how effective bans are (e.g., does a ban guarantee that the material/product banned is replaced with a less environmentally damaging one, and if so, how must they be structured to achieve this end?). Underlying each of these issues is how flexible bans are. In other words, are bans so rigid that they polarize industry and government interaction?

Mandatory reusable bottle laws, unlike container deposits, reduce packaging by changing product design. No state has a mandatory reusable bottle law and none are considering one. Typically, states pursue reduction in the disposal costs of beverage containers by passing container deposit laws. West Germany, however, has placed a high-deposit on non-reusable bottles. Although this is a financial incentive, the result has been the withdrawal of plastic soft drink bottles from store shelves.[18]

Quality control

Quality controls aim to regulate the type of solid waste produced by packaging products. Quality control measures include conditional bans, product specifications (design requirements), and placing minimum warranty requirements on durable goods. The most popular quality control initiatives are conditional bans.

Typical conditional packaging bans require a package to be either recyclable or degradable. The most comprehensive conditional ban was passed by Minneapolis and St. Paul in 1989. The Minneapolis law[19] bans all packaging sold in retail food establishments which is not "environmentally acceptable." Environmentally acceptable packaging is defined as being degradable (excludes degradable plastics), returnable, or recyclable. "Recyclable"

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[18] "Recycling in Other Countries," *Plastics Recycling Update*, March 1989, p. 6.

[19] Ordinance 89-Or-060, "Chapter 204. Environmental Preservation: Environmentally Acceptable Packaging," signed into law on March 31, 1989.

refers to a packages' ability to be recycled in a municipally sponsored program within Minneapolis. The five principal exemptions to the law are: 1) plastic utensils; 2) flexible packaging of a thickness of 10 mils or less; 3) plastic-coated paper packaging; 4) packaging used at hospitals or nursing homes; and 5) packages for which no commercial substitutes are available.

Another comprehensive conditional ban is the Massachusetts Public Interest Research Group's referendum initiative (scheduled for the November 1990 ballot). This referendum will ban all packages by 1995 which are not recycled in the state at a rate of 35 percent (50 percent by the year 2000), do not have a recycled content of at least 50 percent by weight, or are not reusable at least five times. Packages exempted from this initiative include: packages the Food and Drug Administration prohibits from containing recycled materials, packages for prescribed medications, and tamper-proof seals.

A distinction is often made between conditional bans and product specifications (design requirements). But, a product specification (e.g., a package must have 50 percent post-consumer recycled material content) is merely another term for a conditional ban; if the package fails to meet the recycled material content, it is banned. Thus, conditional bans and product specifications are considered one and the same in this study.

Minimum warranty requirements integrate environmental costs into a product by forcing corporations to make durable products with recycled material content. By making manufacturers offer minimum warranty requirements the government is forcing changes in the production process rather than accepting the negative impacts of the products and their manufacturing processes. Today, minimum warranty requirements are rarely discussed as potential source reduction measures, although in a 1977 EPA funded study, David Conn noted its potential use at achieving source reduction.[20] However, minimum warranty requirements are rarely applied to packaging products.

Trends

To facilitate the reduction, reuse, and recycling of packaging wastes, many states are creating packaging advisory committees or are promulgating regulations which empower state agencies to develop packaging guidelines. At least five states -- California, Connecticut, Illinois, Minnesota (the Select Committee on Packaging and the Environment, SCOPE), and Washington -- have created a packaging advisory committee or another similar entity. The northeastern states are also in the process of establishing preferred packaging guidelines (see Regional Initiatives below).

[20]W. David Conn, "Waste Reduction: Issues and Policies," *Resources Policy*, March 1977.

In Connecticut, included in law HB 6641 (passed June 1989), is a provision requiring that the commissioner of the Department of Environmental Protection develop specific packaging reduction measures. It also directs the commissioner to develop an education campaign to promote the use of reusable packages, recyclable packages, or packages with recycled content, and to discourage the use of non-recyclable, difficult to recycle, and virgin content packages. In California, law SB 1322 (passed September 1989) created a Source Reduction Advisory Committee to make recommendations on: packaging and product design improvements, methods for increasing recycled content, and methods for reducing toxicity.

By establishing packaging advisory committees, states are trying to bring coherence to the chaos surrounding the annual introduction of legislative initiatives to reduce packaging in the waste stream. Development of packaging guidelines requires a broad range of knowledge -- from the environmental impacts of a package to the economic and environmental impacts of a ban. Since much of this data is generic across the country, it has been noted that it would benefit states to consider coordinating their efforts and exchanging information through an agency such as the Council of State Governments or the EPA. The U.S. Congress is considering bills which would require the EPA to become a clearinghouse for information on packaging source reduction measures and the environmental impacts of packaging (see Federal Initiatives below).

1.3 REGIONAL INITIATIVES

In response to the high costs of solid waste disposal in the northeastern United States, the Coalition of Northeastern Governors (CONEG)[21] established a Source Reduction Task Force. In September 1989 the Task Force released their final report. The report makes two principal recommendations: 1) establishing a Northeast Source Reduction Council and 2) establishing preferred packaging guidelines. The Council would consist of a Steering Committee, Board of Directors, associate members, and standing committees. The goals of the Council are: 1) to provide industry with a "publicly credible and accountable basis" to undertake voluntary source reduction measures, and 2) to provide a "forum and deliberative process" to develop and recommend solid waste source reduction measures for the nine CONEG states.

The preferred packaging guidelines call for, in order of priority:

1) no packaging;

2) minimal packaging;

[21]CONEG member states are: Connecticut, Maine, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, and Vermont.

3) consumable, returnable, or refillable/reusable packaging; and

4) recyclable packaging/recycled material in packaging.

Also included within these guidelines is the reduction of heavy metals such as cadmium, lead, and mercury in packaging products. These guidelines are then to be developed through voluntary industry cooperation, public education, and legislative initiatives. No specific legislation (e.g., a tax, ban, or mandatory recycling law) is suggested for the source reduction of packaging materials, although the Council is responsible for developing a "detailed action plan to assure implementation of the preferred packaging guidelines." The only area for which legislation is specifically mentioned is the source reduction of heavy metals in packaging materials.

1.4 FEDERAL INITIATIVES[22]

State and local governments are currently filling the gap left by the absence of federal packaging reduction initiatives. Action in the U.S. Congress on packaging issues is centered around efforts to reauthorize the Resource, Conservation, and Recovery Act of 1976 (RCRA). Representatives and Senators have introduced bills to address solid waste problems, but none have become law. Sponsors of bills with packaging reduction include Sen. Max Bauchus (S. 1113), Sen. John Chafee (S. 1112), Rep. Thomas Luken, Rep. George Hockbruechner (H.R. 500), and Rep. Richard Gephardt (H.R. 1804). Other sponsors of solid waste bills include Rep. Collins (H.R. 1810), Rep. Esteban Torres (H.R. 2648), and Rep. James Florio.[23]

Senator John Chafee's bill would create an Office of Waste Reduction under the EPA's jurisdiction. The source reduction measures included within the bill are:

- technical assistance to state and local governments, and industry on source reduction and recycling methods;
- public education;

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[22]The primary source of information for this section is Environmental Action Foundations's report, *RCRA Reauthorization Legislative Summary: Federal Solid Waste Bills in the 101st Congress*, November 1989.

[23]The party affiliations and states of these Congressmen are as follows: Bauchus (D-Montana), Chafee (R-Rhode Island), Luken (D-Ohio), Hockbruechner (D-New York), Gephardt (D-Missouri), Collins (D- ?), Torres (D-California), and Florio (D-New Jersey; presently Governor of New Jersey).

- encouraging source reduction in manufacturing processes;
- reducing toxics in consumer products;
- establishing a Products and Packaging Advisory Board to minimize packaging, minimize use of scarce natural resources, eliminate the use of toxics, maximize reuse and recycling of packaging, and assure human health and the environment are not adversely affected by the use and disposal of packaging products; and
- banning the use of cadmium as a pigment and for all other non-essential purposes.

Senator Max Bauchus' bill would also establish an office under the EPA's jurisdiction: the Office of Waste Minimization. Source reduction powers that would be granted to the Office of Waste Minimization under S. 1113 include:

- establishing a waste reduction and recycling clearinghouse (section 304);
- developing a hazardous substances efficiency standard (section 305);
- establishing a National Packaging Institute (section 307) with the power to develop voluntary national packaging standards which address the design, composition, volume, reuse, recyclability, degradability and disposal of product packages and packaging materials used in consumer products; and
- identifying the use of hazardous constituents in products and promulgating regulations for their proper disposal (section 310).

Rep. Thomas Luken's bill would establish an independent National Packaging Institute with powers similar to the organizations proposed by Senators Chafee and Bauchus. It would also require the EPA to ban the five most common toxics in solid waste and to ban them from production or from disposal in landfills or incinerators. Rep. Gephardt's bill would also establish a clearinghouse, public education program, and a program to reduce toxics in solid waste.

The only bill with provisions to ban packaging materials/products for reasons other than toxic content is Rep. George Hockbruechner's H.R. 500. The two principal packaging provisions of H.R. 500 are: 1) to establish an office of recycling, research, and information; and 2) to ban all non-biodegradable or non-recyclable packaging materials.

The only federal solid waste bill that proposed the use of tax incentives was Rep. James Florio's (Florio is now governor of New Jersey). His bill proposed that "qualified recycling facilities" be eligible for tax-exempt bonds, and that "qualified remanufacturing equipment" be eligible for a 15 percent investment tax credit.

The federal bills, in general, rely on four methods to reduce the quantity and toxicity of packaging products: 1) public education, 2) specific measures to reduce toxics, 3) mandating the EPA to assist states, and 4) creating Packaging Institutes/Boards to make packaging source reduction recommendations. The most aggressive source reduction measures (such as Rep. Luken's proposal to ban the five worst toxics from at least landfills and incinerators) concern toxic substances in solid waste. Most of the bills contain voluntary measures advocating technical assistance to state governments, having the EPA serve as a clearinghouse of information, and promoting public education campaigns. Given the initiatives emanating from the U.S. Congress, it seems likely that state and local governments will continue to develop the most aggressive and comprehensive measures to source reduce packaging. Since the federal government is not responsible for disposing of solid waste, this is not surprising. The federal government, however, may want to develop a federal packaging source reduction policy that will provide manufacturers with a nationally consistent set of policies to follow.

1.5 INTERNATIONAL INITIATIVES

As opposed to the limited efforts by the United States Government to reduce packaging wastes, many European nations have enacted national regulations governing the use of packaging materials, especially non-returnable/non-reusable beverage containers (see Table 2). The federal governments in Denmark, Finland, the Netherlands, and Switzerland all have broad powers to minimize packaging waste, but only Denmark and Finland have put these powers to use.

In Denmark, the Minister of Environment has authority to regulate beverage and milk containers in accordance with Act No. 297 (June 1978), which is designed to encourage the reuse of paper and beverage containers. The law allows the Minister to ban certain products or materials; to define secondary content required in paper products; to assess packaging deposits; and to place the responsibility of collection of reusable *materials* on municipalities.[24] As a result of this law and earlier laws, the Minister banned one-way carbonated beverage packages in 1977 and one-way beer packages in 1981.[25] Whenever a company introduces a new beverage package the company must demonstrate to the Minister that the package is necessary, is compatible with existing return systems, and will be returned and refilled.

[24]K.H. Garmin, "Packaging Legislation in Denmark" (Boustead and Lidgren, eds., *Problems in Packaging: The Environmental Issue*, New York: John Wiley & Sons).

[25]Ibid.

Table 2. Measures to Promote Reuse or Recycling of Beverage Containers in Europe, 1988

Country	Beverage Initiatives	
Denmark	*Ban*	on all non-returnable/non-reusable soft drink and beer containers (1981).
Finland	*Packaging Legislation*	empowers the government to restrict, ban, or tax packaging which causes significant costs or difficulties in disposal.
	Tax	on all non-returnable/non-reusable beer and soft drink containers.
France	*Agreement*	signed by industry and government to reduce packaging energy use and waste through research and development programs, increased private recycling initiatives, and promotion of returnable containers.
Germany	*Consumer Aid*	Blue Angel label used to mark environmentally sound products.
	Restriction	plastic bags must be produced with 1/3 less energy and less air pollution.
	Voluntary Agreement	agreed upon by government, industry, and trade organizations to limit waste from beverage containers by encouraging the use of redeemable deposit containers, and increase glass and tin can recycling.
Italy	*Ban*	all bags and packaging must be either recyclable or biodegradable.
Netherlands	*Packaging Legislation*	Minister of Health and Environment has the authority to impose taxes, bans, or deposits on all beverage containers.
	Private Initiative	to increase recycling of glass containers.
	Recycling	PET bottles can be sold only if 70 to 80% are returned for recycling.
Norway	*Ban*	on non-returnable beer and soft drink containers.
	Restriction	all returnable containers must be of certain size, weight, and color.
	Tax	on all non-returnable containers.
Sweden	*Private Refund System*	on all beer and soft drink containers.
	State Refund System	on all wine and liquor containers.
	Tax	on all beer and soft drink containers, but money goes into the general revenue fund.
Switzerland	*Packaging Legislation*	can require those selling beverages to take them back, and can ban or restrict non-recyclable containers.
	Private Refund System	on all beer, soft drink, and wine containers.

Source: F. Lieben, "Measures to Promote Reuse or Recycling of Beverage Containers; and Environmental Action Foundation, "Solid Waste Legislative Database," August 1988.

In Finland, through legislative powers and state-control of retail beer stores, the state limits sales of non-returnable containers to 10 percent of total sales. This is accomplished by increasing the price of non-returnable beer containers relative to the price of reusable containers; in 1984, beer in cans cost 38 to 55% more than in returnable bottles.[26]

The Netherlands and Switzerland both have the legislative power to severely restrict the use of non-returnable beverage containers, but they are reluctant to use these powers. For example, in 1977 the Netherlands passed a refuse law (Afvalstoffenwet) that empowered the Minister of Health and Environment to ben, tax, or impose deposits on products which are difficult to reuse or recycle, and which cause litter or solid waste problems.[27] But to deflect the movement towards restrictive packaging legislation, the packaging industry in the Netherlands instituted an aggressive glass recycling campaign. In general, the packaging industry's strategy has worked, although in 1987 the Netherlands passed a law banning PET bottles unless they are recycled at a rate of 70 percent or more.

Beverage packaging is the primary target of European packaging legislation because it is easy to regulate, is an area with a clear set of available substitutions, and is subject to public exposure (e.g., on shelves at the store and at home, or as litter). Italy, however, has enacted legislation which addresses all packaging types. The law, issued in 1987 by Executive Order, requires that all bags and packaging materials be either biodegradable or recyclable (beginning in 1989).

In contrast to U.S. packaging legislation, European legislation is national in scope and is usually targeted at non-returnable/non-refillable beverage packaging. The Italian law is the only one to address the role of packaging in all areas. The European packaging legislation highlights the importance of beverage packaging as a section of the waste stream, the potential for returnable beverage systems, and the need to address packaging issues on a national scale.

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[26] Jorma Hamalainen, "An Industrial View of the Problems of Public Intervention in the Field of Packaging in Finland" (Bousted and Lidgren, eds., *Problems in Packaging: The Environmental Issue* New York: John Wiley & Sons)

[27] A. Th. van Ewijk, "Present and Impending Regulations Concerning Packaging and the Environemtnal in Netherlands" (Bousted and Lidgren, eds., *Problems in Packaging: The Environmental Issue,* New York: John Wiley & Sons).

1.6 CONCLUSION

This survey of packaging reduction initiatives has identified extensive activity on the state and local level by governments faced with high waste disposal costs. Reflecting these high costs, the most significant legislation is emerging in the more industrialized and densely populated states. Exceptions to this rule are Florida, Oregon, and Minnesota. Florida is unique because of its high water table which makes landfilling of wastes difficult. Oregon and Minnesota are known for taking aggressive actions to reduce the environmental impacts of solid waste disposal.

Two important variables of each packaging initiative, which affect political feasibility, as well as environmental and economic efficiency, are: the point(s) in the product's lifecycle the measure targets and the scope (number of products affected) of the initiative. In general, voluntary initiatives target consumers, financial initiatives target both consumers and manufacturers, and regulatory initiatives target manufacturers. On a spectrum of political feasibility, voluntary initiatives are typically the most palatable to industry (easiest to enact) and regulatory initiatives are the least palatable to industry (hardest to enact), with financial initiatives in the middle. However, in the case of packaging legislation, state and local governments have found it easier to pass regulatory initiatives -- primarily conditional and outright bans -- than to pass financial initiatives.

The more comprehensive bans (Minneapolis/St. Paul, Newark, and Suffolk County), however, are being enacted by municipal and county governments; whereas state-wide packaging taxes and comprehensive bans have not been passed into law. Local governments also have the best incentive to pass stringent regulatory packaging initiatives: they must dispose of the garbage; and as disposal costs approach $100/ton, towns are trying to reduce waste through all methods. The further removed are government bodies from solid waste disposal, the more conservative they become in their approaches to source reduction.

Although outright bans are an aggressive approach to source reduction (usually quality control), they pose a problem by being too product-specific. For example, one product which has generated significant controversy and is banned in a few states is the "plastic" can. Yet the packaging industry could conceivably develop an even more environmentally egregious packaging product and the only recourse to states would be to then ban that product as well. Thus, the most comprehensive bills are the packaging taxes which cover the majority of packages, and conditional bans similar to Minneapolis', which delineate between environmentally acceptable and non-acceptable packages. Conditional bans, as compared to outright bans, provide governments with a much greater level of certainty in regards to how industry will react to the new law in terms of what packaging products they will produce.

Whether or not packaging taxes will have the intended effect of reducing the amount of packaging produced, and/or encourage the use of more environmentally benign packaging has not yet been demonstrated. Confounding this problem is the issue of not knowing exactly what is a more "environmentally benign" package. The purpose of this research project is to address both of these issues.

Section 2 reviews the existing research on environmental assessments of various packaging components, and the final section of this report reviews the existing research on the economic impacts of packaging taxes.

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SECTION 2 PACKAGING AND THE ENVIRONMENT

In this section we review and evaluate six studies that have examined the environmental impact and energy requirement of various packaging materials. Five of these studies were carried out in the late 1970's to early 1980's when more extensive federal funding for solid waste research was available. Since technologies have changed since this time period, the usefulness of data contained in older reports is limited. However, these reports are still valuable as they provide a framework for analyzing the environmental impacts of packaging material and provide examples of the methodologies which may be useful for our study.

In Section 2.1 two studies which examine the energy impacts associated with packaging materials are reviewed. The first study presented is broad in scope, examining many types of plastic products including packaging, while the second study examines only packaging used by consumer goods. Section 2.2 reviews a study that analyzes the environmental and energy impacts of non-fluid food packaging and Section 2.3 reviews three studies that examine beverage containers.

2.1 ENERGY IMPACTS OF PACKAGING MATERIALS

The two studies reviewed in this section explore the energy required for the manufacturing of packaging materials. The first study approaches this question by quantifying the energy required for manufacturing many types of plastic materials including packaging and then compares this requirement to the energy needed for the manufacture of substitute materials. The second study solely examines packaging materials.

"Total Energy Impacts of the Use of Plastics Products in the United States," prepared for The Society of the Plastics Industry by Franklin Associates, January 15, 1981

The objective of this study is to examine the energy impacts associated with the manufacture of products made from plastics. Twelve resins are examined: low density polyethylene (LDPE), polyvinyl chloride (PVC), polystyrene, high density polyethylene (HDPE), polypropylene, acrylonitrile-butadiene-styrene (ABS), styrene-acrylonitrile (SAN), nylon, polyester, phenolics, urea, and melamine. The analysis includes 264 end uses of these resins, including certain packaging materials. This report defines energy impacts as the difference between the energy consumed for the manufacture and use of plastic products and the energy consumed for manufacture of non-plastic alternatives.

Consumption of plastic resins by various domestic end uses is quantified by weight. Non-plastic alternative systems are then identified for each end use and the weight of the alternative materials is estimated. Energy profiles, quantifying energy required for raw material extraction, processing, final product formation, and transportation during all these steps, are developed for each plastic material and for each alternative. Transportation of the final product is not considered. In cases where energy sources are used as a material feedstock, the energy content of the material is included in the energy profile. All the above data are then used to quantify the amount of energy (in Btu) necessary to produce 1,000 pounds of the finished product. For evaluating energy use by plastic alternatives, the total "system" of each alternative is analyzed. For example, in comparing disposable diapers with cloth diapers, the energy required to wash and dry the cloth diaper is considered.

To compare the energy use of plastic products and their alternatives, the difference in weight between materials is adjusted. In cases where the plastic product is substituted by only one alternative, the calculation simply involves the product of the energy use per 1,000 pounds of product and the weight of material that replaces the plastic. When more than one substitute is available, the calculation becomes more complicated. First, market share of each substitute is estimated. For example, three alternatives to PVC siding on homes are aluminum siding, wood shingles, and steel siding. To compare the energy impacts of these three materials to vinyl, the market share of vinyl is allocated among the alternatives based upon current and historical market data. For each substitute, the energy use per 1,000 pounds of finished product and the weight of the substitute material is calculated. The energy requirements for all plastic alternatives are then summed in order to compare the energy required for manufacturing plastic products and their alternatives.

The findings of the report indicate that replacing plastic products with their non-plastic alternatives increases energy consumption by 834.2 trillion Btu. Several examples are given which depict how the use of plastics reduces energy consumption. For instance, the use of plastics in automobiles reduces weight, which in turn reduces gasoline consumption. Disposable diapers consisting of polyethylene film reduce energy consumption since their alternative, cloth diapers, require energy for washing and drying.

While this report deserves credit for undertaking such a broad task and compiling a large volume of data, there are some problems with the methodology used in the report. The unit of comparison for plastic and nonplastic products is the amount of Btu required to manufacture 1,000 pounds of product. However, this is not an adequate basis of comparison for all products. For example, in the production of diapers, energy is required to wash and dry cloth diapers; however, the diapers are not disposed of, so the substitution of disposable diapers to cloth diapers is not one to one. This causes an overestimation of the energy advantages of disposable diapers.

There are some data in this report that may be useful for our study. Energy requirements for production of packaging materials such as plastic film, aluminum foil, and milk bottles, to name a few, are provided. These data may have to be updated since this report is eight years old and technological changes during this time span may have affected energy requirements for production.

"Energy and Materials Use in the Production and Recycling of Consumer-Goods Packaging," prepared for U.S. Department of Energy by Argonne National Laboratory, February 1981

This report summarizes the results of a series of reports by Argonne that details the energy and materials flows in energy intensive industries. The objective of the study is to determine the amount of energy consumed in the production of paper, glass, steel, aluminum, and plastic packaging (see Table 2.1) and to examine reductions in energy use through material substitution, reuse, recycling, and energy recovery through incineration. This study encompasses energy used both for fuel and raw material during each stage of production and characterizes the total energy used to produce a pound of packaging material. This methodology is carried out both for virgin and recycled materials. Fuel types used to make each type of packaging material are quantified and the effect of recycling and incineration is assessed both on the total amount of fuel consumed and the types of fuel consumed. This latter impact makes this study unique.

The core of this report examines the energy required to manufacture various packaging materials and addresses packaging policies that affect total energy use and use of specific fuel types. Examples of this approach are given below.

Paper and plastics compete in several markets for consumer packaging. Depending upon the type of paper packaging produced, the range of energy requirements is 16,050 to 33,400 Btu/lb. Wood provides 75% of this energy input, 7% is provided by electricity and oil and natural gas provide the remaining 18%. The manufacture of plastic packaging materials varies according to the type of resin used. PVC is at the low end of this range - 25,600 Btu/lb and PET is at the high end - 48,700 Btu/lb. More than 50% of this input is the energy content of the feedstock.

TABLE 2.1

ENERGY REQUIRED TO MANUFACTURE PACKAGING MATERIALS

Material	Form	Annual * production (million lb)	Percentage used for packaging	Energy required (trillion Btu)
Paper	Rolls or liner stock	120,000	43	1260
Glass	Containers	30,000	100	261
Steel	Sheet	210,000	6	287
Aluminum	Sheet	13,000	18	281
LDPE	Pellets	7,800	62	186
HDPE	Pellets	5,000	45	82
Polystyrene	Pellets	4,000	36	49
Linear LDPE	Granules	1,000	80	26
PET	Pellets	4,200	7	14
PVC	Pellets	6,100	7	11

* **NOTE:** No date is given. These data are presumably from the late 1970's.

While plastics require more energy per pound than paper, paper products are heavier. The weight ratios of paper to plastic is 1.4-2.8 for bags, 1.8 for milk containers, 0.8-1 for cups, and 3.7 for meat trays. Adjusting for this factor, the author recommends that plastic bags, milk containers, and meat trays should be used to minimize total energy inputs but that paper rather than plastic cups should be used. However, if a policy decision is made to reduce oil and natural gas use then paper products are favored over plastics since less oil and natural gas is required for paper manufacture.

Recycling of paper and plastic products presents another set of policy decisions. Paper recycling requires 35% of the energy used for making new paper, approximately 11,500 Btu/lb, but requires a 4,000 Btu/lb increase in purchased energy (oil and gas) as compared to virgin paper. Plastic requires 1,000 Btu/lb to be remelted and various resins must first be separated to produce a high-quality product. The author therefore concludes that for policies designed to minimize oil and natural gas, the use of virgin paper which relies mostly on wood for energy may be preferable to recycling paper or plastic.

Glass, aluminum, and steel compete for the singe-serving beverage container market. Glass manufacturing requires approximately 8,700 Btu/lb with more than 70% of this energy supplied by natural gas and 10% by electricity. Steel production from virgin ore is more energy intensive than glass, requiring 22,800 Btu/lb. Coal supplies 50% of this energy, electricity 10%, and oil and natural gas provides the remaining 40%. Aluminum manufacturing is an even more energy intensive process; 120,000 Btu/lb are required for virgin aluminum sheet. Fifty percent of this energy is provided by electricity and the remainder is provided by oil and natural gas. Thus, when containers are used only once, glass bottles are the favored container as they require the least amount of energy per use. Glass reuse provides yet a greater energy savings, even though the manufacture of the heavier glass used for refillables requires more energy. This energy saving is realized since the only energy required is for transporting and washing the glass.

Recycling of glass, aluminum, and steel provides energy savings. Glass recycling provides a 25% energy savings as compared to manufacturing from raw materials. Steel recycling provides a 30% energy savings while aluminum recycling provides an 84% energy savings. Even with the large energy savings that aluminum recycling provides, if beverage containers are only recycled one time, producing beverage containers from recycled glass and steel requires less energy since aluminum production is an energy intensive process. However, as the recycling rate increases for these three containers, the energy requirements become more similar.

The report concludes that several factors should be considered when analyzing the energy impacts of packaging materials. In addition to the energy requirements to manufacture these materials, the potential for recycling and energy recovery should also be considered.

2.2 IMPACTS OF PRODUCTION OF NON-FLUID FOODS PACKAGING

The study reviewed in this section examines the energy and environmental impacts of production of packaging materials used for non-fluid foods. Of all the studies reviewed, it is the broadest in scope. As a result its treatment of detailed energy and environmental impacts is relatively limited.

"The Application of Technology-Directed Methods to Reduce Solid Waste and Conserve Resources in the Packaging of Non-Fluid Foods," prepared for National Science Foundation by Franklin Associates, February 1978.

The historical trend of packaging used for non-fluid foods was quantified in this study for the time period from 1960 through 1975. In 1975 packaging represented 33% of the total municipal solid waste stream with non-fluid food packaging accounting for 30% of all packaging waste (or nearly 10% of the total waste stream). Thus, packaging is a significant component of the waste stream. The methodology and findings of this study are summarized below.

This study was conducted by obtaining time series data for 14 non-fluid food packaging types and for 10 non-fluid food groups during the 16-year time span. The 10 food categories include: baked goods; cereal, flour, and macaroni; meats, poultry, fish and seafood; candy and chewing gum; non-fluid dairy products; frozen foods; canned fruits and vegetables; produce; fats and oils; and other foods. The 14 consumer and shipping packaging types include: folding boxes, set-up boxes, specialty paper, molded pulp, composite cans, metal cans, aluminum foil, rigid plastic, flexible plastic, glass, corrugated containers, paper shipping sacks, wirebound boxes and shook, and nailed wooden boxes.

To derive time series data for food delivered to consumers, government documents and private data sources were referenced. Agricultural food production was traced for each of the 16 years to determine time series data for the 10 food groups. For determination of packaging time series data, company and trade association experts provided information on the 14 types of packaging materials used for the 10 food groups.

Analysis of trends in weight of packaging by food groups showed weight decreases for some categories. For example, packaging weight for meat, poultry, and seafood has decreased as plastic foam trays have replaced molded pulp trays. Changes in can manufacturing have resulted in weight reduction for canned foods. On the other hand, certain food categories have experienced a weight increase as the use of glass has increased. Shipping packaging also increased during the 16 year time span. This increase can be partially accounted for by the increase in centralized food processing centers resulting in longer shipping distances.

The trends in packaging by packaging material was also analyzed. Several categories exhibited increases in use by weight including glass, composite cans, rigid plastics, flexible plastic, aluminum foil, and set-up boxboard. Specialty paper, molded pulp, and metal cans experienced a decrease in use by weight due to changes in the type of materials used for packaging certain food categories.

These food and packaging data were converted to pounds of packaging per pound of food. Food consumption for each of the 10 food groups was determined for the 16-year time span. The weight of packaging used for each of these categories was divided by the food weight for each of the respective years to derive the weight of packaging required per pound of food. The results show that total packaging weight (consumer plus shipping packaging) per 1,000 pounds of food consumed increased during the 16-year period with the largest growth occurring both for corrugated and glass packaging. When these data are broken down into consumer packaging and shipping packaging, it is evident that this growth in packaging weight is due to increased shipping packaging while the amount of consumer packaging has decreased.

The weight of packaging per capita and weight of packaging per dollar food expenditure were also determined in this study. The weight of packaging per capita, an estimator of the quantity of packaging associated with a consumer's purchases, increased during the study time period but this increase could also be attributed to an increase in shipping packaging. A slight decrease was found for the weight of packaging per dollar spent on food.

The impact of packaging materials on solid waste and natural resources was assessed in this study. Time trend data were developed for the non-fluid food packaging component of solid waste based both upon weight and volume. From 1960 to 1975 the amount of packaging waste by weight per pound of food decreased slightly with the consumer packaging weight decrease offsetting the shipping packaging weight increase. By volume, the amount of packaging waste again increased with a decrease in consumer packaging volume and an increase in shipping packaging volume.

The effect of the use of packaging materials on natural resources including fossil fuels and minerals was evaluated. The authors found a decrease in the amount of minerals industry output being used for non-fluid food packaging over time. Energy requirements were not included in this scope.

The final factor affecting packaging trends that is assessed by this study is technological innovations, with most innovations causing a reduction in packaging weight. Reduction in metal can packaging weight has been achieved by using thinner metal and reducing the amount of seams in the can. A change in process allows production of lighter weight glass bottles. Numerous other examples are provided.

This study is interesting in that presents the changes that have occurred to the packaging industry through time. It shows the researcher that this area is not static and that such changes should be accounted for when projecting future impacts of packaging material. At the same time, these data are more than a decade old and thus one can assume that many such changes have occurred during this time interval as well.

2.3 IMPACTS OF PRODUCTION OF BEVERAGE CONTAINERS

Three studies are examined in this section. The first two studies examine the impacts of production of various soft drink containers while the last study examines packaging materials used for milk. In contrast to the broader studies considered above, the focus on beverages alone allows these studies to examine a crucial packaging policy question in great detail.

"Family-Size Soft Drink Containers - A Comparative Energy and Environmental Impact Analysis," prepared for Goodyear Tire and Rubber Company by Franklin Associates, January 1978

This study was performed during a time period when PET bottle manufacturers were attempting to increase their market share. Thus, the objective of this study is to compare the environmental impacts and energy requirements of manufacturing PET bottles and other containers used for soft drinks. The beverage containers included in this study are 12 ounce aluminum cans; 12 ounce bimetal cans; 12 ounce all-steel cans; 16, 32, and 64 ounce nonrefillable glass bottles; 2 liter nonrefillable plastic-coated glass bottles; 16 and 32 ounce and 2 liter refillable glass bottles, and two types of 1 and 2 liter PET bottles.

Use of resources, including energy, and environmental impacts are assessed in each step beginning with raw material extraction and ending with final disposal or recycling. Environmental impacts that are quantified include air pollution, water pollution, and solid waste generation. Energy requirements for plastic include both the energy content of the fuels used as raw materials and fuels used in manufacturing. Impacts are quantified on a basis of 1,000 pounds of output and then converted to 1,000 gallons of delivered soft drink.

The data developed in this report are based upon two prior reports, "Resource and Environmental Profile Analysis of Nine Beverage Container Alternatives," by Midwest Research Institute for EPA and "Energy and Economic Impacts of Mandatory Deposits," by Research Triangle Institute and Franklin Associates for Federal Energy Administration. The data from these reports were updated from a variety of sources including reports and interviews with industry sources.

The energy and environmental impacts of the four types of PET bottles are compared to the other beverage containers and to each other. The PET bottles differ in size, and in type of base - free-standing versus base cup. In addition, since marketing of PET bottles was in the early stages at the time of this study, several methods of packaging the bottles for transport were examined. For the various PET bottles, the 2 liter free-standing bottle without corrugated packaging has the lowest impacts for all categories.

The results reported by Franklin are that PET produces less impact in each impact category as compared to the various nonrefillable beverage containers. For refillable containers, the impacts of PET are compared for various return trip rates. At lower trip rates, PET has lower impacts; at higher trip rates glass produces less impacts. The breakeven point for these two types of containers is at a return trip rate of 4 to 6.

"Comparative Energy and Environmental Impacts for Soft Drink Delivery Systems," prepared for National Association for Plastic Container Recovery by Franklin Associates, March 1989.

The purpose of this study is to compare the energy consumption and environmental impacts of nine soft drink containers and their packaging components (i.e. labels, closures, wraps). The containers examined include four sizes of PET bottles - 16 ounce, 1, 2, and 3 liters; 12 ounce aluminum can; and 4 types of glass bottles - 10 ounce nonrefillable, 16 ounce nonrefillable, 16 ounce refillable, and 1 liter nonrefillable. Energy requirements, both for manufacturing and transportation, and environmental impacts (air, water, and solid waste) are assessed in a cradle to grave fashion, starting at raw material extraction and ending with final disposal. Thus for fuels, the energy needed to extract, process, and transport each fuel type is calculated. For plastics, both the energy content of the fuels used as raw materials and the fuels used in the manufacturing processes are included in energy requirement calculations. Air and water pollution impacts are reported in tons produced per 1,000 gallons of delivered beverage for each beverage container even though the nature of the impacts may differ. The authors acknowledge that "no attempt has been made to determine the relative environmental effects of these pollutants." These data were developed for a base year of 1987 and projected to 1990 and 1995.

In order to calculate energy consumption and environmental emissions created by soft drink containers, a materials balance was determined, with outputs measured by weight. This was then translated into unit weight of packaging material which in turn was converted to volume. Projections for 1990 and 1995 were generated by making assumptions for industry conservation measures and changes in packaging that will result in lighter weight containers. These impacts were calculated for virgin materials and recycled materials where the definition of recycling is "containers are manufactured from recovered used container material." Rather than predicting future recycling rates, the authors calculated energy and environmental impacts for all containers at recycling rates of 0, 25, 50, 75, and 100 percent.

The definition of recycling is contradictory throughout the report. For instance, the above definition would warrant the conclusion that the recycling rate for PET bottles is 0% since old PET bottles cannot be used to manufacture new PET bottles. Instead, Franklin Associates uses a 20% current recycling rate for PET. They justify this figure by the fact that PET can be used as a postconsumer fiber. Therefore, "the PET bottle can be compared to a 2-trip container (or, in other words, a container with a 50% recycling rate). In this example the first use is as a container, and the second use is a fiber." Yet, this philosophy is not applied to glass. According to the report, recovery of beer and soft drink glass containers is approximately 20%. Some of the recovered glass is manufactured into food containers rather than beverage containers. In addition, some food containers are recycled into beverage containers. Thus, they claim, only 10% of postconsumer material ends up in glass container production.

Another problem with this study is that only summary data are reported. The raw data are reported in an appendix which Franklin Associates is not releasing to the public. It is therefore impossible to verify the methodology used in the study or to apply these data to our study.

The report concludes that, based on current recycling rates, the 16 ounce refillable glass bottle is the most energy efficient followed by the various PET bottles. Air and water emissions are lowest for the 16 ounce refillable bottle and the 3 and 2 liter PET bottles. By volume, PET containers contribute the least amount of solid waste. These data are summarized in Tables 2.2 and 2.3.

"Resource and Environmental Profile Analysis of Five Milk Container Systems," prepared for Environmental Protection Agency by Midwest Research Institute and Franklin Associates, August 17, 1977.

A resource and environmental profile analysis is presented for five milk container systems in this study. The container systems include the refillable glass bottle, refillable HDPE plastic bottle, nonrefillable HDPE plastic bottle, nonrefillable paperboard carton manufactured from bleached kraft paper and coated with LDPE, and nonrefillable LDPE plastic pouch. Three container sizes are included for each system - gallon, half-gallon, and half-pint. The process analysis includes raw material extraction, processing, production of intermediates, production of the final product, and reuse, recycling or disposal of the final product. As this is the most detailed report found in our literature review, we provide a more extensive review of the report.

TABLE 2.2

ENERGY USE FOR THE DELIVERY OF SOFT DRINKS IN SELECTED CONTAINERS AT 1987 RECYCLING RATES (Million Btu per 1,000 gallons)

Containers	Total Energy
16 oz PET	31.59
1 L PET	25.50
2 L PET	18.86
3 L PET	18.55
12 oz aluminum can	32.94
10 oz nonrefillable glass	41.66
16 oz nonrefillable glass	34.82
16 oz refillable glass	15.43
1 L nonrefillabe glass	36.68

TABLE 2.3
ENVIRONMENTAL IMPACTS FOR THE DELIVERY OF SOFT DRINKS IN SELECTED CONTAINERS

	Air Emissions (pounds)	Water Emissions (pounds)	Solid Wastes (pounds)	Solid Wastes (cu. ft.)
<u>Containers Manufactured from Virgin Raw Material</u>				
16 oz PET	98.7	16.6	939.7	56.2
1 L PET	78.9	13.6	687.9	42.9
2 L PET	59.0	10.3	478.9	29.0
3 L PET	57.4	10.4	463.8	28.1
12 oz aluminum can	137.0	44.1	1938.0	40.4
10 oz nonrefillable glass	189.6	20.7	5725.7	117.4
16 oz nonrefillable glass	157.0	16.9	4721.2	96.9
16 oz refillable glass (1 trip)	271.5	24.8	9066.3	184.4
1 L nonrefillable glass	172.1	17.5	5354.6	110.1
<u>Containers Manufactured at Current Recycling Rates</u>				
16 oz PET	92.3	15.9	814.6	46.1
1 L PET	74.1	13.1	592.1	35.1
2 L PET	55.8	10.0	415.1	23.9
3 L PET	54.2	10.1	403.3	23.0
12 oz aluminum can	91.7	26.9	1068.1	21.5
10 oz nonrefillable glass	183.8	20.4	5273.2	109.2
16 oz nonrefillable glass	152.0	16.6	4347.6	90.2
16 oz refillable glass (8 trips)	53.8	8.2	1505.5	29.7
1 L nonrefillable glass	165.2	17.2	4915.7	100.9
<u>Containers Manufactured from 100% Recycled Materials or Reused</u>				
16 oz PET	66.9	13.4	363.6	8.5
1 L PET	54.6	11.3	232.5	4.9
2 L PET	43.0	8.8	176.6	3.7
3 L PET	42.5	9.1	173.3	3.6
12 oz aluminum can	46.3	9.7	198.2	3.2
10 oz nonrefillable glass	130.0	17.0	1198.4	19.4
16 oz nonrefillable glass	107.7	13.8	985.2	16.2
16 oz refillable glass (20 trips)	37.9	6.4	521.3	8.8
1 L nonrefillable glass	102.3	14.0	965.6	13.9

A detailed resource and environmental profile for each container system is presented. Raw material, water, and energy inputs were quantified for each operation and expressed in terms of 1,000 gallons of milk delivered. Several types of environmental outputs expressed as pounds of pollutant per 1,000 gallons of milk delivered were quantified. Emissions reported are emissions released after pollution control technology has been applied, assuming emissions are in compliance with 1977 EPA guidelines. There is no attempt to distinguish between various pollutants, although the authors acknowledge that the potential to cause harm differs between pollutants. Industrial solid waste impacts include process losses, fuel combustion ash residue, and mining wastes. If the container is refillable, these impacts are divided by the return trip rate.

Several assumptions are made in this report. The energy content of oil and natural gas feedstocks are accounted for in the energy required for production but are not included as a raw material. No preferential treatment is given to renewable resources or to the biodegradability of the product as a postconsumer waste. However, the compactability of the postconsumer waste is considered. It is assumed that plastic containers retain 65% of their volume and that paper cartons retain 22% of their volume when landfilled. Finally, for materials which are less than 5% of the container by weight, the environmental impacts were not assessed. However, these materials were accounted for as raw materials.

An interesting accounting approach is used for marketable by-products formed during the manufacturing process. Material inputs and environmental outputs are adjusted to reflect only that portion attributable to the product of interest. For instance, if manufacturing a 1,000 pounds of a product results in the production of 500 pounds of a useful by-product, material inputs and environmental outputs would be reduced by one-third (or 500 out of the total 1,500 pounds of output). However, this causes an artificial reduction in the inputs needed for manufacturing the packaging material.

Several types of air pollutants were quantified including particulates, nitrogen oxides, hydrocarbons, sulfur oxides, carbon monoxide, aldehydes, other organics, chlorine, sulfur compounds, ammonia, hydrogen fluoride, lead, and mercury. Twenty-three water pollutant emissions are quantified including BOD, COD, suspended solids, dissolved solids, oil, fluorides, phenol, sulfides, acidity, alkalinity, metal ions, ammonia, and cyanide. Postconsumer solid waste was measured by the volume of the container being disposed assuming that 9% of the waste is incinerated and 91% is landfilled. Data for these impacts were collected from industry and government sources. The findings for each beverage container are summarized below.

For glass container manufacturing, eight processes are quantified: sand mining, limestone mining, feldspar mining, natural soda ash mining, lime manufacture, glass container manufacturing, corrugated packaging, and transportation. When impacts are calculated for the manufacture of 1,000 pounds of glass bottles, container manufacturing and corrugated packaging represent the largest environmental impacts of all processes.

The only impact category which is not dominated by these two processes is postconsumer waste, where the container naturally dominates, and industrial solid waste, where feldspar mining wastes present the largest impact. Container manufacturing and corrugated packaging also dominate the total energy required for manufacturing glass containers with the former requiring 72% and the latter requiring 11% of the total energy.

Impacts of 1,000 gallons of milk delivered in gallon, half-gallon and half-pint refillable glass bottles are also presented. It is assumed that these bottles have a reuse rate of 20 trips. The impacts are presented for the glass system which includes the manufacturing of the container, transportation of the glass bottles, washing and filling, delivery of the containers to the store and home, postconsumer solid waste system, and the return of used bottles. In general, bottle manufacturing and filling and washing are the two processes which present the highest impacts. Table 2.4 summarizes the impacts for each size bottle.

The impacts of delivering 1,000 gallons of a one gallon 50-trip plastic bottle system are quantified for the nine system processes: HDPE resin system, manufacturing the bottle, the bottle packaging, the paper closure system, bottle washing, bottle filling, transportation of the bottle, transportation of the filled bottle to the store or residence, and bottle disposal. The results of this analysis show that the paper closure system uses the most raw materials; while the bottle can be reused 50 times, each refilling requires a new closure. The delivery of the milk presents the largest energy requirement - 47.3% of the total system energy. The next largest energy requirement is HDPE manufacturing which uses 18.2% of the total system energy. Bottle washing presents the highest water usage, industrial solid waste impact, and waterborne wastes while milk delivery is the process with the greatest air emissions. The half gallon and half pint 50-trip container systems are qualitatively similar to the one gallon system. Quantitatively the systems differ, with the gallon system impacts being smaller than the half gallon and half pint systems due to the greater container weight per unit volume for these smaller containers. These impacts are summarized in Table 2.5.

TABLE 2.4

IMPACTS FOR 1,000 GALLONS OF MILK DELIVERED IN REFILLABLE GLASS BOTTLE

Impact Category	Gallon	Half Gallon	Half Pint
Energy (mil Btu)	2.90	3.70	8.61
Water (thou gal)	0.64	0.76	1.84
Industrial solid waste (cubic ft)	0.60	0.78	1.83
Air emissions (pounds)	15.1	18.4	34.4
Waterborne wastes (pounds)	4.01	4.40	6.96
Postconsumer solid waste (cubic ft)	5.02	5.44	6.94

TABLE 2.5

IMPACTS FOR 1,000 GALLONS OF MILK DELIVERED IN REFILLABLE PLASTIC BOTTLE

Impact Category	Gallon	Half Gallon	Half Pint
Energy (mil Btu)	1.56	2.21	4.35
Water (thou gal)	0.44	0.82	2.80
Industrial solid waste (cubic ft)	0.11	0.16	0.32
Air emissions (pounds)	8.81	10.8	15.5
Waterborne wastes (pounds)	3.24	3.48	4.14
Postconsumer solid waste (cubic ft)	2.00	2.65	3.54

The production of nonrefillable plastic bottles is similar to refillable plastic bottles. The system consists of eight processes: HDPE resin manufacturing, bottle manufacturing, bottle packaging, bottle closure, filling and sealing, transportation for milk delivery, transportation of the containers, and the postconsumer solid waste system. In general, HDPE resin manufacturing system has the highest impact in each impact category. Table 2.6 shows that the magnitude of the impacts increases as bottle size decreases due the smaller containers requiring more raw materials per unit volume of milk.

Paperboard milk containers are manufactured from bleached kraft paperboard and coated with low density polyethylene. The cartons are shipped to dairies where they are filled and sealed. The container system consists of eight processes: bleached kraft system, plastic resin system, container formation, filling and sealing, corrugated packaging system, transportation of the paperboard, milk delivery, and postconsumer solid waste. For each size container, the bleached kraft system represents the largest impact in each category. The impacts for the three container sizes are summarized in Table 2.7. As found with other container systems, the paperboard container impacts increase for the smaller containers due to the greater amount of material required to produce the smaller containers.

The last container system examined in this study is the nonrefillable plastic pouch manufactured from low density polyethylene (LDPE). This container consists of the plastic pouch, an outer LDPE bag to package the milk, and an HDPE pitcher to serve the milk. The system consists of seven processes including LDPE resin manufacturing, LDPE film manufacturing, packaging for the pouch system, transportation of the pouch, filling and sealing of the pouch, milk delivery, and postconsumer waste. In general, the LDPE resin system presents the greatest impact, especially for the half gallon and half pint system. The impacts for each pouch are quantified in Table 2.8. In contrast with other container systems, the gallon size container does not have the lowest impact as it consists of two half gallon pouches with an outer bag. Therefore, the half gallon system requires the least amount of raw materials per unit volume of milk and is the most favorable size.

Table 2.9 compares the gallon size container for the five container systems. The table show that the refillable HDPE or nonrefillable LDPE pouch container have the lowest impacts in each category.

TABLE 2.6

IMPACTS FOR 1,000 GALLONS OF MILK DELIVERED IN NONREFILLABLE PLASTIC BOTTLE

Impact Category	Gallon	Half Gallon	Half Pint
Energy (mil Btu)	8.78	10.1	22.7
Water (thou gal)	0.66	0.72	2.12
Industrial solid waste (cubic ft)	0.38	0.44	1.14
Air emissions (pounds)	30.9	34.8	70.2
Waterborne wastes (pounds)	3.13	3.50	7.42
Postconsumer solid waste (cubic ft)	75.4	75.8	79.6

TABLE 2.7

IMPACTS FOR 1,000 GALLONS OF MILK DELIVERED IN NONREFILLABLE PAPERBOARD CONTAINER

Impact Category	Gallon	Half Gallon	Half Pint
Energy (mil Btu)	8.62	9.04	13.8
Water (thou gal)	2.87	3.07	4.51
Industrial solid waste (cubic ft)	0.85	0.90	1.39
Air emissions (pounds)	26.1	26.7	38.2
Waterborne wastes (pounds)	6.24	6.48	9.60
Postconsumer solid waste (cubic ft)	30.4	31.0	35.5

TABLE 2.8

IMPACTS FOR 1,000 GALLONS OF MILK DELIVERED IN NONREFILLABLE PLASTIC POUCH

Impact Category	Gallon	Half Gallon	Half Pint
Energy (mil Btu)	3.59	2.37	4.49
Water (thou gal)	0.32	0.21	0.52
Industrial solid waste (cubic ft)	0.13	0.08	0.18
Air emissions (pounds)	14.0	10.6	15.1
Waterborne wastes (pounds)	1.51	1.19	1.70
Postconsumer solid waste (cubic ft)	1.05	0.58	1.37

TABLE 2.9
COMPARISON OF IMPACTS FOR ONE GALLON MILK CONTAINER SYSTEMS

Impact Category	Ref. Glass	Ref. HDPE	NR HDPE	NR Paper	NR LDPE
Energy (mil Btu)	2.90	1.56	8.78	8.62	3.59
Water (thou gal)	0.64	0.44	0.66	2.87	0.32
Industrial solid waste (cubic ft)	0.60	0.11	0.38	0.85	0.13
Air emissions (pounds)	15.1	8.81	30.9	26.1	14.0
Waterborne wastes (pounds)	4.01	3.24	3.13	6.24	1.51
Postconsumer solid waste (cubic ft)	5.02	2.00	75.4	30.4	1.05

Note: NR = nonrefillable
Ref. = refillable

A few conclusions can be generalized across all milk container systems examined in this report. First, for refillable containers, as trip rate increases, naturally the environmental impacts decrease. However, the rate at which the impacts decrease diminishes with increasing trip rate. Second, the largest containers have the smallest impacts because a larger container requires less packaging per unit volume. The only exception is the gallon nonrefillable plastic pouch as it consists of two half gallon pouches with an outer bag. For refillable containers, the largest containers also have the lowest impact since they have the least surface area requiring washing per unit volume.

This report is one of the most detailed reports found in this literature review. Unlike several other reports examined in this literature review, this report contains a detailed appendix explaining the process flows for manufacturing the various container systems along with documented numbers for this impacts. The drawback to this study is that it is 12 years old and the data are therefore dated. In fact, some of the milk containers described in this report are no longer used. While the report concludes that the refillable HDPE container has the lowest impacts, this container is not in general use in the United States. Thus, of all the containers currently in use (refillable glass, nonrefillable HDPE, and nonrefillable paper), the refillable glass container appears most favorable. However, this report is beneficial to our research in that it describes a framework for considering the impacts due to the manufacturing of packaging materials.

2.4 CONCLUSIONS

Previous studies examining the environmental and energy impacts of packaging materials have differed in the breadth and depth of their scope. Some studies have considered a wide range of packaging materials, but as a result lack details of specific product packaging choices, while other studies have analyzed specific product packaging choices in great detail. A problem with many of these studies is that proprietary databases have been used. Without access to these data, it is not possible to verify the results or recalculate the results for those studies using old and obsolete data. However, these studies help define the cradle-to-grave framework of analysis that we will use for analyzing the impacts of packaging production.

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SECTION 3 SOLID WASTE PRODUCT CHARGE

3.1 INTRODUCTION

Rationale for a solid waste product charge

Every material product enters the solid waste stream at one time or another. Waste management creates a cost to society: the immediate monetary cost of collection and disposal and the cost incurred through the environmental impacts of various disposal measures like landfilling and incineration. Also, depletion of nonrenewable resources constitutes a cost for present and future generations.

This cost is imposed on society at large. It is an external cost, i.e. the agents who carry out the activity which gives rise to the cost do not have to bear it. Economic theory suggests that efficiency in the operation of the economy is improved when external costs are internalized - i.e. when agents who engage in activities imposing costs on others are made to bear all of these costs. The decision framework of private agents like firms and consumers has to be modified in such a way that it reflects the full costs borne by society as a whole. Agents who have to face the true cost of their activities will find it profitable to alter their behavior if the (full social) cost of their activity outweighs the benefits accruing to them. This, in brief, is the rationale for a policy imposing taxes or fees on those agents who cause externalities.

The solid waste externality arises with the production of materials that need to be disposed of eventually. Therefore, every material product should bear a disposal charge according to its contribution to solid waste management costs and environmental costs associated with disposal. A product made from recycled material should bear no charge since it has already been charged during its first lifecycle when it was made from virgin materials.

As materials are modified in subsequent stages of production, additional external solid waste costs may arise. For example, if two materials which by themselves do not cause harmful environmental impacts are blended, and if this compound generates toxics when it is incinerated, then the process of blending adds to external cost.

Design Aspects of a Solid Waste Charge, as discussed by Congress and the EPA

Federal discussion of a solid waste product charge has a long history. It seems that it was first proposed at the 1970 Senate Hearings on the Resource Recovery Act [1]. Later, Senator Hart of Colorado introduced a proposal on the topic [2].

Different aspects of the design of the tax have received attention at different times. Initially, there was significant of interest in the way the tax revenue from a product charge should be distributed - should the funds be earmarked for solid waste management purposes and given to municipalities, or should they be returned to the taxpayers in the form of an income tax break? While this question has important political ramifications, it is an issue entirely separate from the problem of correcting externalities. The following design aspects will be addressed below:

1. Which products should be covered by the program, and what is the rationale for singling out packaging and paper products?
2. What should be the pricing rationale for a product charge?
3. Which product characteristic should the charge be based on?

All of these questions were addressed in the course of the 1970's round of policy discussion.

1. The issue of greatest importance is how many products should be included in the charge program. In theory, all material products should bear a charge, since they all contribute to solid waste management costs. However, levying a tax on every material product was thought to be a suboptimal strategy [3]. Administrative costs were perceived

[1] Statement of Leonard S. Wegman, Leonard S. Wegman Co., Inc., New York. In: U.S. Congress. Senate. Committee on Public Works. Resource Recovery Act of 1969 (pt. 3). Hearings before the Subcommittee on Air and Water Pollution, 91st Cong., 2d sess., on S. 2005, Feb. 20, 23-25, 1970. Washington, U.S. Printing Office, 1970. p.1854-1866. Quoted in: U.S. Environmental Protection Agency, Office of Solid Waste Management Programs. Resource Recovery and Solid Waste Reduction. Fourth Report to Congress, U.S. Government Printing Office, 1970. p.89.

[2] Proposal number: S.1281. Quoted in: Solid Waste Disposal Charge Design Issues. Staff Background Paper No. 9 (June 1978)

[3] U.S. Environmental Protection Agency, Office of Solid Waste Management Programs. Resource Recovery and Solid Waste Reduction. Fourth Report to Congress. U.S. Government Printing Office, 1977, p. 90.

to be rising faster than benefits, as more and more products were included in the charge program [4]. Thus, as formulated in the 1970's policy discussions, the charge was to cover only a certain group of products - all paper products and packaging. There are several reasons for singling out these two categories:

a) Paper was then and still is the single largest item in the waste stream. Non-paper-packaging also constitutes a large proportion of waste generation [5].

b) From the point of view of administrative cost, one is interested in the number of establishments from which the tax would have to be collected. A charge on paper and packaging alone would involve relatively few establishments [6].

c) Packaging is extremely shortlived. Nondurable and extremely shortlived items aggravate the waste crisis in that they yield services only for a very short time, yet cause waste management costs similar to those for durables. For nondurables and packaging, the amount of services yielded per unit of waste management cost is very low.

Another reason for focussing on non-durables is the uncertainty about future waste management cost. Durables will need to be disposed of at a date in the future for which the correct solid-waste management costs are difficult to assess. (This touches on the issue of marginal versus average cost pricing. See below.) For this reason, deposits or bounties were considered a policy instrument better suited to deal with the solid waste externality of durables [7].

d) The product charge will constitute a larger proportion of the price, the cheaper the product is on which it is levied. Consumers react to percentage increases in

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[4] ibid., p. 90 and p. 93. Opinions on the magnitude of the administrative cost of a product charge scheme apparently have differed widely. See f.ex. ibid., p. 93 f., where a study for the EPA is quoted which found that "collection costs for IRS (or other product charge collection authorities) under the plan would be moderate - certainly not in excess of 1 percent of gross yield and probably in the vicinity of 0.5 percent." [For a more pessimistic opinion, see "Benefits and Costs of a National Solid Waste Disposal Charge". Staff Background Paper No. 12, March 1979, p.11 f.]

[5] Solid Waste Disposal Charge Design Issues. Staff Background Paper No. 9 (June 1978), p.49.

[6] ibid., p. 48 f.

[7] ibid., p. 50.

price: A 30 cent product charge on the HDPE content of a milk container will create a larger response by consumers than the same 30 cent charge on the packaging of a camera (assuming that the product charge will be passed on entirely to the consumer). Since packaging waste from nondurable, low-priced consumer products dominates the retail packaging waste (food products, toiletry and household goods), a charge on packaging promises a noticeable consumer response.

2. Another aspect that has to be clarified is the pricing rationale for such a charge. Exactly what externalities are to be addressed, and how can an accurate measure be found for them? As mentioned above, one can identify three types of externalities which arise with the disposal of materials: a) Waste management costs, i.e. the costs of collection and disposal, b) Costs of environmental impacts of disposal, and c) Depletion of non-renewable raw materials. The last point seems to have caused some concern[8]; however, it was not to be addressed by the policy instrument of a product charge because it was felt that it would be too difficult to find a political consensus on how to value these non-renewable resources [9]. The second point is more closely related to municipal solid waste management practices. It was conceded that "environmental damage costs would be a valid basis for a charge"[10], but at the time there were no estimates for costs of this kind. Thus, the externality that was to be addressed by the product charge was the cost of municipal solid waste management [11].

The Senate proposal based its tax rate suggestion on the average cost of collection and disposal prevailing at the time. This measure was obtained from a sample of various cities' waste management expenditures. However, since many cities disposed of their waste in an uncontrolled manner, causing unacceptable pollution, it was felt that the existing waste management practices should not be a basis for assessing the costs of waste management. The product charge should allow for the realization of the intention to improve disposal practices [12].

Also, to ensure optimal resource allocation, the charge should reflect the marginal rather than the average cost of waste management. It is the cost of an additional unit of

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[8] Rationale for National Solid Waste Disposal Charge Legislation. Staff Background Paper No. 8 (June 1978), p. 39 and p.43.

[9] ibid., p.45.

[10] Staff Background Paper No. 9, p. 53.

[11] Staff Background Paper No. 8, p. 38.

[12] Staff Background Paper No. 9, p.52.

waste which agents who create waste should be made to take into account. However, marginal cost is very difficult to determine, especially in the long run.

3. Related to the question of which externalities should be addressed is the question of which product characteristic the charge should be based on: weight or volume? Weight might be easier to measure, but for many products, collection and disposal costs are more closely related to volume. An alternative tax which was supposed to take into account volume rather than weight was suggested: a unit tax on rigid packaging (i.e. a tax on containers) [13].

3.2 SELECTIVE REVIEW OF STUDIES ON THE SOLID WASTE PRODUCT CHARGE

In this section, a few studies on the solid waste product charge are presented. Steve Buchanan offers some reflections on the merits of a product charge program, attempting to assess the relative magnitude of its benefits versus its costs while allowing for errors in the specification of the charge rate. Spillers and Havlicek estimated the impacts on waste generation of a charge on the packaging of selected food products, employing a model with very restrictive assumptions. The two comprehensive and very detailed studies which have been carried out for the EPA in 1974 (Bingham et al.) and 1980 (Miedema et al.) are discussed in a separate point.

Evaluating the Efficiency of the Solid Waste Charge, S. Buchanan

In this paper, Buchanan undertakes two tasks: First he determines the lower boundary for the net benefit from an "ideal" product charge, i.e. a product charge which accurately reflects the marginal cost of collecting and disposing of a unit of waste. Then he examines the sensitivity of this lower bound estimate to errors in specifying the charge level. Throughout the paper, the author abstracts from the benefits of improvements in environmental quality.

The net welfare change ($\blacktriangle W$) of a product charge with an exemption for recycled materials is the sum of the following elements: on the benefit side, there are tax revenue (R), and savings from the reduction in the waste disposals of, due to reduction in waste generation (WR) and due to increased recycling (SR). (One unit of output q produced and purchased translates into one unit of waste.) On the cost side, there are the losses in consumers' and producers' surplus (CS, PS) and the administrative costs of collecting and distributing the tax revenue (f(R)). Part of the losses in the consumers' and producers'

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[13] ibid., p. 55 f. See also D.W. Conn: Waste Reduction. Issues and Policies. In: Resources Policy, Vol 3, No. 1, March 1977, p.13.

surplus is redistributed as tax revenue. The part of the loss in surplus which is not redistributed is the deadweight loss (DL). Thus,

$$\begin{aligned} \Delta W &= R + WR + SR - (CS + PS) - f(R) \\ &= WR + SR - DL - f(R) \end{aligned}$$

For estimating a lower bound of potential net benefits, a worst case scenario is assumed: manufacturers of the retail product do not substitute secondary for virgin material and pass on the entire amount of the product charge to the consumer. There will be no cost savings due to recycling (SR=0). However, it is also assumed that the administrative cost of handling the tax revenue is zero (f(R)=0). (This is why the author speaks of "potential" net benefits.) Thus, the lower boundary for the potential net benefit will be the avoided waste management cost due to a reduction in waste generation minus the deadweight loss:

$$\Delta W_{pot} = WR - DL$$

Recall that the tax rate t is assumed to reflect accurately the marginal cost of collection and disposal. The savings from waste reduction therefore equal the tax rate multiplied by the reduction in quantity purchased ($WR=t^*\Delta q$). Approximating the demand curve for product q by a linear function over the relevant range, the deadweight loss is given by $(t^*\Delta q)/2$. Thus, the lower boundary for potential net benefit is:

$$\begin{aligned} \Delta W_{pot} &= WR - DL \\ &= t^*\Delta q - (t^*\Delta q)/2 \\ &= (t^*\Delta q)/2 \end{aligned}$$

This expression shows that the potential net benefit increases directly with t, the charge rate. This result does not come as a surprise: The larger the externality (which is accurately accounted for by t), the greater the benefit from internalizing it. Also, the potential net benefits rise with the price elasticity of demand for the retail products: a greater price elasticity of demand implies a greater reduction in quantity purchased (Δq). The greater the reduction in quantity purchased, the greater the reduction in waste generation.

However, the assumption that a correct charge rate t can be found is not very realistic. The author derives a function for the net benefit depending on the percentage deviation of t from the true marginal waste management cost, c (t = ac). Assuming a constant elasticity of demand, ϵ, the potential net benefits are

$$\begin{aligned} \Delta W(a) &= [(acq\epsilon)/2p]^*(2c - ac) \\ &= (2c^2a - a^2c^2)^*(q\epsilon/2p) \end{aligned}$$

This function is an inverted parabola with its vertex at a=1 (implying t=c, the correct charge level). For $\epsilon = 1$, i.e. unitary elasticity of demand, the function will become zero at a=2 and a=0 and negative for a > 2 and a < 0. (The latter is not applicable to the case of a positive product charge: any positive t implies that a > 0 .) This suggests that an error of 100% (a=2) will leave welfare unchanged (net benefits are zero), and a greater error will decrease welfare (net benefits are negative). The parabolic shape suggests also that net potential benefits are not very sensitive to small errors. F. ex., an error of 25% (a=75% or 125%) still would bring about 93.8% of the maximum potential net benefits, and an error of 50% would still yield 75% of the maximum potential net benefits. If the price elasticity of demand were greater than 1, and < 100% would imply zero net benefits. In general, the greater ϵ, the more sensitive the potential net benefits are to an error in the tax rate specification t.

It is inevitable that errors will be made in specifying the charge rate. How likely is it then that these errors are so large that potential net benefits would become zero or even negative (i.e. would turn into costs)? There are many sources of error in estimating the marginal solid waste management costs. The most important are regional differences in collection and disposal costs, the existence of local user fees, and the heterogeneity of the waste stream (different materials cause different costs).

The author demonstrates the dispersion of collection costs with a sample of 117 cities. In this sample, average collection costs vary from $10 to $70. It can be shown that levying a uniform tax of $25 would reduce the potential net benefits to 65% of what they could maximally be - and that only if average costs behave similarly to marginal cost (since it is marginal rather than average social cost which the charge level should be equated to). Assigning an error margin of 33% to the third error source (the differences in collection and disposal cost of the different materials that constitute the waste stream) and a positive estimate for the administrative cost, Buchanan finds that the solid waste product charge is a "rather marginal policy alterative" because misspecification of the charge level is so likely to occur. It thus seems that the errors in specifying a correct tax rate, t, can greatly reduce the benefits from a product charge.

However, in view of the fact that distortions already exist (solid waste management being financed out of property tax, for example), this conclusion has to be modified. It is likely that the status quo implies a lower level of welfare than the one that would prevail in the absence of all policies (p. 62). The introduction of a solid waste charge would then bring about a larger improvement in welfare than suggested above. Also, remember that the maximum potential net benefits increase with c, the marginal social cost to be internalized. The abstraction from environmental quality leads to an understatement of the marginal cost of waste management. That means that it is less likely that the tax rate t would greatly overstate the social marginal cost c. For a price elasticity of ≤ 1, such an overstatement of c is the only scenario which could bring about negative net benefits, i.e. which would reduce welfare rather than increasing it. In other words: if the potential benefits are not fully realized because of a misspecification of the tax rate, then it is far more likely that the error consists of specifying a tax rate which is too small, rather than too large.

Also, remember that the welfare change was more sensitive to error in specifying t, the larger the elasticity of demand for the product contributing to the waste stream. If one could eliminate some sources of error for products with a high price elasticity of demand, much would be gained. One could specify the cost share of the goods with a relatively high elasticity of demand (for example beverages) somewhat more precisely and thus reduce error in an area where net benefits are especially sensitive to error (p.61).

It is thus crucial to find a good engineering-based estimate of the marginal cost of collection and disposal.

A Simulation Analysis of the Effects of a Product Tax on the Packaging Waste Output of Selected Food Products, P. Spillers and J. Havlicek, Jr.

Spillers and Havlicek estimated the incidence of a product charge on the packaging of 44 selected food products and its effect on waste generation. Experimentally, they determined the material composition of the packaging of each product, i.e. they found input-output coefficients for n food products and m packaging materials, reflecting the generation of material waste j by product i. The tax for each product was calculated as the sum of the collection and disposal unit costs of the j materials, weighted by the coefficients which describe the contribution of each product to the generation of a specific material waste. Assuming that the waste coefficients and the values for the solid waste management costs were correct, this tax level would indeed reflect the contribution of each product to the waste management cost (assuming that the marginal cost of waste management is constant). The incidence of this unit tax was estimated for two scenarios: a uniform waste management cost of $20 for one ton of waste of a given material and a uniform cost of $50 for the same. The price elasticity of supply for each product was assumed to be 1 [14]. Elasticities of demand for the food products seem to have been taken from another study. The authors find that the $20 tax induces a waste reduction of 0.38%, and the $50 tax a waste reduction of 1.0% .

The Spillers and Havlicek model has the severe limitation that it does not allow for factor substitutions in the manufacture of the retail product and its packaging. The waste reduction can come about only by reduced consumption of the taxed products, not by a

[14] In the larger study which underlies this article, the assumption that all materials have the same collection and disposal cost was relaxed. Also, the price elasticities of supply of the different products were varied. Cf. p. 188.

shift to a different packaging material, and not by a reduction in packaging per unit of product. Thus, the waste reduction estimate is conservative. The authors justify this simplification with the observation that "the per unit tax rates and accompanying changes due to the tax ... [are] rather 'small'. Therefore, it is believed that the producer will not alter his fixed packaging technology in response to such small price perturbation."[15]. However, depending on where the tax is levied, producer response might contribute more to waste reduction than consumer response.

The study yields an interesting result: the product charge (which was believed to reflect the average solid waste management costs of a product) constitutes only a small proportion of product price. For example, the highest excise tax rate for the $50 waste management cost scenario is the tax on salad dressing, barely 3 cents per pound of the final product. The taxes on other products generally amount to less than one cent per pound. Thus, while consumers reduce purchases only slightly, and waste reduction therefore is small, the tax does yield a revenue to finance collection and disposal of packaging waste from the selected food products. Of course, this result rests crucially on the correct assessment of waste management costs.

3.3 THE BINGHAM AND MIEDEMA STUDIES

The discussion of these two studies is done in five parts:

Summary
Scope and assumptions of the models
The models described in detail
Estimation results
Critique of the studies.

Tables mentioned in this section will be found in section 3.4 Tables.

Summary

The **Bingham** study examines the effects of 4 policies concerning packaging of consumer products. One is a regulation requiring the use of recycled materials in packaging. The others are taxes on the weight of packaging, on the weight of virgin material in packaging, and on rigid containers.

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[15] p. 188

The approach of this study is to construct an economic model, apply each policy, and assess its impact. The model is a comparative statics model; it compares equilibrium levels of production and consumption in the life of a package with and without the policy in place, but it does not track the economic adjustments in between. In this model there are three stages in the life of a package: 1) production of bulk materials. Both virgin and recycled inputs to packaging are collected and processed in this step. 2) production of the final product. This step also includes production of the package; inputs to each of the 30 final products in the model include 9 packaging materials and one aggregate input "all other". The packaging materials are flexible paper-and-paper closures, flexible plastics-and-plastic closures, metal closures, flexible aluminum, and rigid paper, glass, steel, plastics and aluminum. 3) consumption of the final product. Key assumptions of the model include that virgin and recycled bulk materials are perfect substitutes up to some technical limit. It assumes an upward sloping supply curve for recycled steel and recycled paper and a horizontal supply curve (perfectly elastic) for virgin materials and all other recycled materials. A linear homogeneous production function with constant returns to scale is used to model manufacture of the packaged final product.

Results of the study allow a comparison of the effects of the four policies. In summary, the tax on containers causes the largest reduction in production and consumption of packaging. The tax on the weight of packaging is the least effective policy, both for recycling and for reducing packaging consumption. (The tax per container is equivalent to the tax per ton on all rigid virgin packaging materials, but at a rate higher than any of the tax rates included in this study). Both regulation and the tax on virgin materials produce roughly comparable changes in recycling and in consumption of packaging; overall, however, the tax on virgin materials is most effective relative to cost. Effectiveness is measured as changes in consumption in physical units and cost is measured as losses in consumer surplus. (Consumer surplus is the difference between what a consumer is willing to pay for a good and what the consumer actually pays).

The **Miedema** study considers the effects of a tax levied on the weight of virgin material in packaging. The approach of this study is to design an economic model of the production and consumption of packaging and then to analyze the operation of this model in two ways. First is to systematically vary the tax, and then analyze the benefits and costs of the tax policy. Second is a sensitivity analysis in which a tax is held fixed (at $15 per ton), while raising and lowering the parameters of the model to analyze resulting changes in the amount of solid waste generated from packaging. The purpose of such sensitivity analysis is to provide better general understanding of the behavior of the model.

The model considers five stages in the life of a package: 1) production of bulk material from virgin and recycled inputs; 2) production of 2 general packaging types, rigid and flexible packaging made from various materials, (specifically, rigid paper, plastic, glass, steel and aluminum and flexible paper, plastic and aluminum); 3) combining rigid and flexible packaging into a finished package; 4) production of the consumer good which is made up of the package and other inputs; and 5) consumption of these goods. The goods include baked goods, dairy products, meat-and-seafood, produce, canned foods, other foods, prepared beverages, beer, wine, distilled spirits, soft drinks, household supplies, health-and-beauty aids, pet foods, and an "all other" category.

The model is a comparative statics model; it shows equilibrium levels of production and consumption for each of the five stages of the package before and after the tax. Some of the key assumptions made in this model include assuming values for the elasticity of substitution between secondary and virgin inputs to packaging. (All plastic and flexible aluminum are not recycled; elasticity equals 0). The elasticity of supply of secondary materials and of virgin woodpulp is assumed to be upward sloping. Other bulk materials are assumed to have infinite elasticity of supply. The production function for packaged final products is assumed to be CES, constant elasticity of substitution.

Very briefly, results of the model show that the tax (at $10 and $40 per ton) causes relatively small reductions (1% and 3%) in the production and consumption of bulk materials. It causes somewhat larger increases (9% and 32%) in recycling. Of all the packaging materials, the largest effects are that less glass is produced and consumed, and more paper is recycled. The net social benefits of the $10 and $40 per ton taxes were found to be $4 and $58 million (in 1974 dollars) respectively. The sensitivity analysis reveals that recycling exerts the most influence on model results. Recycling is most sensitive to the elasticity of supply of secondary materials. Social benefits also appear sensitive to parameter changes; this is because of a close relationship between benefits (reduced waste management expenditure) and recycling and because disposal costs are only incurred in the less sensitive stages of the model.

Scope and assumptions of the models

The scope of the Bingham and Miedema studies is very large. Contrary to Havlicek/Spillers, Bingham and Miedema model the production process in a way that takes into account the factor substitution in the manufacture of the retail product and its packaging. Both studies analyze the effects of a solid waste product charge in various designs. In particular,

Bingham et al. look at

a) a tax on all packaging per ton of packaging

b) a tax per ton of virgin content of packaging, i.e. a tax as in a), but with an exemption for recycled material content,

c) a tax on rigid containers,

d) a regulation specifying a minimum percentage of recycled material that a packaging material is to contain.

Miedema et al. look at

> a tax per ton of virgin material content of bulk paper and packaging bulk material (the model also accepts as an input a subsidy for the use of secondary material in bulk paper and packaging bulk materials production).

Bingham et al. collected data on the packaging structure of the economy, i.e. on the packaging consumption of various industries producing nondurable consumer goods. They concentrated on 30 product categories: 29 of the consumer products which utilized relatively more packaging (in weight) and a residual category capturing all other products (see Table 3.1). The nine packaging materials between which they differentiate are given in Table 3.2. Employing a model of production of packaging materials and consumer nondurables, they predicted the adjustments that the aforementioned policies would induce in production and consumption.

The study by Miedema et al. is based to a large extent on the Bingham data. However, it aggregates the product categories to 15 (see Table 3.3) and dispenses with one packaging material ("crowns and closures"). Its model of production and consumption is somewhat more detailed than Bingham's.

The studies have a series of key features **in common**:

a) Both employ comparative statics models, i.e. models which abstract from time. The insight that these models provide is a comparison between equilibrium situations. Since time does not enter the analysis, not much can be said about the chronological sequence of adjustments by which the economy (or a market or whatever is the object of the model) moves from one equilibrium situation to the other. The introduction of time as a variable on which other variables depend would greatly increase the mathematical complexity of the models without providing much more useful insight.

b) Both models can accommodate different tax rates for individual materials, but the authors chose to consider uniform tax rates only. This is in line with their assumption that all materials have the same solid waste management cost, or rather: with their abstraction from the composition of the wastestream. They assume that a ton of waste has a certain collection and disposal cost. In reality, however, weight units of different materials have different collection and disposal costs because of their varying weight to volume ratios. And obviously, materials will differ greatly in the environmental impacts which their disposal gives rise to (see c)).

c) Neither of the studies consider environmental impacts of landfilling or incinerating packaging waste.

d) While the specification of the production function for packaging materials is different for the two models, both models employ the same specification for all packaging materials. The production functions are allowed to differ only in their parameters. (Qualification: Bingham allows for different technological limits in the substitution of secondary versus virgin material.)

This assumption is somewhat restrictive. In reality, production processes for individual material differ in many respects. One of these is how many production steps are carried out in one and the same plant. Glass containers, for example, are produced from cullet and the virgin materials in one single process. In plastics production, there are many stages of processing and refining during which the materials pass from one firm to another.

Another difference between the production processes of individual materials is the processing and upgrading requirements for the secondary material. Secondary paper needs to be de-inked while glass cullet just needs to be melted down.

e) Both studies assume that production occurs under constant returns to scale, and both abstract from market forms in that they assume perfectly competitive conditions to prevail. The former assumption ensures that average cost is constant. The latter ensures that price equals average cost. This implies that packages and retail products are supplied at a constant price. Thus, an increase in production cost (more precisely, in average cost of production) which comes about by the tax on packaging material will translate directly into a price increase.

In reality, average costs are not constant in many industries, which implies that the respective supply curves are upward sloping, i.e. responsive to price - even under perfectly competitive conditions. In consequence, part of the tax on packaging is absorbed by the suppliers, and the decrease in sales of the retail product would not depend any more on price elasticity of demand alone, but also on price elasticity of supply. Also, markets are not perfectly competitive. Apart from horizontal concentration, some firms are also vertically integrated, as, for example, in aluminum production.

f) Neither model differentiates between products with respect to packaging requirements. All packaging users (the retail manufacturers who apply the package to the product) in all 15 product categories make a choice over all the materials. If different packaging requirements show up at all, they do so only in the different parameter estimates for the production function.

The studies **differ** in the way they model the production of packaging materials and final products. In addition, they employ different techniques for estimating the production functions and consumers' response to price increases. These points are discussed in some detail below.

The models described in detail

A model for the production and consumption of packaging needs to look at the essential stages in the "life-cycle" of a package:

mining of the virgin material
processing of the virgin material
reclamation of the secondary material from the waste stream
processing of the secondary material
production of the packaging bulk material
production of the package
production of the final product, i.e. combination of package and content
sale of the final product
Reuse, recycling or disposal of the package

Both Bingham's and Miedema's models abstract from some steps and assume fewer stages of production than described above. Below, there is a description of how each model deals with stages of the life-cycle of packaging.

Bingham et al.

Bingham et al. assume that virgin and secondary are mined resp. reclaimed and processed in one step. Similarly, production of the package and production of the final retail product are modeled as part of the same decision, taken by the retail product manufacturer. In particular, the production process is modeled as follows:

1. Production of the Bulk Material

The production of the packaging bulk material occurs under perfect competition. The inputs into producing bulk material are virgin material (see 1a. below) and secondary material, i.e. material reclaimed from the wastestream (see 1b. below). Virgin and secondary material are perfect substitutes up to a technical limit. This limit was set for each material after discussion with industry representatives. For glass, for example, an upper limit of 50% was specified, i.e. it was assumed that the secondary content of glass could be at most 50%. For steel, 45% was assumed, for plastics 80%, for paper 30% and for aluminum 45% (this latter does not constitute a technical limit on secondary content. Rather, it is supposed to reflect the fact that a lot of aluminum scrap which is the by-product of aluminum production is recycled already, so there would be no more room for utilizing out-of-plant aluminum scrap).

The amount of recycling depends on the relative prices of virgin and secondary material, on the supply elasticity of secondary materials, and on the demand for secondary materials. Since virgin material and secondary material are perfect substitutes, the unit cost of virgin material constitutes the demand curve for secondary. Whenever virgin is more expensive than secondary, the producers switch to the latter. (For a comment on the specification of the demand for secondary materials, see below: Critique of the Bingham and Miedema studies.)

1.a. Virgin material supply

The supply of virgin material is perfectly elastic, i.e. the supply curve is horizontal, because there is perfect competition in the extracting industries and/or the virgin materials are abundant (as in the case of glass production: for all practical purposes, there is a naturally "unlimited" supply of sand and soda ash).

1.b. Secondary material supply

For glass, plastics and aluminum, supply functions with infinite price elasticities were assumed ad hoc. The supply of secondary steel and paper is increasing with price. The authors constructed these two supply functions as follows:

The absolute amount of steel and paper waste generated by each city in the U.S. was calculated from the per capita waste generation, the proportion of steel and paper in the solid waste stream, and the population size of the city. Then a waste processing plant size was assigned to each city (a plant of the size to process the total amount of waste generated by the city). The authors had some estimates for the cost of reclaiming different waste materials for different plant sizes. (These were obtained from a waste processing plant in Ohio.) The cost estimates for reclaiming paper and steel were employed for the hypothetical plants. The result of this exercise are "data"points embodying two values: amounts of secondary steel or paper and average cost of reclaiming these amounts from the waste stream. Then the amounts of steel and paper waste that could be recovered in different cities were accumulated to the national total, starting with the amounts recoverable at lowest average recovery cost up to the amounts that are most costly to recover from the waste stream.

2. Production of the final retail product

The production of the final retail product is also assumed to be perfectly competitive and to occur under constant returns to scale. The production function for each retail product has ten inputs: the nine packaging materials and one aggregated input ("all other") which comprises the product proper, labor, etc. .

3. Consumer response

Consumer response to price increases is given by the price elasticity of demand. Bingham et al. took price elasticities for some products from external sources and estimated the elasticities for the remainder of the products anew. These estimations were done separately for each product, i.e. the interdependence of demands was not taken into account (in other words: cross price elasticities of demand were assumed to be zero).

4. Substitutions which the model allows for

Bingham's model incorporates the following possibilities of substitution:
a. substitution between virgin and secondary material (material reclaimed from the waste stream and to be recycled) in the production of an individual packaging bulk material
b. substitution between different packaging materials (paper, glass, steel, plastics, etc.) in the production of the package
c. "Reduction in the utilization of packaging" (more content per weight unit of packaging). In the context of the model, this substitution can occur via a reduction in the consumption of individual materials for $100 consumer expenditure on a retail product.
d. the change in purchases of the final consumer goods.

Adjustment in all of these four areas will occur in response to the above policies. All of these adjustments affect the solid waste stream:

- Substitution of secondary for virgin materials (a) might not reduce the quantity of waste generated, but will reduce the quantity of waste to be disposed of by the amount of the materials that are recycled, or, if recycling has already taken place before the implementation of the policy, by the amount of the increase in recycling.
- Shifts in the utilization of various packaging materials (b) will change the composition and the size (weight) of the waste stream - the latter occurring when lighter materials are substituted for heavier materials, i.e. plastics for glass (whether this is a desirable effect or not, is a different question. Differential taxes would address this issue.)
- Decrease in overall utilization of packaging (c) will decrease the amount of waste generated per se (true source reduction).
- Decrease in final commodity purchases (d) will also decrease waste generation.

Miedema et al.

Miedema et al. assume that processing of the secondary material and production of the bulk material are done in one step. Also, they insert a step between the production of bulk material and the production of packaging, namely the production of a composite "rigid" versus "flexible" packaging material. The packaging consumers (i.e. final product manufacturers) are only interested in the "rigid/flexible" feature of the package when packaging their products.

In particular, the production stages are modeled as follows:

1. Production of the bulk material

This production occurs under perfect competition. Inputs into this production process are processed virgin material and raw secondary material. For materials for which recycling is assumed to be feasible, the substitution between secondary and virgin is smooth, i.e. the percentage of secondary which a product can contain is a continuous interval. Flexible and rigid plastics and flexible aluminum are assumed not to be recyclable, i.e. the elasticities of substitution between virgin and secondary are assumed to be zero for these materials. For glass and rigid aluminum, the elasticities of substitution are assumed to be infinite (implying that secondary and virgin are perfect substitutes); and positive constants for rigid and flexible paper and steel.

1.a. Virgin material supply

For the packaging material manufacturer, the processed virgin is supplied at a constant price, i.e. with infinite supply elasticity (packaging manufacture constitutes a small part of overall demand for the virgin material).

1.b. Secondary material supply

Supply of raw secondary is upward sloping. The authors took price elasticities of supply from an external source[16]. The processing of the secondary material occurs during the production of the bulk material. The demand for secondary results from the specification of the production function. The proportion of recycled content depends on the elasticity of substitution of secondary for virgin which is assumed to be constant.

2. Production of the final retail product

The retail product manufacture occurs under perfect competition, in two steps. First, a package is produced from two compounds, "flexible" and "rigid" materials. (These compounds, in turn, are produced from the three flexible resp. five rigid packaging bulk materials.) Then, the package is combined with all other inputs.

3. Consumer response

Miedema et al. estimated price elasticities of demand jointly, i.e. estimated a matrix of own and cross price elasticities. This joint estimation is based on a system of simultaneous interdependent equations which is a more accurate representation of reality than the model implied by the Bingham estimations.

4. Substitutions which the model allows for

Miedema's model allows for the same substitutions as Bingham's. The main differences between the two models is the way the supply and demand for secondary materials are specified and the "derived demand module" which Miedema employs for estimating the adjustments that occur in the last three stages of production and consumption.

In the derived demand module, three decisions are determined simultaneously: 1. The packaging manufacturers' decision to combine flexible and rigid materials, 2. the final product manufacturers' decision to combine the packaging with all other inputs - content,

[16] ICF, Inc. . Not further specified.

i.e. the product proper, labor, and capital - and 3. the consumers' decision about how much to buy of the retail product.

Results of the Estimations

How the results are compared

The authors apply different measures to evaluate the various policies. Bingham et al., for example, use three "effectiveness measures": reduction in solid waste generation, increases in consumption of secondary materials, and reduction in raw materials and energy consumption. To arrive at cost-effectiveness measures which allow to rank the policies according to different criteria, the cost (in loss of consumers' surplus) is divided by the three effectiveness measures for each policy.

Miedema et al. employ a net benefit measure: gross benefits minus costs of the policy. Gross benefits are the reduction in solid waste management expenditure, i.e. the avoided disposal cost. Costs of the policy are the loss in consumers' surplus, net of the tax revenue, plus the loss in producers' surplus, net of the rent gains accruing to secondary suppliers and processors. Miedema et al. assume that the tax is equal to the true social marginal cost of collection and disposal. Thus, the avoided disposal costs equal the reduction in the quantity of waste to be disposed of[17] multiplied by the tax rate. (Miedema et al. emphasize that their study was to provide only a model which allows the simulation of the effects of different tax rates.)

These different assumptions and evaluation measures make it difficult to compare the evaluations of the two studies directly. Therefore, we will only look at the effects which different policies have on waste reduction and secondary materials consumption. However, even these effects are not compared easily. The authors did not analyze the same tax rates.

Individual results

The **Bingham study** looked at several tax rates on the weight of packaging, with and without an exemption for recycled materials, and at a regulation requiring that packaging be made from certain percentages of secondary. The base year for which these policies are analyzed is 1970. Without comparing the individual specifications for each policy instrument, one can make a few general observations:

[17] A reduction in the amount of waste to be disposed of can come about by a reduction in the waste generated and by an increase in the utilization of secondary materials. See footnote 18.

The tax on the weight of packaging with an exemption for secondary content brings about the largest reduction in disposal[18]. The highest tax rate with exemption ($100 per ton) induces a reduction of 10.18% in disposal - the sum of a 1.29% reduction in waste generation and an 8.89% increase in the consumption of secondary (table 3.4: rows 1.c. and 2.c.). The strictest regulation, requiring a secondary content of 30%, reduces disposal by only 7.82% (table 3.5: rows 1.c. and 2.c.). The other policy instruments, taxes without an exemption, are far less effective: they do not induce any recycling (tables 3.6. and 3.7.) -- and it is recycling more than source reduction which brings about the reduction in disposal.

A tax on containers brings about the largest decrease in packaging utilization, i.e. the largest reduction in waste generation (2.9% reduction in waste generation for a 2-cent per container tax, 2.1% for a 1-cent per container tax) but no increase in the amount of secondary consumption (see table 3.7.). The tax on containers compares most closely to the tax-on-weight-without-exemption policy. However, the rates assumed for the tax on containers are much "higher" than the rates assumed for the latter. The tax rates on weight which would be equivalent to a 1-cent per container tax rate are:

Rigid paper	$ 398.12 per ton
Rigid plastics	$ 206.51
Glass	$ 36.73
Steel	$ 90.17
Rigid aluminum	$ 133.45

If taxes on weight (the highest rate looked at is $100) were increased, they would induce a greater waste reduction and probably come close to the effects of the container tax rates examined by Bingham.

It is also interesting to see that for the tax with an exemption for secondary materials, the two highest tax rates ($50 and $100 per ton) bring about a greater utilization of secondary material than the strictest regulation requiring a minimum content of secondary, 30% : A $50 tax induces recycling of 8.01% of the waste stream, a $100 tax induces recycling of 8.89% (table 3.4: row 2.c.), whereas the regulation causes only 7.58% of the wastestream to be recycled (table 3.5: row 2.c.). The $50 per ton tax (or 2.5 cent

[18] "Disposal" stands for "waste to be disposed of". In this context, it is useful to make explicit the materials flows underlying the models: Waste is either recycled or disposed of. Thus,

Waste generation = Waste recycling + Waste disposal.

Waste recycling is also referred to as "consumption of secondary material". See footnote 17.

per pound tax) seems easy to implement compared to an across the board regulation requiring each material manufacturer to employ at least 30% secondary material. The tax leaves room for different adjustments which individual material manufacturers find optimal to make. The tax-cum-exemption achieves reduction in waste disposal at a lower cost than the regulation.

The **Miedema study** undertook two sets of simulations for the baseline year of 1985:

First, for a given set of parameters, the tax rates were increased from $10 to $40 (in 1978 dollars), in increments of $5. The parameters for which values were assumed are: the elasticity of substitution between secondary and virgin in the production of each bulk material and the elasticities of supply for secondary and virgin materials. (Some important assumptions about parameters are: Rigid and flexible plastic and flexible aluminum cannot be recycled (the elasticity of substitution between virgin and secondary materials is zero), whereas secondary aluminum and glass are perfect substitutes for their virgin counterparts.)

Second, the tax rate was held constant at $15, and a sensitivity analysis for different parameter values was carried out.

The results that are of interest in comparison with the Bingham study are the results for some tax rates, say $20 and $40, and some "parameter points" (i.e. combinations of different parameter values) that are similar to the assumptions embodied in the Bingham model - if there are any. Recall that Bingham et al. assumed an infinite elasticity of substitution between virgin and secondary material, up to a technical limit. Also, they assumed virgin material supply to be infinitely elastic. For the secondary material supply, they assumed ad hoc prices (infinite elasticity) for plastic, glass, and aluminum, and constructed upward sloping supply curves for steel and paper. There is really no parameter point in the Miedema model that reflects exactly these assumptions. The point that comes closest to the Bingham assumptions is parameter point 4 (see table 3.8).

The simulation result for a tax rate of $20 per ton of packaging bulk material and paper is that the quantity of disposal[19] reduces by 3.5%, of which 1.5% is due to a reduction in waste generation and 2% to increased recycling. The $40 tax reduces the quantity of waste disposal by 7%, of which 3% is due to a reduction in waste generation and 4% to increased recycling (see figure 3.1).

As to individual materials, glass and rigid paper lead the reduction in waste disposal. For glass, there is a larger reduction in waste generation, for rigid paper, a greater increase in recycling (see figures 3.2 and 3.3). At first sight, this result is surprising. Secondary glass is a perfect substitute for virgin materials, so why does recycling not increase greatly

[19] See footnotes 17 and 18.

in the glass industry? The answer lies in the assumptions about the supply elasticities for the materials used in glass manufacture. A rather low supply elasticity for cullet is assumed (see table 3.8 con't), whereas the supply elasticity for the virgin inputs into glass manufacture is assumed to be infinite. Thus, although there is no technical limit to 100% recycling, the market conditions pose a considerable obstacle for even a small extent of recycling to occur. For rigid paper, on the contrary, the elasticity of substitution between virgin and secondary is only finite. The supply elasticity for secondary, however, is 1.1, whereas the supply elasticity for virgin woodpulp is 0.5 . Thus, recycling increases greatly for rigid paper. This indicates that the model outcomes are rather sensitive to supply elasticities of materials.

The sensitivity analysis for different parameter values demonstrates the same phenomenon: the effect of high supply elasticities for secondary "outweighs" the effect of high substitution elasticities in bulk material manufacture. Recycling increases fast when moving from parameter point 2 (high substitution elasticity, low secondary supply elasticity) to point 3 (low substitution elasticity, high secondary supply elasticity).

Parameter point 4 brings about a 3.4% reduction in total waste to be disposed of, of which 1.15% is a reduction in waste generation.

Critique of the Bingham and Miedema studies

Three aspects will be highlighted below:

1. The specification of the production functions,
2. the specification of the supply of secondary materials, and
3. the specification of demand for secondary materials.

<u>1. The specification of the production functions</u>

The most critical feature of the studies is the modeling of the production functions: The product categories are aggregated in a way that does not reflect packaging requirements of the final retail product. While there is a need for aggregation of product categories, this aggregation should not distort the modeling of the decision which the packaging user faces. In some of the retail product categories which Bingham and Miedema form, the manufacturers face similar decisions (for example soft drink industry, beer industry). In other categories, the retail product manufacturers face very different decisions because the products within a single category have such different physical characteristics (for example household cleaning agents: there are powders, pastes, liquids, sprays etc).

The parameters of the production functions for the different product categories were estimated from time series with 13 annual observations. The product categories are very broad in some cases - thus, it is likely that the packaging trends for products in one and the same category have followed very different patterns. Therefore, the estimates for the use of different packaging materials have to be interpreted cautiously.

2. The specification of the supply of secondary materials

There are several problems with the way in which **Bingham et al.** modeled the supply of secondary paper and steel.

First, the assumption that a constant proportion of the waste stream is secondary paper (steel), reclaimable at a constant cost per unit (thus, a constant marginal cost), is not accurate. The reclamation of secondary material from the waste stream exhibits rising marginal cost, as more capital and labor are needed to reclaim an additional unit (of a given quality) from the waste stream.

Second, the accumulation of scrap steel and waste paper available in cities all over the nation presumes that the secondary materials market is a nationwide market. This is very unrealistic. Some secondary markets for light materials might be expansive, but for heavier materials, transportation costs make it unprofitable to ship the material over vast distances.

Miedema et al. take estimates for the supply elasticities of secondary materials from an external source. (They do this in the analysis of different tax rates. For the sensitivity analysis, they vary the elasticity values somewhat.) Since the outcomes of their model are so sensitive to the magnitudes of these elasticities, some insight into how these elasticities were determined would have been helpful. Some elasticities seem rather low, especially the one for glass.

3. The specification of demand for secondary materials

Bingham et al. specify the demand for secondary materials as follows: "The unit costs [of (virgin) raw material, $ per metric ton of output] can be viewed as maximum prices that would be paid for recycled materials in each region" (p.258). "The maximum prices paid for secondary" are the demand prices for secondary. There is a problem with this specification: Differing unit costs as defined above can come about via a) differing ratios of quantity of virgin material utilized per unit of output, b) differing prices paid for virgin materials, or both.

Regarding a): Since secondary materials are perfect substitutes for virgin materials (up to a technically given limit), there should be no difference in unit cost of virgin and secondary material. The authors seem to assume b), implying that "virgin material unit costs" is a synonym for "purchase prices for the virgin material". This would make the demand functions consistent, but it would require an explanation of why different plants face different virgin material prices.

Also, the virgin material unit costs are calculated in a rather indirect way which requires very restrictive assumptions: For individual packaging bulk material industries, regional unit costs were calculated by dividing the regional expenses on virgin raw materials by the regional number of employees. The latter is a proxy for regional output, for which data were not available. For this estimate to be valid, it has to be assumed that a) labor intensities are the same across regions and b) plant sizes do not differ across regions (or, alternatively, the production functions of the industries exhibit constant returns to scale - an assumption which is made anyway.)

In the study by **Miedema et al.**, demand for secondary material is a function of the elasticity of substitution between virgin material and secondary material in bulk manufacture. The production function for bulk manufacture is specified as a CES-function (constant elasticity of substitution) - a surprising choice. The authors make a very restrictive assumption about a variable on whose magnitude many results depend crucially.

3.4 TABLES

Table 3.1: Consumer product categories
(Bingham et al.)

A. Food and kindred products
- Perishables --
  1. Baked goods
     - bread and rolls; crackers and cookies; sweet goods
  2. Dairy products
     - cheese; eggs; milk; butter
  3. Frozen foods
     - ice cream; frozen desserts and baked goods; meat, fish, poultry; prepared foods; vegetables, fruits, juices, drinks
  4. Fresh and cured meat
  5. Fresh and cured fish and seafood
  6. Fresh and cured poultry
  7. Produce
- Beverages --
  8. Distilled Spirits
  9. Wine
  10. Beer
  11. Soft drinks
  12. Prepared beverages
      - cocoa; coffee; tea; breakfast drinks
  13. Candy and chewing gum
  14. Canned foods
      - canned vegetables; canned meat, fish, and poultry; canned fruits and vegetables; canned soups; canned baby foods, canned juices and fruit drinks; canned milk
  15. Cereals, flour, and macaroni
  16. Pet foods
  17. Tobacco products
  18. Other foods

B. General merchandise
- Household supplies --
  19. Soaps and detergents
  20. Other cleaning supplies
      - dry cleaners; laundry supplies; waxes and polishes; other cleaners and cleansers
  21. Pesticides
  22. Other household supplies
- Health and beauty aids --
  23. Packaged medications
  24. Oral hygiene products
  25. Cosmetics and hand products
  26. Hair products
  27. Shaving products
  28. Other beauty aids
  29. Other health aids
- Other general merchandise
  30. Other general merchandise

Table 3.2. Packaging materials
(Bingham et al.)

1. Flexible paper and paper closures
2. Flexible plastics and plastic closures
3. Metal closures
4. Flexible aluminum
5. Rigid paper
6. Rigid plastics
7. Glass
8. Steel
9. Rigid aluminum

Table 3.3. Materials and products
(Miedema et al.)

Materials	Product categories	
Rigid	Baked goods	Wine
Paper	Dairy products	Distilled spirits
Plastics	Meat and seafood	Soft drinks
Glass	Produce	Household supplies
Steel	Canned foods	Health and beauty aids
Aluminum	Other foods	Pet foods
Flexible	Prepared beverages	All other personal consumption expenditure
Paper	Beer	
Plastics		
Aluminum		

Table 3.4
Summary of effectiveness, 1970:
Tax on packaging with exemption for recycled materials
(Source: Bingham et al., p. 88, table 40.)

	Tax rate (dollars per ton of packaging)			
Measures of effectiveness	$10	$22	$50	$100
1.a. Reductions in solid waste generation (thousand tons)	198	395	783	1,402
1.b. % of packaging waste.	.45	.91	1.8	3.22
1.c. % of total solid waste	.18	.36	.72	1.29
2.a. Increases in the consumption of postconsumer waste materials (thousand tons)	3,894	5,911	8,742	9,703
2.b. % of packaging waste	8.93	13.56	20.05	22.25
2.c. % of total solid waste	3.57	5.42	8.01	8.89

Note:
Total packaging waste in 1970: 43,600,000 tons
Total municipal solid waste: 109,100,000 tons
Reduction in disposal is reduction in solid waste generation plus increase in consumption of postconsumer solid waste (i.e., recycling).

Table 3.5
Summary of effectiveness, 1970:
Regulation requiring the use of recycled materials
(Source: Bingham et al., p. 74, table 25.)

Measures of effectiveness	Recycling regulation (percentage recycled materials as share of packaging weight)		
	10%	20%	30%
1.a. Reduction in solid waste generation (thousand tons)	87	173	259
1.b. % of packaging waste	.2	.4	.59
1.c. % of total solid waste	.08	.16	.24
2.a. Increases in the consumption of postconsumer waste materials (thousand tons)	2,774	5,529	8,272
2.b. % of packaging waste	6.36	12.68	18.97
2.c. % of total solid waste	2.54	5.07	7.58

Table 3.6
Summary of effectiveness, 1970:
Tax on packaging, without exemption
(Source: Bingham et al., p. 88, table 39.)

Measures of effectiveness	Tax rate (dollars per ton of packaging)			
	$10	$22	$50	$100
1.a. Reduction in solid waste generation (thousand tons)	201	441	988	1,930
1.b. % of packaging waste	.46	1.01	2.27	4.43
1.c. % of total solid waste	.18	.4	.91	1.17
2. Increase in the consumption of postconsumer waste materials	-	-	-	-

Table 3.7
Summary of effectiveness, 1970:
Tax on containers
(Source: Bingham et al., p. 102, table 55.)

Measures of effectiveness	Tax rate (cents per container)			
	¢ 0.5	¢ 1.0	¢ 1.5	¢ 2.0
1.a. Reduction in solid waste generation (thousand tons)	1,549	2,317	2,766	3,183
1.b. % of packaging waste	3.55	5.31	6.34	7.3
1.c. % of total solid waste	1.42	2.12	2.54	2.92
2. Increase in the consumption of postconsumer waste materials	-	-	-	-

Table 3.8
Parameter points used in sensitivity analysis
Summary of Parameter Points
(Source: Miedema et al., p. 60)

Parameter Point	Substitution Elasticity	Secondary Supply Elasticity	Virgin Supply Elasticity
1	Low	Low	Infinite, except rigid and flexible paper
2	High	Low	Infinite, except rigid and flexible paper
3	Low	High	Infinite, except rigid and flexible paper
4	High	High	Infinite, except rigid and flexible paper
5	Best	Infinite	Infinite

Table 3.8 (Con't)
Parameter points used in sensitivity analysis
(Source: Miedema et al., p. 61)

	Parameter point (elasticities)*										
	1+		2+		3+		4+		5+		
Material	Substitution	Secondary supply	Substitution	Secondary supply	Substitution	Secondary supply	Substitution	Secondary supply	Substitution	Secondary supply	Virgin supply
Rigid											
Paper	2.0	0.5	4.3	0.5	2.0	1.7	4.3	1.7	3.15	∞	∞
Plastic	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	∞
Glass	∞	0.1	∞	0.1	∞	0.7	∞	0.7	∞	∞	∞
Steel	2.6	0.4	10.4	0.4	2.6	0.8	10.4	0.8	6.5	∞	∞
Aluminum	∞	1.1	∞	1.1	∞	4.3	∞	4.3	∞	∞	∞
Flexible											
Paper	0.4	0.5	4.9	0.5	0.4	1.7	4.9	1.7	2.65	∞	∞
Plastic	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	∞
Aluminum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	∞

*For the derivation of these parameter values, see appendix F.

+The virgin supply elasticities are infinite (very large) for all bulk materials except rigid and flexible paper. For these materials, a value of 0.49 was used for the virgin supply elasticity.

3.5 FIGURES

Figure 3.1
Product charge impacts on total waste stream in 1985
(Source: Miedema et al., p 45)

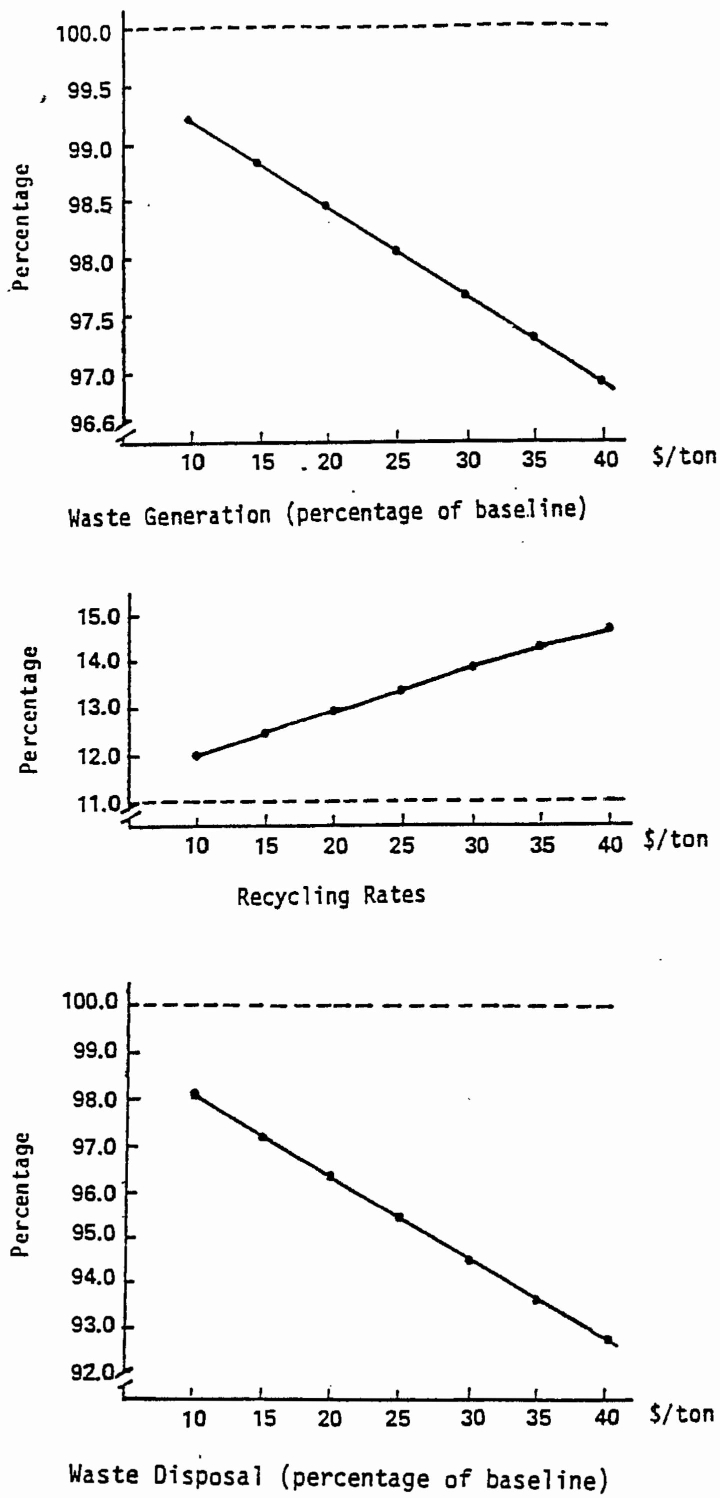
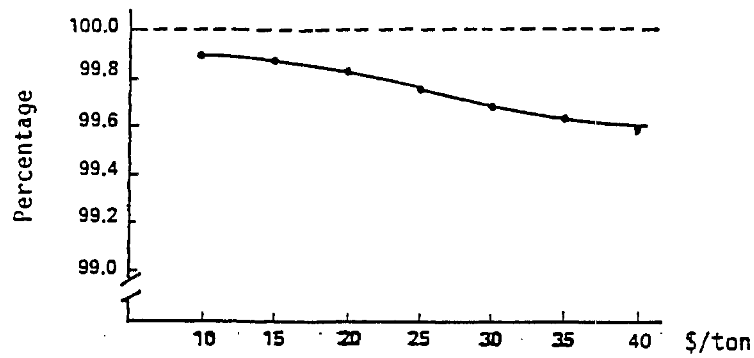
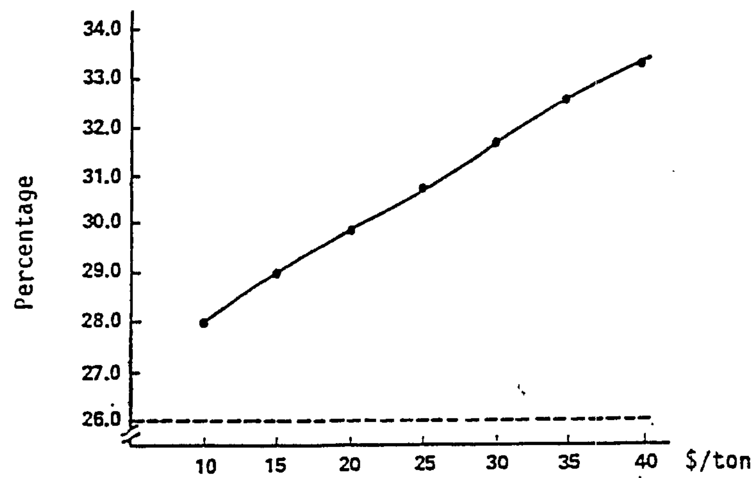


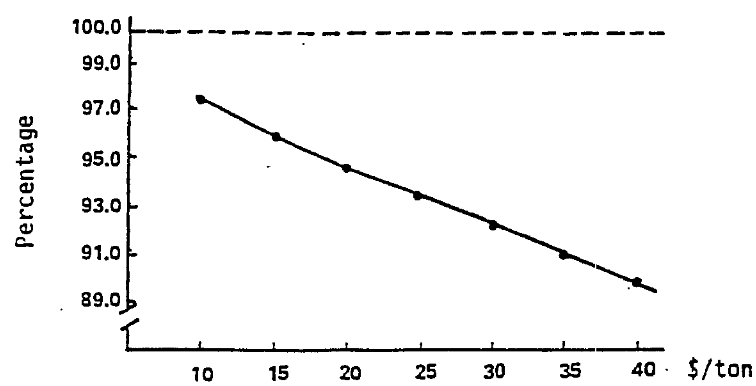
Figure 3.2
Product charge impacts on rigid paper in 1985
(Source: Miedema et al., p 48)



Waste Generation (percentage of baseline)

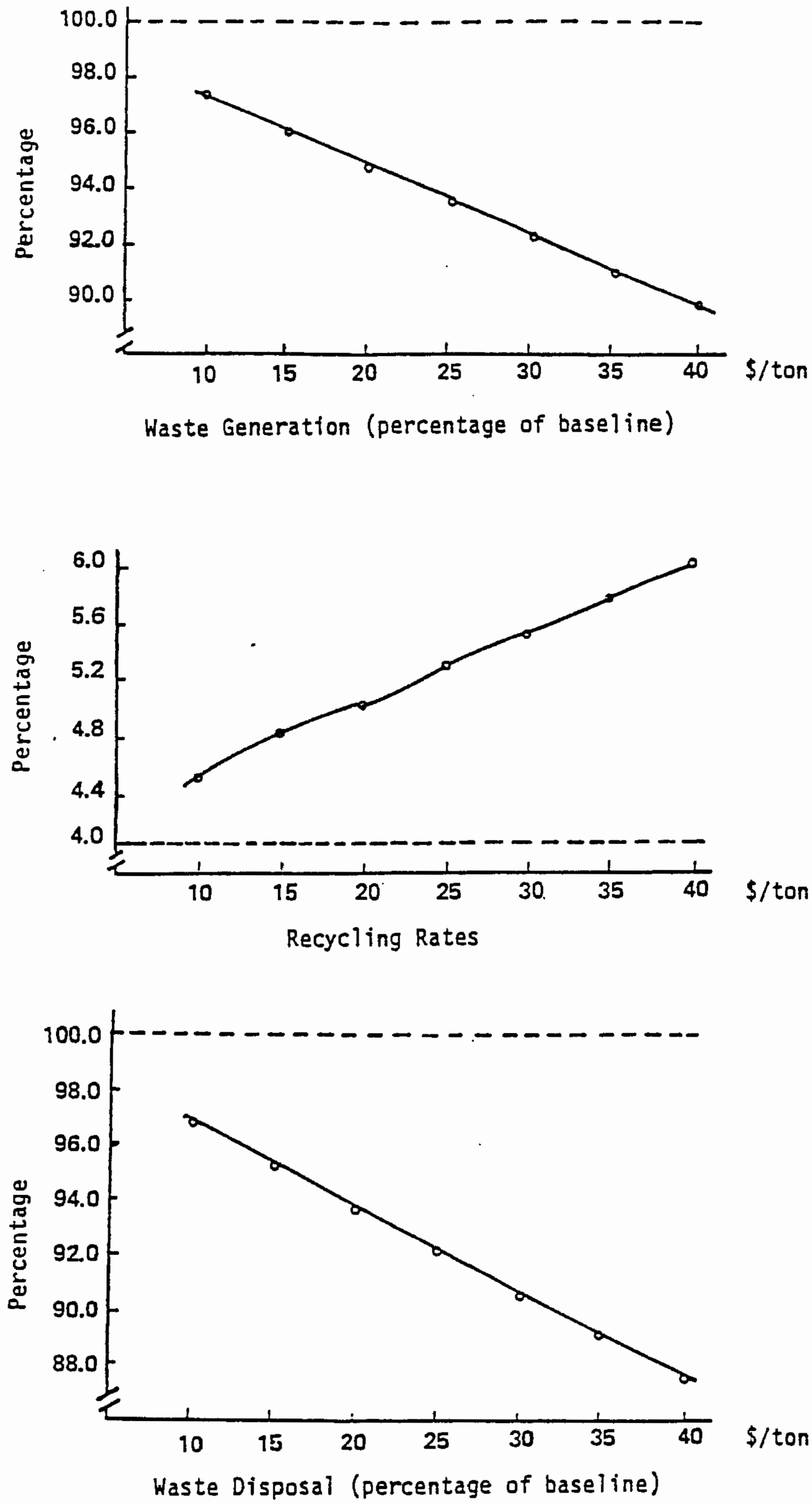


Recycling Rates



Waste Disposal (percentage of baseline)

Figure 3.3
Product charge impacts on glass in 1985
(Source: Miedema et al., p 49)



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89-024/3NJ

Assessment of Impacts of Production and Disposal of Consumer Packaging on the Environment

NJ DEPE Contract P31152

Report #3:

The Marginal Cost of Handling Packaging Materials in the New Jersey Solid Waste System

prepared for:

New Jersey Department of Environmental Protection and Energy

prepared by:

Tellus Institute
89 Broad Street
Boston, MA 02110
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May 1992

(printed on recycled paper)

PREFACE: NEW JERSEY'S CURRENT SOLID WASTE MANAGEMENT

a statement by New Jersey Department of Environmental Protection and Energy, May 1992

The New Jersey Department of Environmental Protection and Energy was a cofunder in this overall research process and initiated the packaging research proposal. In this regard, New Jersey was selected and used as the pilot for the collection of solid waste disposal data. While the report's conclusions can be transferred to other state databases, it is important to understand some of the basis for the information used in the report.

The report represents a point in time in New Jersey's solid waste management system. It is noted by Tellus in Section II that New Jersey's counties are in transition and that the ultimate configuration of the integrated solid waste disposal system developed by the counties is still, in part, developing. What might have been reasonable scientific assumptions to make in the Spring/Summer of 1990, the point in time investigated by Tellus, are no longer accurate because of that continued development.

The Governor's Emergency Solid Waste Assessment Task Force in their Final Report, published in August 1990 and endorsed by Governor Florio in November 1990, established a goal of 60% recycling of the total waste stream within five years. Currently New Jersey is recycling 46% of its total solid waste stream. In addition the Task Force established a source reduction goal which caps 1990 per capita generation rates over the next five years, caps total waste generation by 1995 and reduces total waste generation by the year 2000. The remaining 40% will be managed through regionalized resource recovery facilities including composting and sanitary landfills.

One of the most important aspects to note in the overall cost is that New Jersey's solid waste tip fee includes an additional tax of $14-$20 per ton on the disposal fee. These taxes pay for various solid waste programs in New Jersey's system. This is one of the reasons New Jersey's tip fee structure, on an overall basis, appears higher than most states. In addition, in some counties the tip fee at the landfill or resource recovery facility pays for the entire or parts of the entire solid waste system. The Warren County RRF tip fee includes bypass, nonprocessible and ash disposal costs and the Cape May County SLF tip fee includes the IPF, bulky waste processing/recycling and leaf composting costs.

The changes that have occurred in the New Jersey solid waste management plan since the Tellus report developed its database include the following:

<u>Source Reduction</u>:

Source reduction is defined to encompass activities which eliminate or reduce the weight or volume of materials, decrease the toxic components contained within products and packaging, and increase product durability, reuse, refillability, and repair. Source reduction is recognized as the first priority in the state's solid waste management program. The overall goal for source reduction is to cap per capita generation of waste at 1990 levels, cap total waste generation within five years and then reduce total waste generation within 10 years.

Recycling:

New Jersey's Statewide Mandatory Recycling Act established a 25% recycling goal for the municipal solid waste stream. The present recycling goal established by the state is at least 60 percent of the total solid waste stream and at least 50 percent of the municipal solid waste stream. New Jersey recycled 34% of the municipal solid waste stream and 46% of the total solid waste stream.

Resource Recovery:

Bergen, Pennsauken, Somerset, Monmouth and Passaic County will no longer be building mass-burn RRF's. Middlesex and Hudson, while still in the plan, are considered inactive. There are currently four operational mass burn RRFs, one under construction and one more in the permit stage. Burlington, Cape May, Somerset, and Ocean County are finalizing plans and in some cases permitting alternate technology RRF's including compost facilities and refuse derived fuel facilities, in an overall integrated management approach.

Landfills:

Middlesex County's landfill has been expanded as permitted by the Department. While there are 11 counties currently using landfills in-state for their primary and long-term means of disposal they must meet the source reduction and recycling goals in addition to evaluating regionalization for all remaining components of their plan.

Regionalization:

Several counties have established regional agreements such as Atlantic and Mercer County, Somerset, Hunterdon and Warren, Bergen and Essex County's and Bergen and Union County to share solid waste disposal and management capacity thereby redistributing the overall costs and benefits of the systems. The long-term goal is to establish regional facilities to increase the state's self-sufficiency in solid waste management.

Self-Sufficiency:

While New Jersey was a state that had relied on out-of-state disposal through long haul transfer stations, with the above facilities on-line New Jersey is again becoming a self-sufficient state in terms of solid waste disposal. Currently, New Jersey exports only 20 percent of its total waste generation. The remainder of the waste generated is managed in-state through recycling and disposal. This number has decreased over the last two years based on an increased recycling rate, new disposal facility capacity and regional agreements.

REFERENCE LIST:

The Preliminary and Final Task Force Reports, Task Force Guidance Document and the State Plan (when available) may be obtained by writing to:

New Jersey Department of Environmental Protection and Energy
Division of Solid Waste Management
840 Bear Tavern Road
CN 414
Trenton, New Jersey 08625

Table of Contents

I. INTRODUCTION

Packaging materials are a fast growing and controversial portion of the waste stream. Reduction of the amount of packaging materials has been difficult in part because neither the producer nor the consumer is aware of the disposal cost of the product. As states begin to develop policy to manage and reduce these materials, policymakers need to understand the economic impacts of these materials on the solid waste system. Quantifying these impacts is a first step toward developing policy measures affecting the mix or quantity of packaging in the waste stream.

This section of the Tellus Institute Packaging Study addresses this need by investigating the economic impacts of various packaging materials on the solid waste system. That system includes collection programs and the facilities to either process recyclables or dispose of waste through incineration, landfilling or transfer to out-of-state facilities.

To measure economic impact, we have estimated the marginal cost to the solid waste system of an additional ton of each material. The marginal cost is the increase or decrease in the cost of the solid waste system when the amount of a particular material in the waste stream is increased or decreased. Policy is best guided by marginal costs, which measure the impact of policy-induced changes rather than by average costs. In later stages of the Tellus Institute Packaging Study, will use these figures to determine the marginal costs of various packaging options for specific product types.

Any variability in the costs of managing different packaging materials results from differences in their characteristics. Material density, recycling rate, material revenue, Btu content and ash content all affect the costs of various programs and facilities.

In brief, the methodology for producing marginal costs is a two-step process. First, a model of the cost structure of the solid waste system was developed. The State of New Jersey's residential solid waste system was used as the basis for the waste management practices and costs used in the model. Data was collected on the types of collection programs and recycling and disposal facilities used in New Jersey for handling residential waste, along with the costs of these systems. The Tellus Institute WastePlan model, a computer-based solid waste planning tool, served as the tool for both modelling the cost structure of the New Jersey Solid Waste System, and for conducting this economic analysis.

Second, for each packaging material, we determine the percent recycled, buried, burned, or transferred out of state, based upon the current New Jersey Solid Waste System, and planned changes to the types of disposal capacity in use.

Third, marginal costs for each packaging material were estimated by individually increasing the amount of each packaging material in the waste stream. The resulting cost increase calculated by WastePlan for handling this additional material is the marginal cost. It is comprised of the cost increases to each of the solid waste system components based upon the percentage of the additional material that is recycled, buried, burned, or transferred. Marginal costs have been produced for the entire solid waste system and for individual collection programs and facilities.

The various types of packaging in the waste stream have been broken down into eight categories: aluminum, glass, ferrous, corrugated cardboard, paper packaging (paperboard), PET plastic, HDPE plastic and non-recyclable plastic containers. The categories broadly cover all of the types of packaging found in the residential waste stream.

The results of this report are summarized in the table below, in which the marginal costs of each packaging material on the New Jersey solid waste system are reported by both weight and volume. Aluminum has the lowest marginal costs whether measured by weight or volume. Other materials are ordered roughly according to their density, which is also listed in the table. Curbside (as-disposed) densities are listed, though material density will increase as products are compacted in garbage trucks or disposed in landfills. Denser materials, particularly glass, have lower costs per ton. But less dense materials, particularly plastics, have lower costs per cubic yard. However, all materials do have a significant cost impact upon the solid waste system. Clearly, the lowest cost option, not considered here, is packaging reduction -- at zero disposal cost. A full analysis of the impacts of various material characteristics and recycling factors upon marginal costs is presented in Section V.

Summary of Packaging Material Marginal Costs

	Marginal Cost Per Ton	Marginal Cost Per CY	Material Density
Aluminum	23.73	0.71	60
Ferrous	134.30	10.07	150
Glass	70.98	23.07	650
Corrugated	118.18	8.86	150
Other Paper	110.47	9.67	175
PET	249.80	4.37	35
HDPE	243.45	4.26	35
Other Plastics	235.06	4.11	35

The remainder of this report is divided into four sections. The next two sections report on the modelling of the New Jersey solid waste system: Section II describes the methodology for creating the model and the results of the New Jersey scenarios; Section III reports solid waste data from New Jersey and the assumptions used in the WastePlan model.

The final two sections report on the analysis of marginal costs: Section IV reports on the methodology and Section V reports the results of the analysis. The results include not only total marginal costs for each material, but marginal costs in each of the solid waste management programs and facilities modelled.

II. MODELLING THE NEW JERSEY SOLID WASTE SYSTEM

In order to estimate the marginal waste management costs of packaging materials, we first developed a cost model of the New Jersey solid waste system. This model replicates the generation of residential waste and the costs of systems to collect, process and dispose of waste. Only current costs and conditions (as of summer-fall 1990, when the report was written) are included in the analysis. Future changes in waste generation and new waste management plans were not considered. WastePlan, a computer based solid waste planning tool developed by the Tellus Institute, was used in producing this model. Data inputs to the model were collected from New Jersey state agencies, counties, and municipalities as well as other sources. This section describes the structure of the model, the types of systems analyzed and the methods used to model specific systems.

County Groupings

Our New Jersey solid waste model divides the state into 7 different regions. Atlantic, Camden, Ocean and Somerset counties were modelled individually. All other counties were grouped into one of three scenarios, based on their primary method of waste disposal. Originally, we had considered modelling each of the 21 counties in New Jersey, but due to data limitations and modelling feasibility, we adopted a more modest approach. The four counties modelled individually were chosen to reflect a range of disposal options, levels of recycling collection and processing and demographic conditions.

The remaining 17 counties have been placed into three groups, defined by the primary disposal method that each county will use in the early 1990's. Because many counties' disposal methods are in transition, an effort was made to determine what the method would be used in the next three to five years in each county. The counties included in these three scenarios are listed below.

Incinerator Counties	Bergen Essex Gloucester Hunterdon Passaic Union Warren
Landfill Counties	Burlington Cape May Cumberland Monmouth Salem Sussex

Transfer Counties	Hudson Mercer Middlesex Morris

Currently there are 11 counties which use landfills exclusively or as the primary method of disposal, 9 which transfer waste out of the county, and only 2 that rely exclusively on incineration (Camden County uses both a landfill and transfer station). The future of these systems is somewhat uncertain. The Emergency Solid Waste Assessment Task Force set up by Governor Florio has recently submitted its recommendations for a regional disposal plan which may influence the county solid waste planning boundaries previously set forth in legislation. If implemented, these plans affect the number of incinerators the state will eventually have and will likely also affect future recycling capacity.

In addition, a number of counties are in a transition period in their solid waste infrastructure. Several incinerators are under construction and several more await final permitting. At the same time, many counties are in the process of constructing or procuring advanced recycling facilities.

However, it is clear that a number of landfills will close in the near future, and at least some new incinerators and recycling facilities will be built. In grouping counties by disposal type, we have made assumptions based upon conversations with county and DEP officials about what the most likely future disposal source would be for each county. We assumed that incinerators would be built in Camden, Bergen, Essex, Passaic, and Union Counties. Also, we assumed that Gloucester will use incinerators more than at present and that Warren and Hunterdon counties will continue to use the Warren incinerator. Elsewhere, we assumed that incineration would not be the primary disposal method.

Of the 11 counties currently using landfills, 6 had at least 6 years worth of remaining capacity. These, as well as Monmouth and Cape May, which have the ability to expand their landfills, were grouped as landfill counties. The remaining counties either currently use transfer stations, or will soon be out of landfill space; we therefore classified them as transfer counties.

In the four counties individually modelled, the costs of collection programs and facilities were modelled based on real costs whenever possible, constrained at times by the lack of complete and reliable information. In the three scenarios with multiple counties, costs are based upon the averages across these counties, or across all New Jersey programs in some situations. The exact procedures used for modelling the different types of systems are described in the next section.

In all three of the multi-county scenarios, recycling collection and processing activities are a mix of commingled and multiple-material separation programs. The types of recycling facilities likewise include a large mix from highly automated intermediate processing facilities (IPFs), such as the newly opened facility in Cape May, to small, municipally operated depots where separately collected materials are processed and marketed. The recycling programs modelled in each scenario are intended to represent the appropriate mix of these different program alternatives. Similarly, the modelling of garbage collection is intended to represent the mix of different alternatives, collection frequency, operator (municipal vs. private) and crew size found in the different counties.

WastePlan

WastePlan is a solid waste management planning tool which simulates the generation of a waste stream from a particular region and the operations and costs of the systems which collect, process and dispose of that waste stream. The process of modelling the solid waste system is broken down into three modules: generation, collection and facilities. Generation calculates the waste stream size and composition based upon the demographic characteristics of the region being modelled.

Recycling, composting and garbage collection are modelled in the collection module. Information such as the percentage of materials diverted (for recycling and compost programs), truck type with cost and physical characteristics, collection schedule, and collection efficiency (number of households collected per hour) is input into WastePlan. The program then 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 costs.

Finally, the different facilities needed to process and dispose of collected materials are modelled in the facilities module. The types of facilities which can be modelled includes drop-off recycling, recycling processing facilities, compost sites, incinerators, landfills and mixed solid waste composting facilities. Materials flow through each facility, and annual facility costs, are calculated; a system summary reports overall material quantities handled by each waste management system and their associated costs.

For each region in New Jersey which we modelled, we developed a separate WastePlan scenario with the region's population and waste generation characteristics, recycling and garbage collection programs, recycling and disposal facilities, and costs of these systems. These WastePlan scenarios serve as the "base case" quantitative description of current New Jersey conditions, from which the marginal costs are calculated.

Types of Solid Waste Systems Modelled

The solid waste systems modelled include only the collection and processing of the residential waste stream. All residential solid waste activities which involve packaging materials are modelled; thus our model omits composting, bulky and white goods collection and household hazardous waste collections.

Composting operations, though an important part of New Jersey's solid waste program, have not been modelled because packaging materials are not composted. Existing residential compost programs handle only vegetative waste, such as leaves, brush or grass clippings. Though some counties are considering mixed waste composting facilities which handle most organics in the waste stream (including packaging materials such as corrugated cardboard and paperboard), none have been constructed at present. Therefore, the costs of the composting system will not be affected by changes in quantities of individual packaging materials. However, composted leaves and yard waste are removed from the waste flow in our model, so that accurate tonnages are recorded at disposal sites. Bulky and white good collections and household hazardous waste collections are similarly ignored: they also do not affect packaging materials.

Commercial waste is not included in this analysis, due to the difficulty in accurately modelling the collection of commercial recyclables and waste. Commercial waste generators are a very heterogeneous group, generating waste of variable quantity and composition, and most importantly, being serviced by different sized containers and different frequencies. Because of this diversity, quantitative, per-unit modelling of these collections is problematical. In contrast, for residential collections each generator is a household producing a more nearly similar composition and quantity of waste and subscribing to the same type of service. For disposal facilities, the situation is simpler: commercial and residential waste is accepted at the same facilities so marginal costs will be the same. Marginal costs of recycling facilities may be different because residential and commercial recyclables may be processed at different facilities with different cost structures.

The exclusion of commercial waste from the analysis will likely not have a large impact upon the marginal costs of aluminum, ferrous, and glass containers, as the majority of these materials are generated in the residential waste stream. The plastic containers considered in this analyses focused primarily upon materials used for consumer use, such as PET, HDPE and to a lesser extent PVC, Polystyrene and films. There are a large amount of plastic packaging and containers in the commercial waste stream, though they tend to be more film, banding straps and less actual containers. These plastics packaging uses were not addressed.

Excluding commercial wastes will likely have a large impact upon the resulting marginal costs for corrugated cardboard. The majority of the corrugated cardboard in the waste stream is used by commercial businesses for bulk packing, so the costs for the collection of the majority of corrugated cardboard will not be factored into the analysis.

These collection and processing costs will probably be very different for commercial business than the estimates produced for residential programs. In businesses, corrugated is collected in dedicated bulk containers or compactors, which greatly decreases the costs of both collection and processing. However, costs will not be uniform for all businesses. Smaller businesses will have higher costs since they will not be able to afford compactors, and often do not generate enough material to economically justify separate collections.

We modelled the collection of residential recyclables through both dropoff and curbside collections. The dropoff programs modelled all utilize the same basic system. Drop boxes are positioned at various places throughout the county where residents can drop off recyclables. These materials are then collected and delivered to the central recycling facility in the county where they are processed. Only the costs of the containers and transportation of the materials are included in dropoff collection. All processing costs are included in the costs of the recycling facility.

Curbside collection of recyclables is modelled using WastePlan's collection module. Materials are collected from both single-family and multi-family households and a variety of different collection mechanisms are modelled, including private and municipal collection, and commingled and material-separated setout. The percentage of single family and multi-family households is based upon actual New Jersey data.

All materials from dropoff and curbside collections are brought to a central processing facility, often called an intermediate processing facility, or IPF. A wide variety of IPFs exist in New Jersey, representing a wide range of sizes, types of technology and types of materials accepted. For the four individual county scenarios, data is based on actual facility information, while in the three multi-county scenarios, data is based upon averages of a variety of actual facilities in New Jersey.

As with recycling collection, garbage collection is modelled for both single and multi-family systems. Due to the relatively small amount of data received, cost averages were developed from data provided by a handful of municipalities. Different averages were used for programs differing in municipal vs. private operation, frequency of collection and crew size.

Collected garbage was sent to one of three types of facilities: an incinerator, landfill or transfer station. In the four individual county scenarios, the actual costs of the county's disposal facility was used, while in the three multi-county scenarios, costs were averaged based on the facilities in all of the counties covered in this scenario.

The development of data for the solid waste systems will be presented in greater detail in Section III, where actual information collected from New Jersey will be presented along with the data used in the WastePlan model.

Results of the Scenarios

Presented below is a brief description of the program assumptions made for each of the seven scenarios, along with the costs calculated by the cost model. A summary of the costs for each scenario is presented in Table 1. The per ton costs of each solid waste option is presented in Table 2.

Table 1 - Baseline WastePlan Output for New Jersey

	Atlantic	Camden	Ocean	Somerset	Incinerator	Landfill	Transfer	Total
Recycling Collection ($)								
Single Family	980,189	1,939,418	2,534,528	1,230,087	8,927,955	5,984,251	7,027,501	28,623,929
Multi-Family	42,956	189,759	86,057	65,764	909,606	401,061	616,731	2,311,934
Single Family Tonnage	16,106	35,297	39,746	26,162	178,853	120,614	131,664	548,442
Multi-Family Tonnage	1,050	6,564	2,583	3,018	27,152	14,228	19,669	74,264
Garbage Collection ($)								
Single Family	5,889,955	12,268,580	12,332,670	5,498,167	78,730,840	42,059,512	50,951,580	207,731,304
Multi-Family	319,837	1,234,592	459,120	374,973	11,434,720	2,899,404	4,406,565	21,129,211
Single Family Tonnage	112,530	239,856	216,460	104,815	1,490,406	697,825	967,314	3,829,206
Multi-Family Tonnage	14,041	53,426	19,393	15,692	334,724	95,268	192,782	725,326
Recycling Facility ($)	795,512	254,000	693,685	596,341	3,083,990	2,063,101	3,026,666	10,513,295
Tonnage	17,156	41,861	42,329	29,181	205,599	134,843	151,333	622,302
Incinerator Cost		28,527,900			143,941,296			172,469,196
Tonnage		293,282			1,825,129			2,118,411
Landfill Cost			17,560,160			51,384,448		68,944,608
Tonnage			239,239			793,092		1,032,331
Transfer Cost	12,406,980			13,898,130			135,882,022	162,187,132
Tonnage	126,571			120,507			1,160,096	1,407,174
Total Cost	20,435,429	44,414,249	33,666,220	21,663,462	247,028,407	104,791,777	210,911,065	673,910,609
Total Tonnage	143,727	335,143	278,182	149,688	2,031,135	927,935	1,311,429	5,177,239

Table 2 - New Jersey Current Scenario

Per Ton Costs	Atlantic	Camden	Ocean	Somerset	Incinerator	Landfill	Transfer
Recycling Collection	59.64	50.86	61.91	44.41	47.75	47.35	50.51
Garbage Collection	49.06	46.04	54.24	48.74	49.40	56.69	47.72
Recycling Facility	46.37	6.07	16.39	20.44	15.00	15.30	20.00
Incineration	·	97.27			78.87		
Landfill			73.40			64.79	
Transfer	98.02			115.33			117.13
Total	142.18	132.52	121.02	144.73	121.62	112.93	153.96

Atlantic County

Atlantic County operates a county-wide program for the collection of recyclables. Materials are collected commingled and bi-weekly using Eager Beaver recycling trucks. In addition, there are 14 dropoff sites throughout the county. Collected materials are sent to a county processing facility which relies almost solely upon manual sorting of materials. Though the county is considering procuring a new facility with more automated separation in the near future, no estimates or proposals for its costs were available. Therefore, the cost information on the existing facility was used.

Garbage is collected by a combination of municipal crews and private haulers, with some rural residents self-hauling to a public transfer station. About 90% of the waste is sent to a publicly owned, privately operated transfer station in Egg Harbor Township. For this study, we have assumed that all waste goes through this transfer station at a tip fee of $98 per ton. The county has plans to develop an incinerator facility, but the development is only in its initial stages. We assumed that the county will continue to transfer wastes for the foreseeable future.

Total waste management costs modelled in our scenario are about $20.4 million, or about $142 per ton. Waste disposal is the largest single item at $12.4 million annually, with garbage collection the next largest at $5.9 million annually.

Of particular note are the high costs of the recycling processing facility, which relies heavily upon manual labor for the separation of material. This process is less efficient than automated separation, and results in a $46 per ton processing cost for recyclables, which is about $26 per ton larger than in any of the other scenarios. Recycling collection costs about $60 per ton, the second highest of all the scenarios. In each of the seven scenarios, the recycling collection costs and weights are the combination of single-family curbside collection, multi-family bulk container collection and satellite dropoff centers.

Camden County

Commingled, curbside collection is provided by municipalities in all 37 towns in Camden County. In addition, there are four dropoff sites which accept only plastics. The frequency of collection varies from once a week to once a month. The materials are sent to the IPF in Camden, which is publicly owned and privately operated. This is one of the first automated separation facilities, and has been operating since 1986. Like many recycling facilities in New Jersey (particularly other private facilities), it accepts only the mixed container portion of the recyclables stream. Newspapers and corrugated cardboard are simply dumped at the facility site and sold directly to a broker in this unprocessed form.

Garbage is collected by municipalities, either with municipal crews or through a contract with private haulers. Material from about 9 towns is currently going to the Pennsauken landfill, and the remainder of the material is being shipped out of state. However, a new incinerator is being developed in Camden which will likely come on-line within the next two years. Because this facility will likely come on line soon, we have assumed all of the county's waste will be incinerated.

Camden County's total recycling and disposal costs are about $44.41 million annually, or about $132 per ton. The largest cost is for the incineration of waste, which totals $28.5 million annually, or about $97 per ton. Garbage collection is next largest at $12.3 million annually, or $46 per ton. Recycling processing costs have the lowest per ton net costs of all scenarios, at $6 per ton. The annual net cost of the facility is only $0.25 million, though it handles about 12% of the waste stream. Recycling collection costs about $50 per ton.

Ocean County

Commingled curbside collection is available in 26 of the 33 Ocean County towns, while the other 7 towns have material separated collection. Most materials are delivered to the Ocean County Recycling facility, which is a very basic, manual sorting facility, though a few towns deliver to Rossetta Recycling, a merchant facility. However, Ocean County has just signed an agreement with Empire Returns to develop a new, automated recycling facility which should be completed in the next few years. Once this is completed, it is anticipated that all towns will deliver to the new facility and provide commingled collection. We modelled the anticipated new recycling facility and countywide commingled collection.

Garbage is collected primarily by the municipalities, with about half using municipal crews and the other half relying on private contracts. About 10% of residents contract directly with haulers. Waste is sent to the privately operated Ocean County Landfill. Estimates for the current site's remaining lifetime range from six to seven years; there is also the potential for future expansion once the landfill nears capacity. In addition, Ocean county is exploring MSW composting as an option.

Ocean County's total recycling and disposal costs are $33.7 million annually, or $121 per ton. Landfill disposal is about $17.6 million annually, or $73 per ton; the cost of garbage collection is $54 per ton, the second largest for any of the scenarios. Recycling processing costs are relatively low at $16 per ton, though the recycling collection costs are the largest of any scenario at roughly $62 per ton.

Somerset County

Like Atlantic County, Somerset County provides curbside collection of recyclables for all of its towns. Collection occurs bi-weekly and is made primarily with side-loading compartmentalized trucks. Recyclables are currently sent to the Somerset County Recycling Center which is owned and operated by the county. Commingled materials are manually separated and undergo minimal processing before marketing. The county is planning to develop a newer facility which will have automated separation and more extensive material processing. We have modelled the future processing facility in our scenario for Somerset County.

The majority of residents in Somerset County contract individually with private haulers, though about 15% receive some form of municipal collection. Waste goes to one of two private transfer stations operating in the county. All residential waste goes to Bridgewater Resources while Somerset Intermediate Recycling Center (SIRC) takes some commercial waste. Bridgewater Resources sends some material to a landfill in Taylor, Pennsylvania and some to the Warren County, New Jersey incinerator.

Total waste collection and processing costs in Somerset County are $144 per ton, the second highest of all scenarios; the total is $21.7 million annually. Transfer of waste out of state is relatively expensive at $115 per ton, while garbage collection is low at $48 per ton. Recycling facility costs are about $0.6 million annually, or $20 per ton, while collection is $1.3 million annually, or $44 per ton.

Incinerator Counties

Counties included in the incinerator scenario include Bergen, Essex, Gloucester, Hunterdon, Passaic, Union, and Warren. Currently, the only operating incinerators in New Jersey are in Essex, Gloucester, and Warren Counties. Hunterdon has a long term arrangement to send its waste to the Warren facility. The Bergen, Passaic and Union facilities have all undergone most permitting procedures and only await final approval before construction can start. In some cases, holdups in federal approval of NO_x emissions is the only barrier between commencing construction.

Total waste management costs in the incinerator counties are relatively low at $122 per ton, resulting in a total cost of $247.0 million annually. The average costs of incinerators are $79 per ton, while the costs for garbage collection are $49 per ton. Recycling collection costs are $48 per ton, while the average of the costs of the recycling facilities is $15 per ton. As in Somerset County, the costs of recycling collection is slightly less than garbage collection because recyclables are collected less frequently.

Landfill Counties

Counties included in the landfill scenario include Burlington, Cape May, Cumberland, Monmouth, Salem and Sussex. Counties are included in the landfill scenario if the remaining lifetimes of their landfills are six years or more, or if the opportunity exists for in-county landfill expansion. Salem and Sussex counties have long remaining landfill lifetimes, of 18 and 24 years (see Table 16). Burlington, Cape May and Monmouth have shorter remaining landfill lifetimes of 2.3 to 6.5 years. However, Cape May and Monmouth, with the shortest remaining lifetimes, will likely develop extensions to their current landfills and continue landfilling into the foreseeable future (given appropriate resolution to site specific issues, such as pinelands protection). In addition, Cape May is considering adoption of MSW composting.

Per-ton total costs in the landfill counties are the lowest of all scenarios at $113 per ton, resulting a total cost of $104.8 million annually. This low cost is due to the average landfill cost of $65 per ton, the lowest garbage disposal cost paid in any of the scenarios. However, some of the disposal savings is offset by the costs of garbage collection, which are the highest of all scenarios at $57 per ton. Recycling collection costs are $48 per ton. As in some other scenarios, recycling collection is cheaper than garbage collection because it is less frequent. Recyclables processing is relatively cheap at a net cost of $15 per ton.

Transfer Station Counties

Counties included in the transfer station scenario include Hudson, Mercer, Middlesex and Morris. Hudson, Mercer and Morris Counties are currently sending waste out of state via transfer stations. Though Hudson and Mercer have incinerator proposals under technical review by DEP, it is anticipated that these facilities either will not be constructed, possibly due to changes proposed by the governor's Task Force, or will be delayed three to four years or more. Hence we did not model these facilities.

Middlesex County is currently landfilling much of its waste at the Edgeboro landfill, which is expected to close within the year. Middlesex also has an incinerator proposal under DEP review but, as in Hudson and Mercer, it is not likely that the facility will be constructed in the near future. Until a permanent disposal site is obtained, the county will likely ship waste out of state, or possibly in-state to landfills or incinerators in other counties.

The transfer station counties' total waste management costs are the highest for all scenarios at $154 per ton, or a total cost of $210.9 million. This high cost is due to transfer station costs which are $117 per ton, the highest disposal cost paid by any of the counties. These costs include transfer station processing, shipping and final disposal costs. Garbage collection is $47 per ton; recycling collection is $50 per ton and processing is $20 per ton.

III. NEW JERSEY SOLID WASTE DATA

Data for the New Jersey solid waste model was collected from a wide range of sources including county recycling and solid waste plans, New Jersey Department of Environmental Protection reports, independent periodicals and reports and a survey of county solid waste and recycling coordinators and municipal coordinators conducted by Tellus. The primary focus of data collection was on program and facility operating costs though demographic data, waste generation and composition data, and many other program detail were required as well to perform the WastePlan analysis.

This section report some of the actual data collected from the various New Jersey sources together with assumptions (based on the data) used in creating WastePlan inputs.

We will first present the demographic information on New Jersey counties, then the waste characteristics, and finally the costs and operating characteristics of the collection programs and facilities.

DEMOGRAPHICS

Demographic information was collected for each of the 21 counties within New Jersey. Population, people per household, percentage of multi-family housing, and total road miles are the basic information required for WastePlan. A summary of this information for each county can be found in Table 3.

Table 3 - New Jersey Demographics

COUNTY	Population (1)	People per House-hold (2)	Multi-family House-holds (3)	Road Miles (4)
	(000)		(%)	
Atlantic	217	2.53	11	1682
Camden	506	2.76	18	1660
Ocean	420	2.52	8	2344
Somerset	226	2.81	13	1233
Landfill	1401	2.50	12	8358
Incinerator	3066	2.80	17	9944
Transfer Station	1980	2.70	15	5799

1 1990 estimates from the Department of Labor Population Projections 1990-2010 (as listed in Preliminary Report of Emergency Solid Waste Assessment Task Force, July 6, 1990).

2 Atlantic, Camden, Ocean and Somerset figures are from Census Bureau's 1985 "Estimate of Households for Counties."
Data for other counties derived from 1980 population and housing statistics in 1980 Housing Census.

3 Five or more units in structure, as listed in 1980 Housing Census.

4 Sum of County and Municipal Roads, as reported in 1988 New Jersey Department of Transportation.

Table 3 - New Jersey Demographics

LANDFILL COUNTIES	Population (1)	People per House-hold (2)	Multi-family House-holds (3)	Road Miles (4)
	(000)		(%)	
Burlington	401	2.99	15	1989
Cape May	100	2.70	10	798
Cumberland	140	2.81	11	1143
Monmouth	566	2.70	19	2508
Salem	66	2.67	10	770
Sussex	128	2.64	6	1150
Total	1401			8358
Average		2.75	12	

Table 3 - New Jersey Demographics

INCINERATOR COUNTIES	Population (1)	People per House-hold (2)	Multi-family House-holds (3)	Road Miles (4)
	(000)		%	
Bergen	841	2.75	18	2677
Essex	841	2.68	36	1570
Gloucester	220	2.88	12	1196
Hunterdon	101	2.90	6	1097
Passaic	473	2.79	20	1193
Union	503	2.75	19	1320
Warren	87	2.71	13	891
Total	3066			9944
Average		2.78	18	

Table 4 - Waste Composition of Packaging MaterialTypes in Waste Stream

	Monmouth w/recyc 1987/88 %	Passaic w/o recy 1987 %	Union w/o rec 1987 %	Camden mixed 1986 %	Portland w/ recy 1987 %	Berkeley w/o recy 1989 %	Lamoille w/ recy 1989 %	Maryland 1990 %	Washington 1987 %
Paper	43.5	44.2	39.0	43.2	34.4	40.0	38.4	39.5	30.5
Corrugated	4.7	2.1	7.0	10.6	7.7	7.8	5.5	7.0	4.8
Paperboard									
Other	38.8	42.1	32.0	32.6	26.8	32.2	32.9	32.5	25.3
Aluminum	1.1	1.4	1.2		2.6	0.7			
AlumCans	0.5	0.8	1.0		2.0	0.5			
Other	0.5	0.6	0.3		0.6	0.2			
Ferrous	2.9	4.1			4.0	2.2			
Ferr Cans	1.6	3.1			2.1	1.7			
Other	1.3	1.0			2.0	0.5			
Plastic	11.6	7.3			5.8	5.0	5.5	8.1	8.0
PET	0.5				0.2	0.2	1.1	0.5	0.4
HDPE	2.4	3.3			0.5	0.4	2.8	0.6	0.5
Films						2.9			
Rigid Cont					1.0				
Other	8.7	4.0			4.4	1.6	1.6	7.0	7.1

Table 4 - Waste Composition of Packaging Material Types in Waste Stream

Notes:

Monmouth - These figures used as the basis for Camden numbers which are not presented here.

Passaic - (1) Other aluminum includes all non-ferrous

(2) PET included in 3.3% HDPE figure.

Union - (1) Average of summer and fall sorts.

(2) Other aluminum includes all non-ferrous.

Portland - (1) Figures from St. John's landfill were used.

(2) Other plastics includes a classification "durable" plastics

(3) The 600 TPD recycling level for HDPE is an estimate based upon 1988 figures.

(4) HDPE figure includes only plastic milk jugs, and not other HDPE containers.

(5) Aluminum and PET includes all beverage cans from deposit system, not all of residential origin.

Lamoille - Lamoille Solid Waste District, Vermont

Table 5 - New Jersey Waste Composition

	MONMOUTH COUNTY (1)		CAMDEN COUNTY (2)		PASSAIC COUNTY (3)		UNION COUNTY (4)		NEW JERSEY (5)	
Paper	43.77		46.44		45.08		40.90		45	
Corrugated		4.77		5.13		2.24		6.69		5
Newspaper		12.73		13.48		11.85		9.20		12
Paperboard										7
Other		26.26		27.84		30.99		25.00		21
Glass	6.37		7.01		6.73	0.00	3.34		7	
Glass Bottles		6.37		6.89		6.54		3.34		6
Other				0.13		0.19		0.00		0
Aluminum	1.06		1.13		0.77	0.00	1.00		1	
Aluminum Cans		0.53		0.52		0.77		1.00		1
Other		0.53		0.60		0.00		0.00		0
Ferrous	2.92		3.37		3.97	0.00	3.23		3	
Ferr Cans		1.59		1.92		3.02		3.23		2
Other		1.33		1.45		0.95		0.00		1
Non-Ferrous	0.27	0.27	0.27	0.27	0.57	0.57	0.23	0.23	0	0
Plastic	11.67		12.44		7.52		11.43		10	
HDPE		2.39		2.45		2.38		0.00		2
PET		0.53		0.63		0.48		0.00		1
Films										2

Table 5 - New Jersey Waste Composition

Rigid Cont										3
Other		8.75		9.36		4.67		11.43		3
Bulky					1.51	1.51		0.00	1	1
Organics	32.89		28.11		29.65	0.00	38.91		32	
Wood		1.86		1.86		0.95		2.17		2
Yard Waste		15.12		16.09		14.04		19.15		16
Food Waste				4.48		11.52		12.71		8
Misc. Organics		15.92		5.68		3.14		4.88		7
Misc.	1.14	1.14	1.22	1.22	4.19	4.19	0.97	0.97	1	1
TOTAL	100.08	100.08	100.00	100.00	100.00	100.00	100.00	100.00	100	100

(1) Monmouth County Resource Recovery Facility, Preliminary Environmental and Health Impacts Statement. HDR Engineering, July 1989.

(2) Waste Reduction, Recycling, and Composting for Camden County, New Jersey: A Common Sense Approach. Prepared by Self-Reliance, Inc. for the Camden County Board of Chosen Freeholders, January 1990.

(3) Passaic County Weighing and Composition Study. Prepared by Malcolm Pirnie for the Passaic County Utilities Authority, November 1987.

(4) Final Environmental and Health Impact Statement for the Union County Resource Recovery Project: Volume 1. Prepared by Malcolm Pirnie, February 1988.

(5) Tellus estimates based on the four county studies, and on data from Table 4, as explained in the text.

Packaging materials make up about 30% of the New Jersey waste stream. Paper products comprise about 13% to 15% of the waste stream, with corrugated comprising 5%, paperboard about 6.5% and other paper packaging about 1% to 3%. Glass, aluminum and ferrous containers comprise about 6.3%, 0.7% and 2.4% respectively. Plastic packaging makes up about 7.5% of the waste stream, with HDPE comprising about 2.4%, PET about 0.6%, films about 2% and other rigid containers about 2.5%.

Physical Characteristics of Packaging Materials

Physical characteristics of each packaging type have a large impact on the relative costs of collection programs and recycling and disposal facilities. Material density, Btu content and inert material (ash) content are the three factors with the largest impacts incorporated in this study. The values for these three characteristics for each packaging material are listed in Table 6. Because many operations, such as collection and landfills, are dependent upon material volume and not weight, the density of the material will be the determining factor in the costs of these programs. This density changes as materials are compacted under different conditions, such as in a packer truck and in a landfill.

Table 6 - Packaging MaterialCharacteristics

	Density (pounds/cubicyard)			Btu Content	Ash
	As-Disposed	In-Packer	In-Landfill	(Btu/pound)	Content
Plastic	35	105	350	14,500	8%
Aluminum	60	180	325	50	99%
Ferrous	150	450	700	50	99%
Corrugated	150	450	750	5,900	5%
Glass	650	1,950	2,800	50	99%

The range of packaging material densities is very wide. The densest material is glass, which has a density of about 650 pounds per cubic yard (lb/CY) as-disposed (i.e. at the curb) but a density of about 2800 lb/CY in the landfill. At the opposite end of the spectrum, plastic are the lightest material with an as-disposed density of roughly 35 lb/CY and a density of approximately 350 lb/CY in the landfill. Aluminum is the other particularly light material with an as-disposed density of about 60 lb/CY. Ferrous containers, corrugated cardboard, paper paperboard and paper packaging all have roughly the same densities, ranging from 150 to 175 lb/CY as-disposed. Estimates for as-disposed density were based a number of sources, most of which are listed in Table 7. Because of the lack of real data on compacted material densities in packer trucks (most data listed under the "compacted" heading in Table 7 does not replicate packer conditions), a generic assumption that compacted density was three times as disposed density was used. For landfill densities, we used data from the report "Estimates of the Volume of MSW and

Selected Components in Trash Cans and Landfills" produced by Franklin Associates in conjunction with The Garbage Project. The list of various sources and values is in Table 7.

Btu content and inert material content both affect the costs of incineration facilities. Btu content is the amount of energy released when a material is burned. Combustion of materials with higher Btu content will release more energy; if the incinerator was not already at its maximum capacity[1], the additional combustion energy will generate more electricity and hence more revenues. Inert material content is the percentage of a material's weight which is non-combustible. When sent through an incinerator, this material becomes ash, requiring disposal and therefore increasing costs.

Packaging made of plastic or paper is much less costly to incinerate than other materials. Plastics have a very high Btu content (14,500 Btu/lb) and a relatively low ash content (8%). Paper products also have high Btu content, ranging from 5,900 to 6,300 Btu/lb and low ash content, ranging from 4% to 4.5%. The other materials have characteristics much less desirable for incineration. Glass, aluminum and ferrous containers all have a Btu content of 50 Btu/lb and an ash content of 99%. This information was gathered from a number of solid waste handbooks and studies on the physical characteristics of waste components.

[1] Additional combustion energy above the incinerator's design limit (measured, for example, in Btus per hour) will not generate additional electricity; in fact, if incoming waste exceeds the expected Btu content per ton, it may be necessary to reduce throughput (tons per hour) so that design limits on Btus per hour are not exceeded.

Table 7 - DENSITIES OF PACKAGING MATERIALS

		LOOSE		TRUCK		LANDFILL	
MATERIAL	SOURCE	DESCRIPTION	LBS/CY	DESCRIPTION	LBS/CY	DESCRIPTION	LBS/CY
Glass	Franklin	Trash can	654			Landfill	2816
	NRC	Whole bottles	600-1000	Semi-crushed	1000-1800	Mech. crushed	800-2700
	EPA	Pure refuse	1000	< 2 " cullet	2300	< 5/8 " cullet	4212
	Tchoban	As discarded	270-810				
		(Typical)	327			Components	
	Wilson	Unbroken	700	Broken	2000		
	RI	Uncompacted	515-600	Compacted	1-2000		
				12:1 vol. reduction			
	ResRec(RI)	As discarded	700				
	Wayne	Whole	1000	Slightly crushed	1800	Well crushed	2700
Steel cans	Franklin	Trash can	212			Landfill	577
	NRC	Whole	150	Flattened	850		
	EPA			Flattened	850		
	Tchoban	As discarded	81-270				
		(typical)	149				
	Wilson	As discarded	160				
	RI	Uncompacted	145	Compacted	405-485		
				3:1 vol. reduction			

Table 7 - DENSITIES OF PACKAGING MATERIALS

		LOOSE		TRUCK		LANDFILL	
MATERIAL	SOURCE	DESCRIPTION	LBS/CY	DESCRIPTION	LBS/CY	DESCRIPTION	LBS/CY
	Mecklin	Loose-mixed	32				
	Wayne	Whole	150	Flattened	800-900		
	Jersey	Whole	150	Flattened	850		
	Bergen		160				
Aluminum	Franklin	Trash can	53			Landfill	317
	NRC	Whole	74	Flattened	250		
	EPA			Flattened	250		
	RI	Uncompacted	50-70	Compacted	430		
						6-8:1 vol. reduction	
	ResRec(RI)	As discarded	50				
	Wayne	Whole	74	Flattened	250		
	Mecklin	1 can = 0.53 oz.					
	Jersey	Whole	74	Flattened	250		
	Bergen		60				
Paper	Franklin	Trash can	48			Landfill	740
	EPA			Baled-shredded	750		
	Tchoban	As discarded	54-216				
		(typical)	138				

Table 7 - DENSITIES OF PACKAGING MATERIALS

MATERIAL	SOURCE	LOOSE DESCRIPTION	LOOSE LBS/CY	TRUCK DESCRIPTION	TRUCK LBS/CY	LANDFILL DESCRIPTION	LANDFILL LBS/CY
	Schwarz	As discarded-brown	135				
Corrugated	Franklin	Trash can	43			Landfill	750
	NRC	Loose	300	Baled	1000-1200		
	EPA	Pure refuse	1161				
		Components					
	Tchoban	As discarded	83				
	Schwarz	As discarded	180				
	RI	Uncompacted	40-45	Compacted	405-475		
				7:1 vol. reduction			
Mixed	Franklin	Trash can	53			Landfill	355
Plastics	Tchoban	As discarded	54-216				
		(typical)	108				
	Wilson	Loose	50				
	RI	Uncompacted	20	Compacted	700		
				16:1 vol. reduction			
	ResRec(RI)	Loose	32				
		Probably HDPE/PET					

Table 7 - DENSITIES OF PACKAGING MATERIALS

		LOOSE		TRUCK		LANDFILL	
MATERIAL	SOURCE	DESCRIPTION	LBS/CY	DESCRIPTION	LBS/CY	DESCRIPTION	LBS/CY
PET	Rutgers	As discarded	34-40	Compacter truck	42-49	Baled	460
	NRC	Loose	40-43			Granulated	506.3-472.5
	RI	Uncompacted	30-40	Compacted 16:1 vol. reduction	515		
	RI paper	Loose (PET/HDPE)	35				
	Bergen		30				
	Mechlenburg		31				
HDPE	Rutgers	As discarded	22-24				
	NRC	Loose	24			Granulated	540-675
	RI	Uncompacted	25	Compacted 16:1 vol. reduction	270		
	Wayne					Ground	621
						Baled	550
	Bergen		25				
Film	Franklin(89)	As discarded	24	Compacter truck	250		
	NRC					Baled	849
	Schwarz	Disc. polyethylene	20				

Table 7 - DENSITIES OF PACKAGING MATERIALS

		LOOSE		TRUCK		LANDFILL	
MATERIAL	SOURCE	DESCRIPTION	LBS/CY	DESCRIPTION	LBS/CY	DESCRIPTION	LBS/CY
Polystyrene	Schwarz	As discarded	175				
Polypropylene	Schwarz	As discarded	100				
Fast food packaging	Rathje					Landfill	261
Diapers	Rathje					Landfill	308

SOURCE DESCRIPTIONS

Franklin	Franklin Associate's "Estimates of the Volume of MSW and Selected components in Trash Cans and Landfills", Final Report, February 1990
NRC	National Recycling Coalition, "Weight to Volume Conversion Factors for Recyclables", May 1989
EPA	
Tchoban	George Tchobanoglous, et al., "Solid Wastes: Engineering Principles and Management Issues, 1977
Wilson	David Gordon Wilson, ed., Handbook of Solid Waste Management, 1977
RI	Rhode Island's "Guide for Commercial Solid Waste", Appendix C
ResRec(RI)	
Wayne	Wayne Engineering Corporation, "Curbside Collection of Recyclable Waste Materials"
Mecklin	
Jersey	NJ DEP's "Steps in Organizing a Municipal Recycling Program", 1986
Bergen	Bergen County Apartment Recycling Manual
Schwarz	Stephen C. Schwarz, "Energy and Resource Recovery from Waste", 1983
Rutgers	S. Rankin, "Recycling Plastics in Municipal Solid Wastes, CPRC at Rutgers
Mechlenburg	Mecklenburg County Recycling Evaluation Report, June 1988, RIS
Franklin (89)	V. R. Sellers, "A Case Study Analyzing the Volume of Residential Plastics", February 1989
Rathje	Various publications, numbers sited in Franklin

COLLECTION PROGRAMS

Dropoff Recycling Collection

Data was collected from many New Jersey counties on the number of dropoff centers available for residents. The actual costs of these dropoff programs were not investigated; instead a set of basic assumptions were made uniformly across the seven scenarios. These assumptions served as the sole basis for determining dropoff costs.

The dropoff program assumes unattended satellite dropoff centers with a number of different containers for depositing materials. These containers are periodically collected and delivered to the central recycling facility where separation and processing occur. The assumptions used in estimating the costs of this program are presented below.

Container	2 cubic yard dumpster
Number of sites	5
Containers per site	3
Average distance to facility	17 miles
Transportation cost	$.20 per ton per mile

Curbside Recycling Collection

While the requirement of mandatory recycling has pushed most communities to begin curbside recycling programs, there is great variety in the types of programs that exist. In the earliest programs, materials were generally collected through multiple-separation programs where each material was set out in a different container. However, commingled collection has become increasingly popular with time due to the advantages of increased household convenience, increased collection efficiency and the economies of scale in large materials recovery facilities. Many counties leave responsibility for collection to the municipalities which have traditionally provided garbage collection, though there are two counties, Atlantic and Somerset, where collection is provided by the county government.

Data was collected from the two county programs and numerous municipal programs on their collection operations and costs. Basic information on these programs, such as collection frequency, crew size, and types of trucks, is provided in Table 8. A few characteristics of these programs are of particular interest in characterizing New Jersey collection programs. The frequency of collection varies widely from once a week to once a month, though the average seems to be roughly once every other week. Moreover, there are a number of local variations in how particular materials are collected. For example, in Sayreville, all materials are collected weekly, though there are two separate set outs (one for plastic, one for all other materials) and actual collection is made by three separate vehicles -- one for newspaper and corrugated cardboard, one for plastics and another for recyclable containers (glass, metal, aluminum).

A wide variety of truck types and crew sizes are used. Many municipalities use specialized recycling trucks, such as the Eager Beaver truck or trailer (towed by a pickup truck) which is particularly popular in New Jersey. However, many communities have chosen to use existing vehicles such as dump trucks for collection. Crew sizes vary from 1 to 3, with most communities choosing 2 person collection crews.

Some communities collect commingled materials; others require separation into multiple containers. Many communities are moving to commingled collection as more counties develop facilities to accept mixed materials. In most municipalities where collection is multiple separated, there is neither a county facility nor a nearby private facility for materials to be delivered.

Table 8

MUNICIPALITY	COUNTY	POP SERVED	H-HOLDS SERVED	TRUCK INFORMATION NUMBER	TRUCK INFORMATION TYPE	CREW SIZE	PICK UPS PER MONTH	H-HOLD SET-OUT	CONTAINERS	MATERIALS
SAYREVILLE	MIDDLESEX	34,892	10,000	9		3	4	COM	(2) 5 GAL	N,G,A,T,PET, HDPE
BERLIN TWP	CAMDEN	5,576	1,800	3		1	4	COM		N,G,A,P,CC
BERLIN BOR	CAMDEN	6,300	2,200	1	TRAILER	1	4	COM	5 GAL	G,A,N,CC,T
				1	STAKE BODY					
				3	DUMP					
UPPER TWNSHP	CAPE MAY	9,671	4,000	2	EAGER BEAVER	2	4	COM	NONE	N,G,A,PET,H DPE
TRENTON	MERCER	22,936	6,000	1	PICKUP	2	2			N,C,P,A,T,G
				3	DUMP					
				2	TRAILERS					
PARAMUS	BERGEN	26,198	1,800	3	COMP GAR	2	2	COM	65 GAL	N,G,A,T,P
EAST ORANGE	ESSEX	70,000	2,700	3			2			
NUTLEY	ESSEX	29,000	10,000	4	DUMP	2	4	SEP		N,G,A
				1	ROLL OFF					

Table 8

MUNICIPALITY	COUNTY	POP SERVED	H-HOLDS SERVED	TRUCK INFORMATION NUM-BER	TRUCK INFORMATION TYPE	CREW SIZE	PICK UPS PER MONTH	H-HOLD SET-OUT	CONTAINERS	MATERIALS
ABSECON	ATLANTIC	8,000	3,000				2			G, ??
PT. PLEASANT	OCEAN	19,000	7,500				2			G,A,P,PLASTI C
EVESHAM	BURLINGTON	30,000	8,500	3	PACKERS	2	4	COM		G,T,A,PLAST IC
				3	15 CY TRAIL					
HADDONFIELD	CAMDEN	12,500	4,400	2	15 CY TRAIL	1	4	COM	YES	N,G,T,A
DOVER	OCEAN	85,000	31,500	14	EAGER BEAVER	2	4	COM	NONE	N,G,A
GLENN RIDGE	ESSEX	7,000	2,400	2	COMPARTMENT ALIZED	3	4	COM		N,G,A,T
COUNTY	SOMERSET	226,000	80,000	10	STEP VAN	4	2	COM	NONE	N,G,A,F,P
				22	SIDE LOADIN	3				
COUNTY	ATLANTIC	217,000	85,770	22	EAGER BEAVER	2	2	COM	NONE	N,G,A,P
				2	STAKE BODY					

Table 9 - Recycling Collection Program Characteristics

COUNTY	MATERIALS (1)	AVERAGE PICK UPS PER MONTH	PROGRAM TYPE COMMINGLED	SEPA-RATED	OPERATOR PUBLIC	PRIVATE
			%	%	%	%
Atlantic	N,G,A,HDPE,PET	2	100	0	100	0
Camden	N,G,A,F,HDPE,PET	3	100	0	40	60
Ocean	N,G,A,F,HDPE,PET	3	78	22	50	50
Somerset	N,G,A,F,HDPE,PET	2	100	0	100	0
Landfill	NA	2	43	57	67	33
Incinerator	NA	2	48	52	51	49
Transfer Station	NA	2	83	18	53	48

1 Estimate based on information from tonnage grants, interviews with county coordinators and lists of materials mandated for recycling.

Table 9 - Recycling Collection Program Characteristics

N = Newspaper

G = Glass

A = Aluminum

F = Ferrous and tin cans

PET = Polyethylene terephthalate

HDPE = High density polyethylene

Table 9 - Recycling Collection Program Characteristics

LAND-FILL COUN-TIES	MATERIALS	AVERAGE PICK UPS PER MONTH	PROGRAM TYPE COMMINGLED	SEPA-RATED	OPERA-TOR MUNICIPAL	PRIVATE
			%	%	%	%
Burlington	N,G,A			100	100	
Cape May	N,G,A,F,PET	1	100		50	50
Cumberland	N,G,A,PET					
Monmouth	N,G,A,F	1	66	34	66	34
Salem	N,G,A,F,PET,HDPE			100	100	
Sussex	N,G,A,F,PET		50	50	20	80
Total			43	57	67	33
Average						

Table 9 - Recycling Collection Program Characteristics

INCINERATOR COUN-TIES	MATERIALS	AVERAGE PICK UPS PER MONTH	PROGRAM TYPE COMMINGLED %COM	SEPA-RATED %SEP	OPERATOR MUNICIPAL %MUN	PRIVATE %PRI
Bergen	N,G,A,F,PET	1	55	45	60	40
Essex	N,G,A,PET	1	50	50	40	60
Gloucester	N,G,A,F,PET	1	33	67	50	50
Hunterdon	N,G,A,F			100	100	
Passaic	N,G,A,PET		100		60	40
Union	N,G,A,PET	1	25	75	50	50
Warren	N,G,A,F,PET,HDPE	0.25	75	25		100
Total		1	48	52	51	49
Average						

Table 9 - Recycling Collection Program Characteristics

TRANSFER STATION COUNTIES	MATERIALS	AVERAGE PICK UPS PER MONTH	PROGRAM TYPE COMMINGLED	SEPARATED	OPERATOR MUNICIPAL	PRIVATE
			%COM	%SEP	%MUN	%PRI
Hudson	N,G,A,PET		90	10		100
Mercer	N,G,A,F,PET,HDPE	1	100		100	
Middlesex	N,G,A,F,PET	1	40	60	75	25
Morris	N,G,A,PET	0.25	100		35	65
Total			83	18	53	48

Complementing the detailed information on specific programs shown in Table 8, we collected information from each county describing the county's overall current system for collecting recyclables, the materials being collected, the frequency of collection and, when available, average crew sizes. This county-level information is summarized in Table 9. The availability of program cost information from programs was much more limited, however. These costs are listed separately in Table 10 for programs where it was available.

Table 10

Per ton Costs of Various New Jersey Collection Programs

	Cost per Ton ($)
Recycling Collection	
Atlantic County	75
Somerset County	67
Long Beach Township	50
Dover Township	45
Garbage Collection	
Abescon	48
Dover Township	57
East Orange	33
Boro of Point Pleasant	60

From this information, a set of recycling collection inputs to WastePlan were developed for each of the seven scenarios; the data closely match the kind of collection systems used in those New Jersey regions. WastePlan then uses this information to calculate truck requirements and truck costs; this accounts for the majority of collection costs. We included an estimate of costs for administration, though these costs may not reflect the actual administration costs (or profit, in the case of private haulers), as this information was difficult to obtain.

The values used for some data items, particularly the number of households serviced per hour, were significant in matching WastePlan outputs to available actual cost information. The assumptions used for the truck costs are described below. Specific information was available for Somerset and Atlantic Counties, so this information was used.

In other counties, a standard type compartmentalized recycling truck was used assumed; it is the same truck which is actually used in Atlantic County.

Truck Information

	Single-Family		
	Somerset County	All other counties	Multi-Family
Type	Side Loading Vehicle	Compartmentalized Truck (Eager Beaver type)	Packer
Capacity	20 cubic yards	23 cubic yards	20 cubic yards
MPG	4.5	5	3
Lifetime	5	7	7
Price	$47,000	$47,000	$100,000
Avg. Salary	$22,000	$21,000	$22,000
Maintenance	$.11/ mile	$.16/mile	$.70/mile
Dump Time	20 minutes	20 minutes	15 minutes
License/ Insurance	n/a	$1,600	$1,600
Fuel cost	$1 per gallon	$1 per gallon	$1 per gallon

Assumptions used in defining the operations of each programs are listed below. Of particular note are the number of households per hour used in each scenario. Very little information is available to determine the average number of stops made by a recycling truck in an hour or the number of households per stop. In order to make these calculations we looked at the type of collection program and operator. As shown below, our assumptions were that private operators were more efficient than municipal, and that commingled collection is more efficient than separated.

Households per Hour

Collector	Efficiency	Commingled	Separated
Private	NA	90 HH per hour	65 HH per hour
Public	High	90	65
Public	Low	65	40

For each scenario, a calculation was made to determine the proportion of the population serviced by the various collection rates. In WastePlan, the three rates of 90, 65 and 40 households per hour were entered for the urban, suburban and rural populations in WastePlan.

	90 HH per hour	65 HH per hour	40 HH per hour	Crew Size	Pickups Per Week
Atlantic	100	0	0	2	0.5
Camden	80	20	0	1.5	0.75
Ocean	58	31	11	1.7	0.7
Somerset	100	0	0	2	0.5
Landfill	28	53	19	2	0.5
Incinerator	37.5	50	12.5	2	0.5
Transfer	60	35	5	2	0.5

Other assumptions common to all collection programs include:

Hours per day of collection:	6.5
Distance to Dropoff (miles):	20
Miles per Hour to Dropoff:	25
Administration Costs ($ per household):	$ .80

This method of modelling recycling collection has led to some discrepancy with the actual costs of programs in New Jersey. For example, the costs of Atlantic and Somerset's programs are actually about $75 and $67 per ton respectively, while the model produces costs of $59 and $44 per ton for these programs. This can be attributed to several causes. First, the actual collection costs from each county are only for single-family curbside collection, whereas the WastePlan per ton costs include multi-family collection and dropoff collection. Though the amount of material collected in these programs is significantly smaller than the single-family collection, they still produce a small decrease in the average cost of collecting recyclables from all programs. Second, WastePlan assumes the use of only one type of truck, whereas in both the Atlantic and Somerset programs, older, less efficient trucks are still in use, though slowly being phased out. These less efficient trucks will increase the actual costs of collection in comparison to the newer trucks. Third, the household per hour estimates, though based upon data from Atlantic, Somerset and other municipal programs, are generalized across all counties, so some approximations of actual costs result.

Garbage Collection

The availability of information on garbage collection was much more limited than recycling collection, primarily because of the need to rely on responses to the Tellus survey as the sole sources of data. This survey provided some information on the types of service provided (municipal versus private) and the frequency of collection but little on the more specific details of program operations and actual costs. Actual costs of municipal programs are listed in Table 11.

Table 11

MUNICIPALITY	COUNTY	OPERATOR	VEHICLE TYPE	CAPACITY (CU YDS)	QUANTITY	CREW SIZE	PICKUPS WEEK
DOVER	OCEAN	MUNICIPAL	REAR LOADER	25	25	3	2
			REAR LOADER	31	3	3	
PT. PLEASANT	OCEAN	PRIVATE					
HAMMONTON	ATLANTIC	MUNICIPAL	COMPACTOR	25		3	1
			COMPACTOR	32		3	
			COMPACTOR	32		3	
BERLIN BOR	CAMDEN	MUNICIPAL	PACKER	25		3	1
			PACKER	20		3	
NUTLEY	ESSEX	PRIVATE					3
EAST ORANGE	ESSEX	PRIVATE					3
VARIOUS TOWNS	SOMERSET	PRIVATE					1

Table 12 - New Jersey Recycling Facilities

	Capacity (TPD)	Capital Cost ($1,000)	Annual Operating Cost ($1,000)	Tip Fee ($ per ton)
Public				
Atlantic	40	2300	1400	
Camden *	65	700	1200	
Cape May	225	5346	1236	
Ocean	200	6747	1153	
Somerset (current)	125	1000	2350	
(future)	300	4900		
Private **				
Monmouth Recycling Corp	43	1500	275	0
Monmouth Processing	25	120	90	-5
West Patterson Recycling	70	1100		0
Rosetta (Ocean and Monmouth)	250	6600		
REI (Newark)	250	900		-12

* Camden County capital costs are low because the facility is leased, and because it does not process paper.

** Private facilities do not take newspaper or mixed paper.

Similar to recycling collection, garbage collection costs are calculated by determining truck requirements and truck costs based upon a set of input assumptions. These assumptions are based upon available information from New Jersey and is supplemented by other national sources where this information was not sufficiently complete. A summary of the assumptions is provided below.

Truck Information (for all counties)

	Single-Family	Multi-Family
Type	Rear Packer	Front-Loading Packer
Capacity	20 cubic yards	31 cubic yards
MPG	3	3
Lifetime (years)	7	7
Compaction rate	3:1	3:1
Price	$100,000	
Maintenance	$.80 per mile	
Labor costs	$22,000 per crew	$22,000 per crew
Dump Time	15 minutes	15 minutes
License/Insurance	$1,600	$1,600
Fuel Cost	$1/gallon	$1/per gallon

Collection Information	Single-family	Multi-family
Households per hour	60 - 50 - 40	300 - 300 - 300
Miles per hour to drop	25	same as single-family for remainder
Miles to drop	20	
Collection days per week	5	
Weeks per year	52	
Pickups per week	1	

Administrative cost per HH $.80

RECYCLING FACILITIES

Materials collected through dropoff and curbside collection are all delivered to recycling processing facilities where they are separated (if necessary), processed and marketed. Facilities run by private operators dominated the first wave of recycling facilities, though several counties ran their own central processing facilities from the beginning. Recently, a number of counties (Cape May, Ocean) without facilities have been developing their own, while many of first generation public facilities (Atlantic, Somerset) are being replaced with more highly automated facilities, incorporating many advances in recyclables processing technology. Private facilities are continuing to update their facilities as well, and some counties and municipalities will continue to rely upon private operators for handling of recyclables.

Materials can be delivered to facilities in a variety of forms, usually depending upon the particular technology used; many private operators will accept materials in a variety of forms. Most public facilities accept two streams, with commingled containers separated from newspaper (the latter is often mixed with corrugated cardboard and mixed paper). Most of the private facilities accept only the mixed containers stream, so newspaper must be sold separately to a broker.

Information on types of facilities and their costs was collected from individual counties and through a number of independent sources. In Table 12, basic cost and capacity information are listed for a number of New Jersey facilities. This information was used to develop per ton costs for processing of recyclables in each of the seven New Jersey scenarios. Because of incomplete information on materials revenues, we used available information from currently operating New Jersey facilities to develop a single set of assumptions. The per-ton net costs used in each of the scenarios is listed in Table 13. These per ton costs are not used in later estimations of the marginal costs for individual materials. The methodology for producing these estimates is presented later in Section IV.

Table 13 - Recycling Facility Costs

	Cost per ton
Atlantic	46
Camden	6
Ocean	16
Somerset	20
Incinerator counties	15
Landfill counties	15
Transfer counties	20

DISPOSAL FACILITIES

New Jersey relies upon either incinerators (with ash going to landfills), landfills or long-distance transfer for the disposal of its waste. Currently, landfills are the most prevalent method for disposal of waste, though with a number of new facilities under construction or in advanced development, incineration may soon rival the disposal capacity of landfills. A large amount of waste is transferred out of state, though with regional and national sentiment growing against out-of-state export of waste, a large growth in the amount of waste leaving the state is unlikely.

A summary of the type, costs and future plans for waste disposal in each New Jersey County is presented in Table 14. This shows only 3 counties (Warren, Hunterdon and Gloucester) currently using incineration, 9 using transfer stations and 10 using landfills as primary disposal sources (Camden currently uses both transfer and landfill).

Table 14 - Garbage Collection Programs in New Jersey

COUNTY	NAME	FACILITY TYPE	TIP FEE (1)	CAPACITY CU YARDS (000)	YEARS	COUNTY FUTURE PLAN
ATLANTIC	AC UTILITY AUTHORITY	TS		68	0	Continue to use new transfer station to ship waste out of state.
BERGEN	BC UTILITY AUTHORITY	TS	$137.90			Proposed resource recovery facility in county.
BURLINGTON	BURLINGTON COUNTY	L	$44.75	4,654	7	Continue to use landfill and potentially expand it.
CAMDEN FORGE	(PHILADELPHIA)	TS				Increase use of resource recovery. New facility will likely be built in Camden County.
	PENNSAUKEN	L		3,900	13	
CAPE MAY	CM UTILITY AUTHORITY	L	$89.00	150	3	Landfill mandated to close at end of 1992. Options unexplored for thereafter.
CUMBERLAND	CUMBERLAND COUNTY	L	$52.11	5,732	19	Rely on landfill.
ESSEX	(NEWARK)	TS (2)	$102.15			Since completion of analysis, a resource recovery facility has begun operations. Current tip fee is $65 per ton.
GLOUCESTER	"SES" (W. DEPTFORD)	I	$86.00			Use resource recovery more and use landfill as ashfill.
	GC UTILITY AUTHORITY	L	$58.92	2,317	17	
HUDSON	HMDC	L	$28.12	1,000	1	Probably transfer waste to a neighboring county's resource recovery facility or out of state.
HUNTERDON	WARREN COUNTY	I	$98.00			Continue to use the Warren incinerator.
MERCER	MC IMPROVMENT AUTHORITY	TS	$92.26			Continue to transfer waste out of state.
MIDDLESEX	EDGEBORO	L	$54.42	1,100	1	Edgeboro Phase I will close, though potential for expansion in adjacent Phase II site. Otherwise, transfer waste out of state or to another county's incinerator.
	EDISON	L	$40.00			
MONMOUTH	MCRC	L	$68.25	2,742	2	Expand landfill.
MORRIS	MC TRANSFER STATION	TS	$117.60			Transfer waste out of state.
OCEAN	OCEAN CTY LANDFILL	L	$73.40	6,716	7	Continue to use Ocean County landfill.
PASSAIC	PC UTILITY AUTHORITY	TS (2)	$89.71			Resource recovery facility likely to be built in county.
SALEM	SC UTILITY AUTHORITY	L	$59.44	2,400	18	Continue to use landfill.
SOMERSET	BRIDGEWATER RESOURCES	TS	$113.95			Continue transfer to Warren incinerator and out of state.

COUNTY	NAME	FACILITY TYPE	TIP FEE (1)	CAPACITY CU YARDS (000)	YEARS	COUNTY FUTURE PLAN
SUSSEX	SC UTILITY AUTHORITY	L	$110.00	3,407	24	Continue to use landfill.
UNION	UC UTILITY AUTHORITY	TS	$132.65			Resource recovery facility to be built in county.
WARREN	WARREN COUNTY	I	$98.00			Continue to use incinerator in county.

(1) Rates for disposal of ID 10 waste including state taxes and host community benefit surcharges from New Jersey Board of Public Utilities, April 3, 1990 (Mercer and Sussex County data updated by county personnel).

(2) Remaining capacity data from New Jersey Department of Environmental Protection, Division of Solid Waste from "Operating Landfills of Regional Significance Status and Capacity, April 1990" Report by Div of SW (Except for Cape May County which received an extension, noted in years column)

However, once new incinerators begin to come on-line this picture will begin to change somewhat. When the 6 additional incinerators we have assumed will come on-line begin to operate, a total of 8 counties will rely of incineration for disposal. The number of counties using landfills is expected to drop to 7 and the number using transfer stations to 6. A list of these counties and the tip fee assumptions used in the scenario analysis can be seen from Table 2.

Incinerators

As noted earlier, there are actually only two incinerators operating at present within New Jersey. The Warren incinerator has operated since 1988 and has a current tip fee of $98 per ton, including ash disposal. The Gloucester facility opened in early 1990 and has a tip fee of about $86 per ton. These costs and all tip fees reported in this section includes costs for ash disposal. Two other facilities -- Camden and Essex -- are under construction and should be open by 1991. Anticipated tip fees at these facilities are $89 per ton at Camden and $66 per ton at Essex, though these estimates are based upon early cost projections and are likely to change by the time facilities open. The other facilities which are likely to be constructed are in Bergen, Passaic and Union Counties and the township of Pennsauken (Camden County).

Capital costs and operating costs for these facilities are presented in Table 15, along with actual or anticipated tip fees. This information was obtained primarily from the Board of Public Utilities, which regulates the vendor-county agreements. Many of the costs cited here represent preliminary agreements between vendors and counties and may change before a final agreement is signed.

Table 15 - New Jersey Incinerators

	Capacity	Capital Total	Cost per ton of capacity (2)	Operating Total	Cost $ / Ton	Tip Fee $ / Ton
	(Tons)	($000)	($000)	(000)	(000)	
Camden	1050	$128,150	$122	$8,912	$8	$89
Pennsauken (3)	500	$62,000	$124	$5,385	$11	$103
Bergen(3)	3000	$253,200	$84			
Essex	2250	$343,000	$152	$10,126	$5	$66
Gloucester	575	$83,750	$146	$6,400	$11	
Passaic(4) (5)	1300	$115,099	$89	$8,900	$7	
Union (6)	1440	$107,516	$75			
Warren(7)	440	$53,216	$121			$98
Totals	10555	$1,145,931		$39,723		
Weighted Average (1)			$108,568		$7,000	$79

(1) Weighted averages based on capacity.

(2) Per ton figures based on daily capacity, as opposed to daily design throughput.

(3) Project has been canceled since the completion of the analysis.

(4) Project completion may be influenced by the findings of the task force report.

(4) Since completion of the analysis, capital cost has been reestimated at $186 millions, which may escalate based upon the vendor service agreement.

(5) Capital cost reestimated at $146 million, which may escalate per vendor service agreement.

(6) Capital cost reestimated at $62 million, which may escalate per vendor service agreement.

Landfills

The costs of landfills varies from a low of $22 per ton at the Galloway Municipal landfill to a high of $110 per ton at the Sussex County landfill. Most of the landfills, with the exception of those owned by county Utility Authorities, are regulated by the state Board of Public Utilities, so tip fees reflect actual costs plus a reasonable rate of return if the facility is privately operated. A list of the tip fees from most of the major landfills within the state is provided in Table 16. Where available, information on the remaining capacity is provided in cubic yards and in years of remaining lifetime.

Table 16 - Landfill Costs and Capacities

		REMAINING CAPACITY (2)	
NAME OF LANDFILL	TIP FEE (1)	CUBIC YARDS	YEARS AT CURRENT LOADING
	($/TON)	(000)	
Burlington County	$44.75	4,654	7
Cape May Utility Authority	$89.00	150	3
Cumberland County Improvement Authority	$52.11	5,732	19
Monmouth County Reclamation Center	$68.25	2,742	2
Salem County Utility Authority	$59.44	2,400	18
Sussex County Utilities Authority	$116.25	3,407	24

TOTAL		19,085	

Table 16 - Landfill Costs and Capacities

Other Currently Operating Landfills

Galloway Municipal Landfill	$22.07
Hammonton Municipal Landfill	$24.78
Hackensack Municipal District Corporation	$28.12
Winslow Municipal Landfill	$49.78
Middlesex County Utilities Authority	$54.42
Gloucester County Improvement Authority	$58.92
Pennsauken Solid Waste Management Authority	$54.84
Ocean County Landfill Corp.	$73.40

Transfer Stations

Current transfer station costs in New Jersey are relatively high because most waste is shipped out of state. The costs of various transfer stations ranges from $90 per ton in Passaic to $133 per ton in Union County (a privately owned facility, receiving mostly commercial waste is actually higher at $136 per ton). The tip fees paid at various transfer stations are listed in Table 17.

Transfer costs will likely decrease when more in-state disposal capacity becomes available and the distance waste is shipped decreases. Transfer stations will then become primarily short-haul facilities moving waste in-state rather than long haul facilities sending waste out of state.

Table 17 - New Jersey Transfer Station Tip Fees

Operator	Cost per Ton ($)
Bergen C. U.A. [1]	137.90
Essex County	102.15
Hunterdon C.U.A. [1]	125.75
Mercer C.I.A.	92.26
Morris County Transfer Station, Inc.	117.60
Passaic C.U.A.	89.71
Somerset Intermediate Recycling	123.00
Bridgewater Resources (Somerset)	113.95
Union C.U.A.	132.65
Ellesor (Union)	136.35

C.U.A. - County Utility Authority

C.I.U. - County Improvement Authority

[1] Not regulated by Board of Public Utilities

IV. MARGINAL COST METHODOLOGY

The purpose of our solid waste system modelling is to provide a basis for determining the marginal costs of collecting, processing and disposing of different types of packaging materials. These marginal costs can then be used to estimate the incremental cost to the solid waste system associated with increasing or decreasing the amount of different packaging materials handled in the residential waste stream. This section will describe the methodology used for developing the marginal cost calculations.

In this analysis, eight different packaging categories are considered: aluminum, glass, paper packaging (paperboard), corrugated cardboard, ferrous containers, non-recyclable plastic containers, HDPE containers and PET containers. System marginal costs are developed for each of these categories, along with marginal costs for individual collection programs and facilities within the solid waste system. The categories represent the range of packaging types within the residential waste stream. Several categories are omnibus groupings for a broad range of packaging types. Non-recyclable plastic packaging encompasses PVC, Polystyrene and other containers as well as film. Paper packaging is primarily paperboard but also includes other packaging paper; such as flexible paper, which is not currently recycled. Corrugated cardboard recycling costs reported here would also apply, with little change, to the small quantity of kraft paper bags which are currently recycled.

What is a Marginal Cost?

By marginal cost, we mean the increased cost to the solid waste system of handling an additional amount of waste. It is calculated by comparing the costs of the solid waste system with the current amount of waste (the scenario results described in the previous section) to the costs of the same system with an additional increment of waste. In this report we express marginal costs as costs per ton or costs per cubic yard. In a later report we will convert them to costs per package for selected products.

The marginal cost of packaging disposal is an important measure for use in policy-making. Public policies affecting packaging use will likely lead to incremental changes in packaging quantities. Policymakers need to know the resulting incremental, or marginal, changes in social costs -- costs which may be substantially greater than the market price of the materials. In our report on production process models, we have estimated the marginal environmental impacts of producing a unit of each major packaging material. In this report, we estimate the marginal costs of getting rid of each material.

Average costs are easier to compute, and are often used in lieu of marginal costs, but this practice can lead to inaccurate conclusions. Average costs include the costs of all waste handled in the system, which may differ from the cost of handling the next ton of waste. For instance, new disposal capacity is frequently more expensive than old capacity; in such cases, marginal cost will be higher than average cost. On the other hand, for curbside collection programs average cost includes the high cost of driving to and stopping at each household. Marginal cost may be much lower, since the impact of additional material set out at each stop is only a slight increase in at-curb pickup time, and an increase in the number of trips to the recycling facility or disposal site.

While marginal costs are more appropriate for our purposes than average costs, limited data availability has at times forced us to use average costs. This is particularly true for facilities; in a number of cases, we know the total cost and tonnage handled by a facility, but do not know precisely how costs vary with increments of tonnage. In addition, we cannot be certain about the mix of facilities to be built in the future. Decisions about the future role of incineration, and other policy questions as well, will affect the types and costs of future solid waste facilities.

It is important to distinguish between short-run and long-run marginal costs. Short-run marginal costs measure the immediate changes that would be required to handle an additional amount of material, using the existing stock of facilities and equipment. These costs could include increased labor costs, utility costs, contract services, equipment rental or other short-term remedies. However, for solid waste management or environmental planning, the long-run marginal cost is a more appropriate number to consider. Long-run marginal cost is the cost of upgrading the system to be able to sustain the increased materials indefinitely; it may include facility and equipment costs, and short-run operating costs.

For our modelling process, this distinction affects the size of the increment of materials added to the waste stream. We want to see the long-run cost impacts of additional material, including the proportionate share of new facility and equipment requirements. So the increment of material to be analyzed must be large enough that it forces new investments in the system, such as the construction of additional processing or disposal capacity and the purchase of additional trucks. (If, for example, a new truck is required for every 2000 tons per year of additional material, then the marginal cost per ton should include 1/2000 of the annual payments on a new truck. An analysis of the costs of handling an incremental 20 tons would likely miss this effect; an analysis of 20,000 incremental tons would not.)

With a more refined understanding of solid waste system dynamics over time, it might be possible to identify the timing of future changes resulting from an increment in waste quantity. Then the long-run marginal cost would be the discounted present value of the anticipated changes. However, the available solid waste system cost and planning information is not sufficient to support that type of dynamic analysis at present. Our method, instead, is what economists call "comparative statics." In effect, we have hurled a large assumed incremental waste quantity at the system, and then recalculated the optimum system size and cost (changing size, but not choice of technology) for the increased quantity.

Modelling Marginal Costs

Using the New Jersey solid waste system model, we calculated marginal costs for each of the materials being studied. The process is relatively straightforward. For each packaging material in turn, we changed the composition of the waste stream by adding a specific increment of the material to the seven New Jersey scenarios, holding all other waste quantities constant. We ran the collection and facilities modules with the new material included in the waste stream; the results can be compared to the New Jersey base case scenarios to compute the statewide system cost increase for handling the additional material. We then divided the cost increase by the number of tons added to the system, yielding the marginal cost per ton. Using standard estimates of curbside (uncompacted) densities of packaging materials, we converted costs per ton into costs per cubic yard.

We decided to use the same increment in the analysis of each material; we selected it to be big enough to cause a change in equipment needs for all major solid waste management programs. The program requiring the largest increment was garbage collection. Since collection costs depend upon volume rather than weight, we used a standard volume increment for each material.

Our incremental volume was quite large, about 15% of the entire waste stream. Thus the incremental-quantity scenarios for each material are not meant to be probable futures. There is little likelihood of a sudden surge in aluminum cans, glass bottles, or any other single packaging material, amounting to 15% of total residential waste volume, while all other waste quantities remain unchanged. However, for the calculation of marginal costs, it is necessary to assume this improbable increase in a single material. The WastePlan model can help in identifying the system costs which would be incurred, using current technology and cost structures, for this unlikely circumstance in which the incremental-quantity scenarios do occur.

The remainder of this section will describe how the marginal costs are calculated for each of the solid waste operations performed in New Jersey. In some cases this is simply using the results of the WastePlan runs, while in others, calculation made outside of WastePlan support the WastePlan analysis.

Recycling and Garbage Collection

The WastePlan collection module is used to model the marginal costs of the recycling and garbage collection programs. When the additional material is added to the waste stream, WastePlan recalculates the number of trucks that are needed to collect all recyclables or garbage, along with the resulting cost increases.

All of the collection data inputs remain the same as in the base case, with the exception of the collection efficiency (number of households collected per hour). Collection efficiency is decreased to account for the slower collection caused by the higher volume set out at the curb. For each New Jersey scenario and each packaging material, the increase in material volume was calculated by WastePlan, and a proportional adjustment was made to the collection efficiency. The adjustment was based primarily on volume because recycling setout containers and garbage cans fill up by volume; the amount of time to collect materials at each residence is primarily a function of the average number of containers set out weekly. Small adjustments were made based upon weight as well, since the weight of a container will also affect the speed at which it can be loaded onto a truck. For example, a container full of glass bottles will take longer to lift and dump than a container of the same size full of plastic bottles.

Recycling Facilities

To calculate the marginal cost for each packaging material at recycling facilities, we estimated the costs of processing each material under each of three different technologies. The three technologies correspond to three types of recycling facilities found in New Jersey: an IPF, or facility with automated material separation and large capacity (150-225 tons per day); a recycling depot, or facility with primarily manual sorting and intermediate capacity (100-150 TPD); and a smaller facility accepting multiple separated materials which only performs basic processing and has lower capacity (40-100 TPD).

The marginal cost for each material at each of these three facilities is listed in Table 18. The costs shown in the table are net costs, i.e. processing costs minus revenues. (Negative numbers mean net benefits, i.e. revenues exceed costs.) The costs of the larger, highly automated facility are the lowest across all materials and the costs of the two other facilities are virtually identical. For each of the seven scenarios, the marginal costs of one or more of these facilities are used, depending upon the type of facilities used in the county or group of counties.

Table 18 - Recycling Facility Marginal Costs

	IPF	Depot	Separated
Plastic	-20	31	26
Aluminum	-730	-725	-753
Ferrous	3	27	18
Corrugated	8	31	27
Glass	-3	16	20
Percentage of County			
Incineration counties	33%	17%	50%
Landfill counties	25%	25%	50%
Transfer counties	50%	35%	15%

These marginal costs were developed by determining the capital and operating costs of the three different types of recycling facilities, and then assigning components of the facility's costs to different recyclables. The remainder of the costs not assigned to specific materials are allocated amongst them based upon the percentage breakdown of the assigned costs. A summary of each of the three facilities is provided in Table 19 through 21. The tables outline the assumptions made for the financing terms, the material throughput and revenues, equipment and facility costs and facility operating costs. In addition, the apportionment amongst materials of specific, single cost items such as buildings, conveyors, paper conveyors, residues and supplies are further detailed.

Table 19 - IPF Financial and Throughput Assumptions

Financing Assumptions					
Interest	8.50%				
Equipment Lifetime	7				
Facility Lifetime	20				
Days Operating per Year		260			Annual
Daily Capacity (TPD)		225	Revenue	Density	Volume
Daily Throughput			($/ton)	(lb/CY)	(CY)
Plastio		12	140	35	178,286
Glass		50	45	650	40,000
Aluminum		5	900	60	43,333
Newspaper		124	5	550	117,236
Corrugated		17	30	150	58,933
Ferrous		17	40	150	58,933
Total		225			496,722
Cost Analysis					
Total Operating Cost ($)		1,271,120			
Annual Capital Cost ($)		658,817			
Total Annual Cost ($)		1,929,937			
cost per ton		33			
Annual Revenue ($)		2,423,174			
Net Annual Cost ($)		-493,237			
cost per ton		-8			

Table 19 - IPF Financial and Throughput Assumptions

II. Capital and Operating Costs

IPF Capital Costs

Cost Item	
Design, Engineering, etc.	700,000
Building and Site	2,900,000
Mobile Equipment	250,000
Separation Equipment	
Magnetio Separators	40,000
Air Knife/Separator	35,000
Glass Sort Conveyor	150,000
Paper Conveyor	75,000
Conveyors	100,000
Processing Equipment	
Glass Crushers	150,000
Ferrous Flattener/Shredder	70,000
Aluminum Blower	35,000
Horizontal Baler (paper)	250,000
Perforator/Baler (Plastic)	70,000
Conveyors	200,000
Total	5,025,000

Apportionment of "Building and Site" Costs	% of total building
Plastic	10.00%
Glass	15.00%
Aluminum	8.00%
Ferrous	8.00%
Newspaper	30.00%
Corrugated	5.00%

Apportionment of "Conveyor" Costs	% of total conveyors
Plastic	26%
Glass	31%
Aluminum	18%
Ferrous	8%
General (remainder)	17%

Apportionment of Paper Conveyor Costs	
Newspaper	80%
Corrugated	20%

Table 19 - IPF Financial and Throughput Assumptions

Operating Costs	
Utilities	80,000
Insurance	35,000
Maintenance	100,000
Supplies	85,000
Residue Disposal	285,870
$/ton	50
% residue	9.77%
Equipment Replacement (%)	5%
Cost	71,250

Residue Costs	Mater	Material
Plastic	14%	0.75%
Glass	9%	2.00%
Aluminum	7%	0.16%
Newpape r	10%	5.51%
Corrugated	10%	0.76%
Ferrous	8%	0.60%
General		
Total		

Apportionment of Supplies Costs	
Plastic	10%
Newspaper	20%
Corrugated	5%
General	65%

Labor	Number	Salary/ Employ	Total
Management	2.00	35,000	70,000
Plastics	4.00	23,000	92,000
Glass	4.50	23,000	103,500
Aluminum	1.50	23,000	34,500
Newspaper	2.50	23,000	57,500
Corrugated	1.50	23,000	34,500
Ferrous	1.50	23,000	34,500
Other	7.50	25,000	187,500
Total	25.00	24,560	614,000
			(average)

Table 20 - Recycling Depot Financial and Throughput Assumptions

Financing Assumption					
Interest	8.50%				
Equipment Lifetime	7				
Facility Lifetime	20				
Days Operating per Year		260			Annual
Daily Capacity (TPD)		112.50	Revenue	Density	Volume
Daily Throughput			($/ton)	(lb/CY)	(CY)
Plastic		6.00	135	35	89,143
Glass		25.00	40	650	20,000
Aluminum		2.50	900	60	21,667
Newspaper		62.00	0	550	58,618
Corrugated		8.50	25	150	29,467
Ferrous		8.50	35	150	29,467
Total		112.50			248,361
Cost Analysis					
Total Operating Costs ($)		1,042,435			
Annual Capital Cost ($)		404,757			
Total Annual Cost ($)		1,447,192			
cost per ton		49			
Annual Revenue ($)		1,188,200			
Net Annual Cost ($)		258,992			
cost per ton		9			

Recycling Depot

Table 20 - Recycling Depot Financial and Throughput Assumptions

II. Capital and Operating Costs		Apportionment of "Building and Site" Costs	% of total building
IPF Capital Costs		Plastic	12.50%
Cost Item		Glass	12.50%
Design, Engineering, etc.	200,000	Aluminum	8.00%
Building and Site	1,800,000	Ferrous	8.00%
Mobile Equipment	150,000	Newspaper	30.00%
Separation Equipment		Corrugated	5.00%
Magnetic Separators	40,000		
Air Knife/Separator	0	Apportionment of "Conveyors" Costs	% of total conveyors
Glass Sort Conveyor	100,000		
Paper Conveyor	25,000	Plastic	26%
Conveyors	50,000	Glass	31%
Processing Equipment		Aluminum	18%
Glass Crushers	90,000	Ferrous	8%
Ferrous Flattener/Shredder	40,000	General	17%
Aluminum Blower	25,000		
Horizontal Baler (paper)	250,000	Apportionment of Paper Conveyor Costs	
Perforator/Baler (Plastic)	70,000	Newspaper	80%
Conveyors	150,000	Corrugated	20%
Total	2,990,000		

Table 20 - Recycling Depot Financial and Throughput Assumptions

Operating Costs				Apportionment of Residue Costs	% of Specific Material	% of all Material
Utilities		70,000		Plastic	14%	0.75%
Insurance		35,000		Glass	9%	2.00%
Maintenance		90,000		Aluminum	7%	0.16%
Supplies		65,000		Newpaper	10%	5.51%
Residue Disposal		142,935		Corrugated	10%	0.76%
$/ton		50		Ferrous	8%	0.60%
% residue		10%		Total		
Equipment Replacement		49,500				
% of Equip Cost		5%		Apportionment of Supplies Costs		
				Plastic	10%	
		Salary/		Newspaper	20%	
Labor	Number	Employ	Total	Corrugated	5%	
Management	2.00	35,000	70,000	General	65%	
Plastics	3.50	23,000	80,500			
Glass	4.50	23,000	103,500			
Aluminum	1.50	23,000	34,500			
Newspaper	2.50	23,000	57,500			
Corrugated	1.50	23,000	34,500			
Ferrous	1.50	23,000	34,500			
Other	7.00	25,000	175,000			
Total	24.00	24,583	590,000			
		(average)				

Table 21 - Separated Recyclables Facility Financial and Throughput Assumptions

Financing Assumption	
Interest	8.50%
Equipment Lifetime	7
Facility Lifetime	20

		Revenue ($/ton)	Density (lb/CY)	Annual Volume (CY)
Days Operating per Year	260			
Daily Capacity (TPD)	90			
Daily Throughput				
Plastic	5.00	135	35	74,286
Glass	20.00	40	650	16,000
Aluminum	2.00	900	60	17,333
Newspaper	49.40	0	550	46,705
Corrugated	6.80	25	150	23,573
Ferrous	6.80	35	150	23,573
Total	90.00			201,471

Cost Analysis

Total Operating Costs ($)	854,315
Annual Capital Cost ($)	306,809
Total Annual Cost ($)	1.161,124
cost per ton	50
Annual Revenue ($)	957,580
Net Annual Cost ($)	203,544
cost per ton	9

Separated Material Recycling Facility

II. Capital and Operating Costs

IPF Capital Costs

Cost Item	
Design, Engineering, etc.	200,000
Building and Site	1,400,000
Mobile Equipment	150,000
Separation Equipment	
Magnetic Separators	20,000
Air Knife/Separator	0
Glass Sort Conveyor	75,000
Paper Conveyor	0
Conveyors	50,000
Processing Equipment	
Glass Crushers	75,000
Ferrous Flattener/Shredder	25,000
Aluminum Blower	20,000
Horizontal Baler (paper)	200,000
Perforator/Baler (Plastic)	30,000
Conveyors	60,000
Total	2,305,000

Apportionment of "Building and Site" Costs	% of total building
Plastic	13.50%
Glass	11.00%
Aluminum	6.50%
Ferrous	6.50%
Newspaper	30.00%
Corrugated	5.00%

Apportionment of "Conveyor" Costs	% of total conveyors
Plastic	26%
Glass	31%
Aluminum	18%
Ferrous	8%
General	17%

Apportionment of "Paper Conveyor" Costs	
Newspaper	80%
Corrugated	20%

Table 21 - Separated Recyclables Facility Financial and Throughput Assumptions

	Apportionment of Residue Costs
Operating Costs	
Utilities	55,000
Insurance	25,000
Maintenance	75,000
Supplies	50,000
Residue Disposal	143,065
$/ton	50
% residue	10%
Equipment Replacement	35,250
% of Equip Cost	5%

Labor	Number	Salary/ Employ	Total
Management	2.00	35,000	70,000
Plastics	3.00	23,000	69,000
Glass	4.00	23,000	92,000
Aluminum	1.00	23,000	23,000
Newspaper	2.00	23,000	46,000
Corrugated	1.00	23,000	23,000
Ferrous	1.00	23,000	23,000
Other	5.00	25,000	125,000
Total	19.00	24,789 (average)	471,000

	% of Specific Material	% of all Material
Plastic	14%	0.78%
Glass	9%	2.00%
Aluminum	7%	0.16%
Newpaper	10%	5.49%
Corrugated	10%	0.76%
Ferrous	8%	0.60%
Total		

Apportionment of Supplies Costs	
Plastic	10%
Newspaper	20%
Corrugated	5%
General	65%

Landfills

The costs of landfilling are often presented in terms of weight rather than volume. WastePlan calculates landfill costs for a specified total waste stream, and reports total costs and average cost per ton (without distinguishing cost impacts of individual materials). In order to obtain material-specific marginal costs per ton, we assumed the same costs are incurred to landfill a cubic yard of any material[2]. Using WastePlan's average cost per ton figure, we estimated

Landfill cost per ton for material A = Average cost per ton * average waste density/density of material A

This adjustment implies the same landfill costs per cubic yard for all materials, resulting in much higher costs per ton for low density materials such as plastics and aluminum and lower costs per ton for high density materials such as glass.

Incinerators

Incinerator costs are almost solely proportional to the tonnage of the incoming material (assuming the incinerator has not reached the design limits on Btus per hour, as discussed above). Since WastePlan assumes costs proportional to tonnage, no adjustments were made to the WastePlan output. WastePlan automatically calculates energy generation and revenues based on Btu content of the waste stream, as well as ash generation; differing energy and ash content accounts for the differences in incineration cost by material.

Transfer Stations

Transfer station costs in WastePlan are calculated based upon the incoming weight of the material. There are two basic components of transfer station costs: final disposal costs and facility/transfer costs. The disposal costs depend upon the type of facility in use and their tip fee. For the purposes of this analysis, we have assumed that this disposal cost is proportional only to tonnage and not volume.

The facility and transfer costs depend on both volume and tonnage. The transfer trucks are filled by volume, but subject to highway limitations on allowable truck weights. Some facility costs, such as equipment costs, will be based upon tonnage; most, such as site and building costs, are proportional to volume. Because facility/transfer costs reflect a mix of factors, and final disposal costs account for about three-fourths of total costs, we have not altered the WastePlan tonnage-based cost calculation.

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[2] Of course, landfill operators frequently charge tipping fees per ton of material; however, many elements of their cost structure are clearly related to the volume occupied by the landfilled waste, rather than the weight of material.

V. RESULTS OF MARGINAL COST ANALYSIS

The final result of the analysis is an estimate of the marginal solid waste management cost, per ton and per cubic yard, for each type of packaging material. This marginal cost depends both upon the physical characteristics of the materials, such as density, compactability and incineration characteristics, and the way in which the solid waste system handles each material, i.e. how much is recycled in each of the counties, and how the remaining material is disposed of.

The results of the analysis are summarized in Table 22. On a weight basis, marginal costs of the various packaging materials vary from $24 per ton for aluminum to $250 per ton for PET plastic. Figure 1 shows the range in per ton costs across materials. On a volume basis, the range of costs is from $0.71 per cubic yard for aluminum to $23 per cubic yard for glass. Figure 2 shows the per cubic yard costs of all materials. These results show that all materials have a negative cost impact upon the solid waste system. One packaging option not considered here is to remove the packaging entirely which have no cost impact to the solid waste system. The results also show that source reduction efforts to reduce both weight and volume of packaging will be effective in reducing solid waste system costs.

Table 22 - A. Summary of Packaging Material Marginal Costs

	Marginal Cost Per Ton	Marginal Cost Per CY	Curbside Material Density
Aluminum	23.73	0.71	60
Ferrous	134.30	10.07	150
Glass	70.98	23.07	650
Corrugated	118.18	8.86	150
Other Paper	110.47	9.67	175
PET	249.80	4.37	35
HDPE	243.45	4.26	35
Other Plastics	235.06	4.11	35

The costs of individual waste management options for individual materials also showed a wide range. Aluminum showed a net benefits at the recycling facility, $734 per ton while PET and HDPE had roughly no cost impact. Plastics showed a net benefit at the incinerator of $2.50 per ton. (See Tables 23 and 25) The highest cost system was recycling collection of plastics at $357 to $367 per ton and landfilling of aluminum at $182 per ton.

Figure 1

Marginal Solid Waste Costs per Ton for Packaging Materials

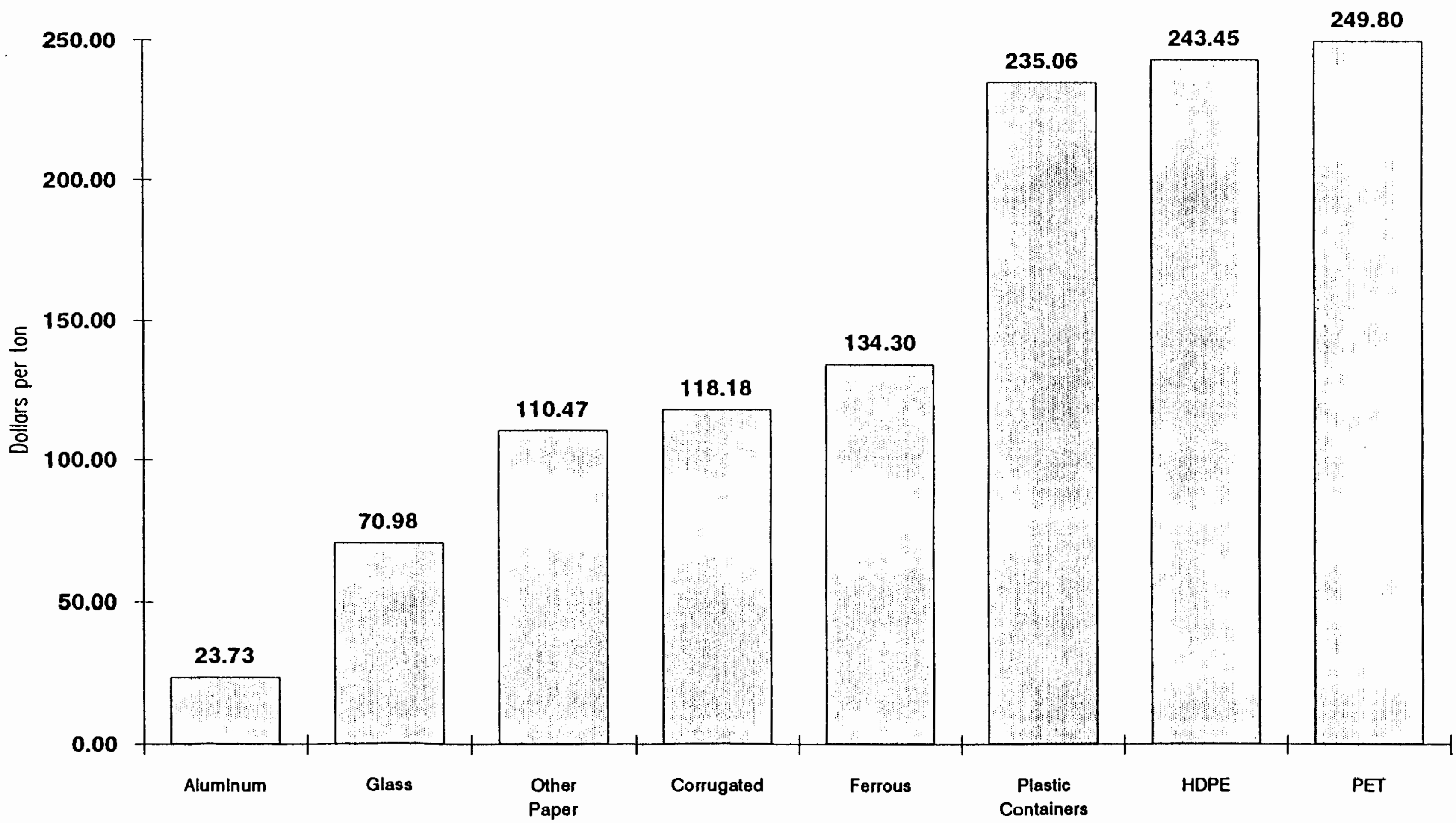
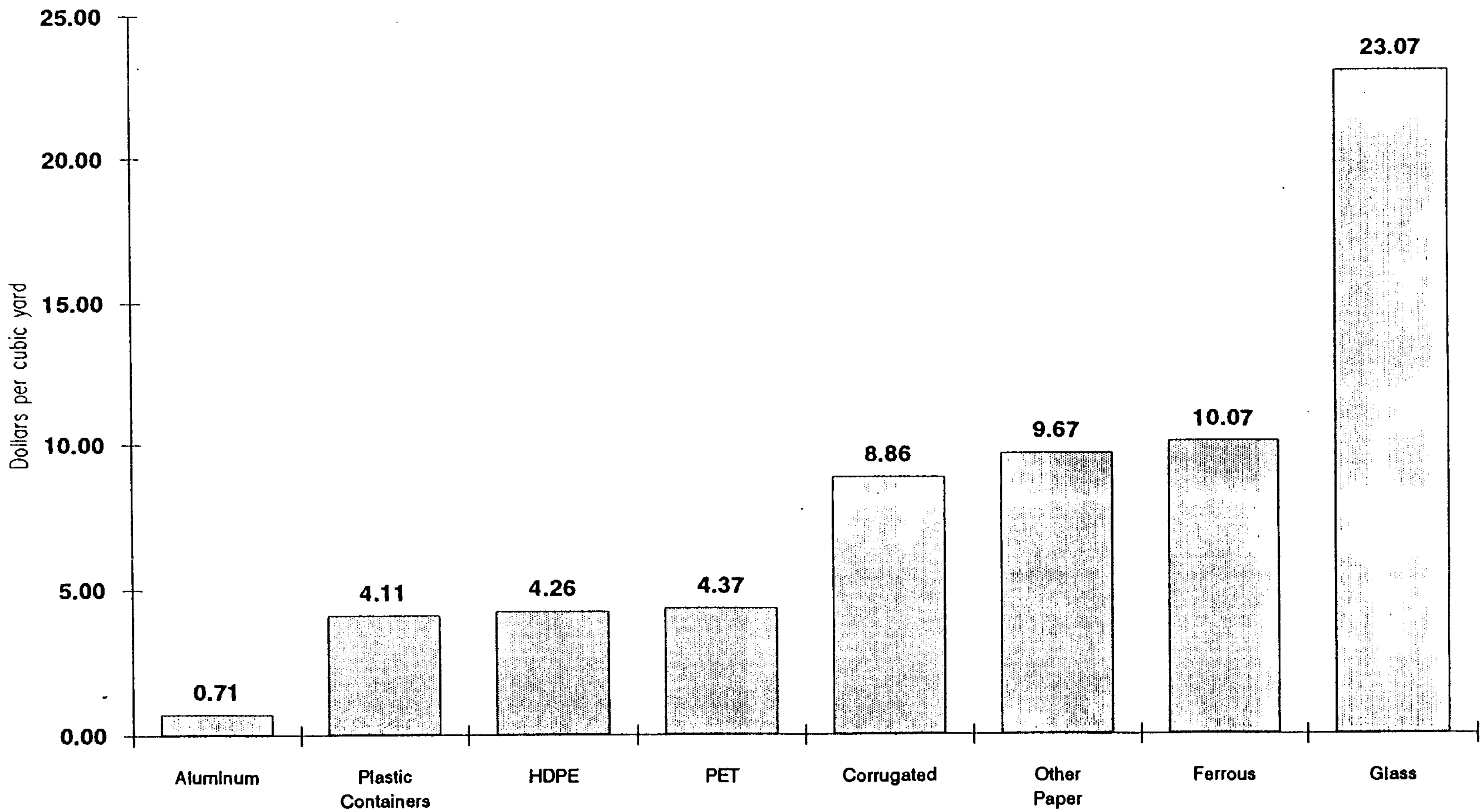


Figure 2

Marginal Solid Waste Costs per Cubic Yard for Packaging Materials



Another step is required to translate these per ton and per cubic yard costs into per package costs, a subject which we will address in later reports. Briefly, two or more packages containing the same quantity of final product may have different weights, or even different volumes. To calculate per package costs, it is necessary to weigh (or measure the volume of) actual packages, then multiply by our marginal costs per ton (or per cubic yard).

Effects of Material Density

Aside from aluminum, there is a clear correlation between the packaging material density and cost. The densest material, glass, is the least expensive per ton and the most expensive per cubic yard. At the other end of the spectrum, plastics, the least dense materials, are the most expensive per ton and the least expensive per cubic yard. In the middle on both a weight and volume basis are paper packaging, corrugated cardboard and ferrous containers, materials with roughly similar densities.

Aluminum, whose marginal cost is $24 per ton and $0.71 per cubic yard, is the clear least-cost winner, whether measured by weight or volume. In comparison to the next cheapest materials, its costs are about one-third of the cost per ton for glass and one-fifth of the cost per cubic yard of non-recyclable plastic containers.

The correlation between density and cost exists because the costs of most waste management operations are proportional to either weight or volume. Collection of materials is primarily a function of how quickly the trucks become full, how frequently containers are set out and how efficiently crews can collect materials. All of these characteristics are affected almost solely by the volume of the material being set out and recycled. Landfills, like trucks, fill up by volume. However, incinerators and transfer stations charge disposal based upon the weight of the material. The only waste management operation which is not primarily affected by weight or volume is the recycling facility, where the highly differentiated processing technology and material revenues play a large role in determining net costs for individual materials.

Effects of Btu Content and Ash Content

Aside from density, there are other factors which have an effect upon the material's marginal costs. Table 23 lists the marginal costs per ton of each of the solid waste operations for each material. That table shows that the marginal cost of incineration varies from a $94 per ton loss to a $2.50 gain. The Btu content and the inert content of the materials explain these differing costs. Materials with high Btu content release more energy when burned and therefore generate more revenues through electricity sales. Materials with high inert content create a lot of ash when burned; incinerator ash must be disposed at ash landfills at a high cost.

Table 23 - Summary of MarginalCost per Ton for Specific Waste Management Options

	Aluminum	Ferrous	Glass	Corrugated	Other Paper	PET	HDPE	Plastic Container
Recycling Collection	180.74	88.95	19.41	102.58	0.00	357.24	366.97	0.00
Garbage Collection	96.14	41.57	12.57	39.20	34.12	168.30	164.69	160.00
RecyclingFacility	-734.24	12.49	7.71	19.72	0.00	0.52	-0.59	0.00
Incineration	93.10	93.10	94.23	54.42	51.74	-2.54	-2.54	-2.52
Landfill	183.51	86.29	21.52	80.21	72.85	170.82	171.13	171.70
Transfer Station	115.01	115.16	115.19	115.06	113.37	115.73	115.51	115.20

Plastics, which have low ash content (about 8%) and very high Btu content (14,500 Btu/lb), actually generate net revenues from their incineration because electrical revenues are larger than operating costs and residue disposal. Their marginal benefit is about $2.50 for all plastics. On the other hand, aluminum, glass and ferrous have a high marginal cost (about $94 per ton) because they generate virtually no electricity but leave a large amount of ash requiring disposal. The marginal cost of paper products is about $52 to $54 per ton, which is intermediate between the values for plastics and for the non-combustible materials. The ash content of the paper is roughly the same as plastics, but the Btu content is much lower (5,700 to 6,300 Btu/lb) so electric revenues are lower.

Lower ash content can be achieved for ferrous materials through the separation and recycling of metal from the bottom ash. While this practice is common for large, non-packaging pieces of scrap metal, it is less common for small pieces of metal such as ferrous food containers. We have assumed here that no material is being recycled at the incinerators, and all material in the ash is landfilled as residue.

Effects of Material Revenue and Recycling Processing Costs

The per ton revenue for each material has a large impact upon the marginal costs of the recycling facility. In addition to revenues, the particular technologies used for separation (if applicable) and processing of the materials will also have an impact upon the marginal costs. Aluminum receives $900 per ton in revenues, and has processing costs, for labor, equipment and operations of about $166 per ton. The processing cost is the average of the three types of facilities modelled in this scenario, weighted according to their use in

New Jersey. On net, aluminum has a marginal cost of about -$734 per ton (the negative cost means there is a net *benefit* of $734 per ton).

Compare this to plastics, which have the second highest revenue at $140 per ton. The processing costs are about $140 per ton, approximately the same as aluminum. So marginal costs are $140 - $140, or about $0 per ton. Even though processing costs are roughly the same for aluminum and plastics, the marginal costs for these materials differ significantly because of the difference in revenues.

Glass containers, with revenues at $40 per ton (an average of clear, brown and green) and processing costs of almost $47, have a net marginal cost of about $7 per ton. Processing costs per ton are significantly smaller for glass than for plastics or aluminum because of glasses high density and the large amount of glass flowing through the facility.

Ferrous containers have a net marginal cost of about $12 per ton even though the ferrous revenue of $35 per ton is roughly the same as for glass. Ferrous processing costs are higher (about $47 per ton) primarily because the tonnage flow is smaller. The only paper product recycled is corrugated cardboard, which has a marginal cost of $20 per ton.

Effects of Recycling Rate and Recyclability

Among the packaging materials studied, recycling rates of each material vary and several are not recycled at all. A material's recycling rate can have a large impact upon its marginal cost if the difference between the recycling costs and disposal costs are significant. We have projected likely rates of recycling and disposal, based on the use of both currently operating facilities and projected new facilities identified in Section III above. The results are the projected recycling rates as presented in Table 24, along with the breakdown of the method of disposal used for the remaining portion[3].

Table 25 presents the costs of recycling versus the three disposal options for each of the packaging materials. For most of the materials recycling is less expensive than disposal. Aluminum is the most extreme case, with recycling saving the system $554 per ton, while disposal options cost from $189 to $280 per ton. For glass, each ton recycled costs about $27, while each ton disposed costs from $34 to $128. Ferrous containers and corrugated both have roughly the same costs for recycling ($100 and $122 per ton for ferrous and corrugated, respectively), and their disposal costs are similar as well. The one exception is incineration, where corrugated is less costly because of its high energy content and low ash content. The comparison of recycling and average disposal costs can be seen in Figure 3.

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[3] The facility mix, as described in Section III, assumed the completion of several new incinerators. Thus Table 24 projects rates of incineration well above actual experience to date. Changes in state policy, which occurred too late for inclusion in this analysis, now suggest that rates of incineration may remain lower than those shown in Table 24. The most important effect of lower incineration rates on our analysis would be an increase in the estimated marginal costs of handling plastics, since (as shown in Table 25) incineration is by far the lowest-cost option for plastic containers.

Figure 3

Marginal Recycling and Disposal Costs of Various Packaging Materials

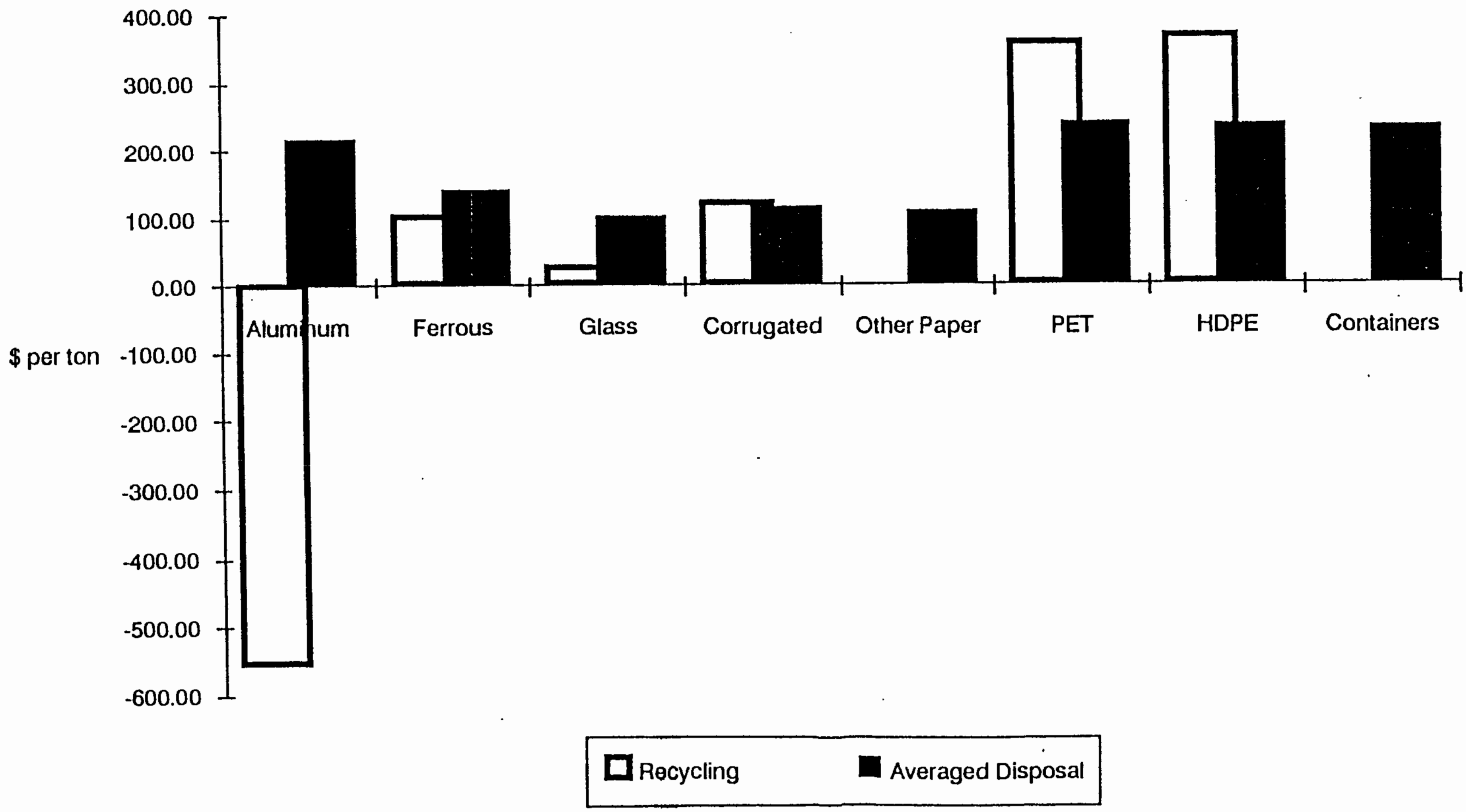

Table 24 - Projected Percentages of Packaging Materials Handled by Different Waste Management Options

	Aluminum	Ferrous	Glass	Corrugated	Other Paper	PET	HDPE	Plastic Containers
Recycling	25%	18%	41%	18%	0%	8%	4%	0%
Incineration	37%	40%	31%	38%	46%	44%	45%	46%
Landfill	18%	13%	10%	21%	23%	21%	22%	23%
Transfer Station	19%	29%	19%	23%	32%	28%	29%	31%

(See text and footnote on page 76 for explanation of these projections)

Table 25 - Per Ton Marginal Costs of Waste Management Alternatives for Different Packaging Materials

	Aluminum	Ferrous	Glass	Corrugated	Other Paper	PET	HDPE	Plastic Containers
Recycling	-553.50	101.44	27.12	122.31	0.00	357.76	366.38	0.00
Incineration	189.24	134.67	106.79	93.62	85.86	165.76	162.15	157.48
Landfill	279.66	127.86	34.09	119.41	106.97	339.13	335.82	331.70
Transfer Station	211.15	156.73	127.76	154.25	147.49	284.03	280.20	275.20

Plastics are an exception to the rule that the marginal costs of recycling are less than disposal options. The marginal cost for recycling plastics is roughly $358 to $366 per ton, while disposal options range from about $20 to $200 per ton cheaper than recycling. The big difference occurs in the area of collection costs. With the marginal cost of recycling collection about $360 per ton, (i.e., the additional cost due to separate recycling collection) while garbage collection is just below $170, the difference between these collection costs is about $190 to $200 per ton! Trucks are expensive, and bulky, light-weight plastics fill them up with relatively little tonnage. Garbage trucks typically compact the wastes, alleviating this problem somewhat; recycling trucks typically do not compact plastics.

When it comes to facility costs, incineration and recycling of plastics are similar, and in fact both are close to break-even (-$2.50 per ton for incineration and on average less than $1.00 per ton for recycling). Thus in a comparison of incineration vs. recycling options, the difference in collection costs is the dominant factor. Recycling fares better when compared to landfill or transfer options, since the high marginal cost of landfilling plastics ($170 per ton) or of out-of-state disposal offsets much of the collection savings.

Differences between recycling and disposal marginal costs make recycling rates very important. In the case of plastics, higher recycling rates, with current technology, will increase the weighted average marginal costs. For all other materials, higher recycling will decrease these costs. This relationship can be seen by comparing the weighted average of the marginal costs for the three plastics in Table 25. PET, which has the highest recycling rate, has the highest weighted average marginal cost at $252 per ton. HDPE is next with $243 per ton while non-recyclable containers are $234 per ton.

For the other materials, the savings by recycling additional materials varies widely. For aluminum, each additional ton recycled saves about $770. Glass savings are next largest at about $74 per ton, while for ferrous, there is a moderate savings of about $40 per ton. Corrugated cardboard is the only material where recycling and disposal costs are roughly the same, with recycling being only $5 per ton less expensive.

The method of disposal for each material varies, as well, making the percentage of packaging recycled, incinerated, landfilled or transferred an important factor in the final marginal cost. This breakdown of disposal method can vary greatly, as projected in Table 24. For aluminum, the amounts landfilled and transferred are roughly equal (18% landfilled and 19% transferred), while for glass, the amount transferred is about twice as great as the amount landfilled (10% landfilled, 19% transferred). This likely means that in counties where landfilling is the method of disposal, recycling rates are higher for glass than for aluminum. Differences like these are specific to the conditions in New Jersey and can have potentially significant impacts upon marginal disposal costs.

Using the Marginal Costs Outside of New Jersey

Under what conditions can our estimated New Jersey marginal costs be used in other regions? For regions with different recycling rates or mixes of disposal options, but similar costs within each option, our marginal costs for individual programs (recycling, incineration, transfer and landfilling) can simply be recombined. A weighted average of our program costs can be calculated, based on the program mix for a given area.

Of course, the marginal costs of facilities or collection programs in another region may differ from those used in this study. While every area is of course unique, we would expect our New Jersey collection systems and recycling facility costs to be most broadly applicable. The costs used in our New Jersey analysis fall within the range of costs found in a number of other regions of the country, though costs of individual programs do vary widely.

Of the facilities, the costs of the recycling facility are the most directly applicable to other regions because they face fewer siting constraints. With its high population density and associated land use constraints, and its active environmental movement, New Jersey has experienced difficulty in siting new disposal (landfill or incineration) facilities. Such factors escalate. Landfill costs in particular are very high, because the amount of available capacity is far less than the amount of waste generated. This produces not only high landfill costs, but also high transfer costs, because waste must be transferred long distances, and high incineration costs, because ash disposal costs are high. These disposal figures may be appropriate for other regions in the Northeast, but will likely be lower for other regions of the country, where landfill costs are lower. However, if landfill costs are rising in other regions, New Jersey costs may be taken as a foreshadowing of future cost levels which may be relevant to long-range planning today.

When using the marginal costs in Table 23 or 25 in regions with different recycling rates, it is important to recognize that there will be a small impact upon the marginal costs of collection programs when changing the recycling rate. This change is produced because of the slight economies of scale associated with collection programs which favor the collection of larger amounts of materials. This occurs because there are significant fixed costs associated simply with servicing each residence, regardless of how much they set out. The larger amount of materials these fixed costs can be spread across, the lower the marginal cost will be.

This relationship can be seen by comparing the costs of collection for the three plastics in Table 23. Recycling collection of PET is less than HDPE because a high percentage of material is collected. While 8% of the PET is collected at $357 per ton, 4% of the HDPE is collected at $367. However, while recycling collection is more expensive for HDPE, garbage collection is less expensive because more HDPE is collected for disposal than PET. While 96% of HDPE is collected at $165 per ton, 92% of the PET is collected at $168 per ton. The non-recyclable containers, which are all collected for disposal, are even less expensive at $160 per ton. One can imagine that if recycling rates for plastics increased, the marginal costs of recycling collection would continue to drop below $350 per ton, and garbage collection costs would increase above $170 per ton.

For accurate region-specific marginal costs of collection programs, a WastePlan analysis using the recycling rates of the new region should be performed. However, for most uses, the marginal costs reported in this analysis should be adequate.

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89-024/4NJ

Assessment of Impacts of Production and Disposal of Consumer Packaging on the Environment

NJ DEPE Contract P31152

Report #4:
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TABLE OF CONTENTS

CHAPTER 1 - ASSESSING THE FULL COST OF PACKAGING PRODUCTION

1.1 INTRODUCTION

1.1.1 Costs Associated with Packaging Production and Disposal

Two broad categories of cost arise from the production and disposal of packaging materials: first, the *conventional* costs of packaging production and disposal and second, the environmental damage or *external* costs associated with production and disposal of packaging materials. Both types of cost are actually borne by people: some costs are borne by each member of society, other costs only by some; some costs will be borne in the near term, other costs over the longer term. It is appropriate to assign explicit dollar values to the various externalities, i.e. environmental/public health impacts, associated with the production and disposal of materials.

The *conventional packaging production cost* is the monetary cost of mining and processing raw materials, transporting materials, and producing the packaging material. Thus, these costs include capital (for example, the steel mill), labor (the steel workers), and natural resource (iron ore, energy) costs. The *conventional waste management cost* is the monetary cost of collection, transport, processing, and disposal of solid waste. For collection and transport, this is the purchase and maintenance of trucks (capital) as well as the collection workers' wages (labor) and the fuel consumed to operate the trucks (natural resource). For processing and disposal, the conventional cost includes the cost of constructing, operating, and closing different waste disposal facilities. These conventional costs are captured in the price that consumers pay both for goods and for waste management.

Environmental damage, or external costs, can occur at each stage of packaging material production and disposal. Packaging production can incur environmental costs due to air emissions and water emissions associated with raw material extraction, raw material processing, and packaging material production. Other costs include resource depletion, human and ecological effects associated with environmental emissions, environmental degradation, and habitat loss/alteration.

Some of these environmental costs are monetized; for example, the cost for installing and operating a pollution control device at a facility will be factored into the price of goods produced at that facility. However, many environmental costs typically are not monetized. Pollutants emitted into the environment (i.e., those emissions that are not captured by the pollutant control devices since these devices are not 100% efficient) can have human health impacts. These pollutants can also have ecological and environmental impacts - fish kills, habitat loss, water contamination, and soil degradation. All of these costs are not typically monetized.

1.1.2 The Case for the Monetization of Externalities

It may well be argued that a large component of the environmental damages cannot or ought not be valued in monetary terms. Many people feel that the human race has a responsibility for creation, and that this has an intrinsic value that lies outside the sphere of money. At the same time, many also believe that there is no objective way by which such intrinsic values -- e.g. a human life, a pristine habitat -- can be quantified.

However, the treatment of the environment is, in our open society, a matter of public policy. On the basis of both scientific results and public discourse, society either broadly or at the local community level may express its willingness to pay to avoid or, alternatively, accept certain levels of environmental degradation. Environmental regulations set limits that affect the costs of production, distribution and consumption of goods. Decisions affecting the treatment of environments and risks to human health -- within the constraints of environmental protection -- are made on a daily basis. These policies and choices imply specific valuations of natural and human resources. Assigning dollar values to environmental impacts makes these existing valuations explicit.

Systems have been suggested that appear to avoid the monetization of environmental damages, for example scoring and ranking systems that assign points to resource alternatives for their impact on the environment. However, these evaluation systems still do contain implicit monetary valuations. The choice of one resource alternative over another implies a monetary valuation of the environmental impacts. If the option with the higher monetary cost is chosen, then this implies that the difference in environmental impacts between the two policies is valued higher than the difference in monetary cost.[1]

Suppose, for example, that the two resource alternatives are compared using a scoring system to account for their environmental impacts. Assume that on the basis of conventional economic costs one is preferable (i.e., cheaper) but that with the environmental scoring system the other is preferable. This implies that the difference in environmental scores is worth at least the difference in conventional economic cost and thereby overcomes it. At their best, scoring systems will logically and consistently embody the monetary values that the relevant community holds. But they would tend to obscure rather than clearly illuminate these values and their relationship to conventional costs.

Systems for choosing between alternative resources or plans with different environmental attributes, as well as systems of environmental targets or constraints, imply or can be expressed in monetary terms. For example, the decision to construct a reservoir that provides water at a lower monetary cost than conservation measures but which destroys a unique ecosystem implies that preservation of this ecosystem is worth less (in dollar terms) than the savings in the cost of water provision. Another example is a ban on the use of polystyrene for the purpose of fast food packaging. This ban implies the judgement that the externality caused by polystyrene in this use is to be valued higher (i.e. more negatively) than the cost imposed on the businesses that have to turn to the next "best" (from their point of view) option for fast food packaging.

Another argument in favor of monetizing environmental externalities is that these costs can be added to conventional costs to provide a single evaluation of the total cost. In our disposal cost report we calculated the conventional solid waste management costs associated with handling packaging materials in the New Jersey solid waste system. By monetizing the environmental costs, we can determine the total (i.e., conventional plus environmental) cost associated with handling packaging materials in the solid waste management system. The monetization of environmental costs associated with packaging production allows these costs to be added to the disposal costs. Thus, monetizing environmental externalities produces a common denominator for making comparisons between various packaging materials and various waste management alternatives.

1.2 ASSESSING THE ENVIRONMENTAL COST OF PACKAGING MATERIALS

In this section we discuss our choice of methodology for valuing environmental costs, describe how it should ideally be applied, and identify some of the problems that arise when this methodology is applied to the environmental costs of packaging production and disposal. This section also presents and explains our valuation of individual pollutants.

Three methods are currently employed to value environmental costs. The first approach attempts to estimate the physical damage associated with the degradation of the environment. This implies tracing the physical environmental impacts and valuing the physical damage. The second approach, favored by academic economists, concentrates on consumer preferences and efforts to elicit them. The third approach uses pollution abatement and remediation costs to indicate the value that society places on environmental damage. This last approach is adopted for this study. A detailed discussion of each method and the reasons for selecting pollution abatement and remediation costs are presented in Appendix I to this chapter.

Our control cost approach is based on the notion that the marginal cost per unit of pollution abatement rises with the amount of pollution abated.[2] The value that society places on residual emissions is a point on this marginal cost function. The *highest* amount that is required, or actually observed to be spent on the abatement of a specific pollutant, can be taken as a lower bound of the value that society places on removing this pollutant from the environment.[3] This value, which is associated with removal of the pollutant, is the cost that is ascribed to the presence of that pollutant.

When society or a community, through its regulations and policies, establishes pollution limits -- either through ambient concentrations, air basin aggregates, facility-specific emission caps, technology specifications, or outright bans on certain materials or facilities -- it is establishing its monetary value for the avoided pollution at the margin. Of course this is an evolving process of revealing the values and their monetary expression, which depends upon science, public discourse, and policy. Thus, the values may change over time.

The task then is to identify regulations and policies that address the pollutants both associated both with waste management and industrial processes, and to determine the costs of complying with these regulations. The pollutants that are typical for waste management and packaging material production include a host of hazardous substances, EPA's criteria air pollutants and greenhouse gases. Each pollutant group and their valuation is discussed below.

EPA's Criteria Air Pollutants

One class of pollutants encountered in production and disposal of packaging materials is the EPA criteria air pollutants. These include particulates, sulfur dioxide (SO_2), carbon monoxide (CO), ozone, oxides of nitrogen (NO_x), and lead. They impair human health, are ozone precursors, and precursors of acid precipitation. Under the Clean Air Act, the U.S. Environmental Protection Agency has been mandated to develop National Ambient Air Quality Standards (NAAQS) that establish permissible ambient concentrations for these pollutants. Regulatory limits for volatile organic compounds (VOCs) were also established, but only as a reference in regard to the ozone standards. The goal of these standards is to "protect the public health."

Enforcement of the NAAQS has been delegated to the states. The states are required to submit a state implementation plan (SIP) which shows how the ambient air standards will be implemented, maintained, and enforced.

Southern California, especially the South Coast Air Quality Management District (SCAQMD) which perhaps has the worst air quality in the nation, has developed and analyzed an array of air quality regulations. The cost of meeting these various regulations has been studied and some of these costs have been adopted by the California Energy Commissions which is planning to internalize the external cost of energy production.[4] We have used this work to establish prices for criteria air pollutants. However, lead has been evaluated below in the hazardous substances category.

Greenhouse Gases

Another group of pollutants are the greenhouse gases. These are carbon dioxide (CO_2), carbon monoxide (CO), methane (CH_4), nitrous oxide (N_2O), oxides of nitrogen (NO_x), and chlorofluorocarbons (CFCs). The most important greenhouse gas is CO_2. While other gases have a higher warming potential per unit, CO_2 dominates the other gases because of its high share among the pool of greenhouse gases which are emitted and which are already present in the atmosphere.

No regulation exists to date that addresses greenhouse gases. However, the ongoing debate about the greenhouse effect and the apparent willingness of nations to subject themselves to protocols does reflect a concern about the issue of global climate change. Some nations have already gone further and have adopted taxes that target the production of greenhouse gases. For example, Sweden taxes CO_2 emissions at $40 per ton.[5] There is ample evidence that societies do not attribute a value of zero to the emission of greenhouse gases, implying that greenhouse gases should be included in the valuation of environmental impacts.

In the absence of regulations and reference doses, one measure that could be used to value greenhouse gases is reforestation, as a means to offset CO_2 production. Trees are a "carbon sink"; they absorb CO_2 and produce oxygen. One could calculate the cost of planting the number of trees required to absorb a certain amount of CO2 and thus obtain a value for the gas. There are no unique values for the cost of reforestation. Much depends on where the trees will be planted. Reforestation in a less developed country with a low wage level will cost less than reforestation in the United States. The costs also depend on the terrain that the trees are planted in, and other circumstances. Clearly, reforestation costs can only be interpreted as a placeholder for a more substantive valuation of CO_2. However, the California Energy Commission has developed a value for CO_2, drawing on discussion of reforestation costs; we adopted the California value.

Other greenhouse gases can be valued on the basis of the estimate for CO_2. These gases have different impacts in the atmosphere; specifically, they differ in their potentials to produce global warming. While the equivalences of the global warming potentials are not exactly known, there are some estimates as to how these gases relate to each other. The global warming potential of methane, for example, has been estimated to be ten times that of CO_2.[6] The environmental costs of the greenhouse gases other than CO_2 are calculated as the product of the value for CO_2 and the global warming potential equivalent of the specific gas.

Hazardous Substances

The largest group of pollutants falls into the group we have termed "hazardous substances." These pollutants are neither criteria air pollutants (except for lead) nor greenhouse gases. As many of these pollutants are not regulated in the environment, the cost method used for criteria pollutants and greenhouse gases cannot be applied to this class of pollutants. Therefore, a different approach is needed to evaluate these pollutants.

Several complications arise in developing prices for hazardous substances associated with the production and disposal of packaging materials. First, what is the appropriate control cost to determine society's willingness to pay for the control of pollution? In order to fully assess the highest price society is willing to pay, a wide range of regulations impacting hazardous substances must be examined including the cost of controlling hazardous emissions from industrial sources and from the solid waste management system.

Second, when one control device or control measure deals with a group of very different pollutants, the question of how to attribute the joint cost of pollution abatement to individual pollutants becomes an important issue.[7] One potential solution is to find different regulations for different pollutants, and to attribute the cost of a control device to only that pollutant that the device was intended to abate. However, this is quite difficult. Moreover, it is possible that the device was intended to control the full mix of pollutants.

Evaluating the entire body of regulations in place affecting emissions of hazardous substances would have been prohibitively time-consuming. Instead, we investigated control measures impacting lead. As lead is a regulated pollutant, control costs for lead can be determined. Assuming pollutant control costs are proportional to the damage associated with a pollutant, the control cost for lead can then applied to other hazardous pollutants based upon the relative damage they cause as compared to lead. (The derivation of prices is discussed later in this chapter.) We have therefore combined the control cost approach with a **health effects ranking system**, a system which ranks pollutants according to the relative damage they cause. Specifically, this ranking system establishes equivalencies between individual pollutants, such that the health impacts caused by any pollutant are expressed in proportion to the impacts of any other. In other words, the system establishes relative numerical values to reflect the relative toxicity of various pollutants.

Construction of such a health effects ranking system is an extremely complex undertaking. There is no unique catalogue of criteria to be employed. No such system can take account of all environmental impacts of all pollutants. Ultimately, the relative impact of various pollutants depends upon many variables such as their transport in the environment, the exposure of sensitive populations, and the exposure-response relationships of those populations. Such analysis, which is included in the risk assessment framework, is beyond the scope of this study. Nevertheless, applying such a hazard ranking system is an improvement over the simple averaging of control costs over pollutants with very different potentials for causing environmental damage. Averaging control costs over different pollutants implicitly assumed for example, that one pound of sulfur dioxide has the same impact as one pound of benzene, two pollutants that have very different health effects.

Numerous hazard ranking systems have been developed in the past decade to help establish priorities for those chemicals requiring regulations and further environmental/health effects studies. (For a further review, refer to Appendix II.) These studies typically look at a wide range of factors for each chemical including indicators of human health, ecological impacts, yearly production quantities, and release into the environment. Each of these factors is then scored independently, yielding a scoring matrix. Interpreting the matrix can be difficult as it requires judgement, or valuation, of the importance of each factor.

Due to the drawbacks of using risk assessment methods and hazard matrices, a simplified ranking system was developed instead. This ranking system is based upon human health effects only (as extrapolated from animal testing); environmental impacts are not considered.

The first step in developing the health effects ranking system was to classify the list of pollutants associated with the production and disposal stages into carcinogens (cancer causing pollutants) and noncarcinogens (pollutants that cause toxic health effects other than cancer). The health impacts of these two classes are measured differently, thereby requiring a separate ranking in each class. Pollutants were assigned to these two classes based upon the U.S. Environmental Protection Agency's classification system.[8] *

Carcinogenic compounds were ranked based upon each pollutant's cancer potency factor, measured as milligrams pollutant/kilogram bodyweight/day (see Table 1.1).[9] This factor is indicative of the cancer risk associated with a pollutant. Isophorone has the lowest potency factor of the carcinogenic pollutants associated with production and disposal; its potency factor was used as the baseline of comparison for carcinogens. The potency factors of other carcinogens were then compared to isophorone to derive "isophorone equivalents." Thus, for example, the isophorone equivalent for benzene is 7, meaning that benzene is 7 times more potent in causing cancer than isophorone.

Noncarcinogenic compounds were ranked based upon each pollutant's oral reference dose (see Table 1.2).[10] The reference dose (measured as milligrams pollutant/kilogram bodyweight/day) is an estimate of the maximum daily level of exposure which will not cause harm.** Less toxic chemicals have a higher reference dose since a higher dose is required to elicit an effect. The inverse of the reference dose (i.e., 1/reference dose) was used as the ranking factor so that a smaller number would be indicative of lower toxicity. As xylene has the smallest value based upon this scale, it was used as the baseline of comparison. The inverse of the reference dose of all other noncarcinogenic pollutants were then compared to xylene to derive "xylene equivalents." Based upon this equivalency, lead is 1429 times more toxic than xylene, for example.

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* The tables in this chapter provide a complete list of pollutants associated with the production of all packaging materials considered in this study. Not all of these pollutants however are ranked in the health effects ranking system.

** While reference doses, or RfDs, may be determined for two routes of exposure - oral and inhalation - we ranked noncarcinogens solely based upon oral RfDs due to the fact that for many pollutants oral RfDs are available in the literature but inhalation RfDs are not. The difficulty in performing inhalation toxicity studies may explain the absence of inhalation RfDs for many pollutants.

While the ranking scheme described above allows a long list of pollutants to be compared, the problem remaining is that there are still two disparate groups of pollutants - carcinogenic and noncarcinogenic pollutants. These two groups do not lend themselves easily to comparison. An exposure to even a small dose still carries a positive, albeit small, cancer risk while theoretically, there is a "safe" dose for noncarcinogenic pollutants. Thus, it is difficult to compare the two groups.

One method that can be used to infer a relationship between the two groups of pollutants is to compare the regulated levels of isophorone and xylene, the least hazardous chemicals from each of the two groups. The only regulations for these two chemicals are in the workplace environment. The Occupational Safety and Health Administration (OSHA) sets permissible exposure levels (PELs) that specify the amount of a pollutant to which a worker can be exposed, averaged over the course of an eight hour workday. The PEL represents the concentration of a pollutant to which daily exposure will not incur an adverse health effect in exposed workers. OSHA has set a PEL of 100 parts of xylene per million parts of air (ppm) and a PEL of 25 ppm for isophorone.

The unitless exposure limits expressed in ppm can be converted to milligrams of pollutant per cubic meter of air. For xylene, a PEL of 100 ppm corresponds to 433 mg/m^3 and for isophorone, a PEL of 25 ppm corresponds to 141 mg/m^3. This implies that a "safe" dose of xylene is 3 times the "safe" dose of isophorone. On the occupational health standards basis, isophorone has a xylene equivalent of 3. A carcinogen such as benzene, with its isophorone equivalent of 7 (as cited above), then has a xylene equivalent of 21. We have used this approach to express all carcinogens in terms of xylene equivalents, producing a unified ranking for both types of hazardous substances. We have used this factor of 3 to weight the isophorone equivalents to reflect the fact that a given dose of a carcinogen is not equivalent to the same dose of a noncarcinogen. Table 1.3 displays the aggregate ranking system. It is important to note that this aggregate system presents *relative* values - that is, it allows relative comparisons between pollutants. Some pollutants in this table can both cause carcinogenic and noncarcinogenic health effects. To determine the combined ranking for these pollutants, the xylene equivalents and the weighted isophorone equivalents for the pollutant were averaged. Thus, from Table 1.3 we can conclude that cadmium, which has a combined ranking score of 4,346 is 3 times worse than lead which has a combined ranking score of 1,429.

Table 1.2 is 4000

Problems arise when using PELs to compare chemicals. Since they are developed for use in the workplace, and workers are typically relatively healthy adults, PELs may not reflect the effect of hazardous substances on more sensitive members of the population (such as children, the elderly, or those with compromised health). In addition, as politics oftentimes plays a role in establishing PELs, this standard may not accurately reflect relative health effects.

Other methods were also explored for ranking and comparing carcinogens and noncarcinogens. For example, other indices as well as PELs are used in evaluating pollutants in the workplace. The American Conference of Governmental Industrial Hygienists (ACGIH), a non-governmental independent organization, issues Threshold Limit Values (TLVs), similar to PELs, which specify the amount of a pollutant to which a worker can be exposed, averaged over an eight hour workday. As TLVs are *recommended* rather than regulated concentrations, we did not use this index. Other worker-related indices such as short term exposure limits (STELs) and immediately dangerous to life and health (IDLHs) are only established for a small number of chemicals and are thus not useful for evaluating the wide array of chemicals emitted from the production and disposal of packaging materials.

Other regulations affecting pollutants were also explored. For example, the Safe Drinking Water Act sets maximum concentration levels (MCLs) for pollutants in community water systems. To date, MCLs have only been set for a handful of pollutants. Likewise, the Clean Air Act regulates toxic air pollutants; only a small number of these pollutants have been regulated to date.

Another alternative we considered was comparing the dose of a carcinogen associated with a one in a million risk of cancer to the RfD for non-carcinogens. The problem with this methodology is that the RfD is considered a "safe" dose while the dose of a carcinogen associated with a one in a million risk of cancer still poses a health risk, albeit a small risk. Thus, these two benchmarks are not equivalent.

While it is not possible to ascertain how much greater society values carcinogens as opposed to non-carcinogens, clearly the health risk posed by carcinogens is perceived to be greater than the risk posed by noncarcinogens. This fact has been the subject of numerous articles and books.[***]

Several pollutants listed in Table 1.3 do not have a ranking attributed to them as no toxicity data were available for these pollutants, or the EPA database classified the data as "inadequate." As discussed in the following section, where possible, we inferred a price for these pollutants so that their environmental costs were accounted for. However, we were unable to include many of the pollutants without health effects data in the environmental costs of production and disposal. As a result, the environmental costs for packaging materials are underestimated. In addition, some pollutants in this table can both cause carcinogenic and noncarcinogenic health effects. However, since each pollutant can only receive one price, as previously discussed we determined a combined ranking score using the xylene equivalents and the weighted isophorone equivalents.

When we assigned pollutant prices (as discussed in the following section) one curious result was produced: paper emissions initially appeared very expensive. This figure was completely dominated by the share of hydrogen sulfide in the cost, caused by the high price

[***] See for example, Edith Efron's book *The Apocalyptics*, 1984, Simon and Schuster, NY.

which was assigned to hydrogen sulfide. This high price, in turn, was due to the very high value which the ranking system assigned to hydrogen sulfide. Further review of the reference dose in the EPA Integrated Risk Information System (IRIS) database indicated "low" confidence in the study on which the reference dose was based. We next compared the OSHA standards for xylene (100 ppm, or 433 mg/m^3) and hydrogen sulfide (10 ppm, or 14 mg/m^3). From this comparison we inferred that hydrogen sulfide is 31 "xylene equivalents" and assigned it the relative price.

1.3 DEVELOPING PRICES FOR POLLUTANTS

At this point we have a relative scale by which one can compare pollutants. The next step is to determine the "price" for each pollutant, a dollar amount per pound of residual pollutant emission. This price is a valuation of the damage that this pound of specific pollutant imposes on society. If the price of one pollutant were fixed, prices could be generated for the other pollutants using the scale developed in the health effects ranking system. As lead is one of the few regulated hazardous pollutants, the marginal control cost of lead can be determined and used as a reference point for comparison.

The Massachusetts Department of Public Utilities has been investigating environmental externality values to be used in energy resources planning. In conjunction with this activity, marginal control costs for lead have recently been determined.[11] Two sources of lead in the environment were examined - lead emitted from secondary lead smelters and lead in paint. For lead smelters, the cost of reducing lead emissions as required under the Clean Air Act Amendments was examined. Based upon the examination of various pollution control devices, marginal control costs for lead were determined to be as high as $500 per pound of lead removed by the control device. For lead in paint, it was determined that the highest marginal control cost is the cost of removing lead painted window sashes. The marginal control cost for window sash replacement was estimated to be as high as $25,000 per pound of lead removed.

Building upon this work, in another study Tellus Institute estimated marginal control costs for lead and found a range of $520 to $4,621 per pound of lead controlled.[12] The average control cost for lead was found to be $1,600. This average cost was used in this study.

To determine the prices for the remaining hazardous pollutants, the combined ranking score for each pollutant was compared to the lead score. For example, as shown in Table 1.3, cadmium has a combined ranking score of 4,346 while lead has a combined ranking score of 1,429. Thus, cadmium is approximately three times as hazardous as lead. Therefore, cadmium is assigned a price three times the per pound control cost of lead, or $4,868 (see Table 1.4).

For criteria air pollutants and greenhouse gases, we used the numbers adopted by the California Energy Commission. The price for methane was obtained as explained above, i.e. by applying the price of carbon dioxide to the global warming potential of methane, measured in CO_2 equivalents.

As discussed in the preceding section, there were several pollutants that we were initially unable to price since we lacked toxicity information for them, as required by our health effects ranking system. Where possible, we inferred costs for these pollutants as described below. Hydrogen chloride was not initially assigned a price due to lack of a reference dose. We assigned the pollution abatement based sulfur dioxide price to hydrogen chloride since both pollutants are controlled with similar control devices. We were unable to initially assign a price to coke oven emissions as it is actually a class of pollutant rather than a single pollutant. As benzene is a major component of coke oven emissions, the benzene price was assigned to the entire coke oven emissions pollutant class.

1.4 IMPACTS OF PACKAGING PRODUCTION AND DISPOSAL

The pollutant prices in Table 1.4 were used to determine the environmental cost of producing and disposing one ton of specific packaging materials. In Chapter 2, the impacts of disposal are presented; the impacts of production are presented in Chapter 3.

1.5 ENDNOTES FOR CHAPTER 1

1. Bernow, S. B., et al., 1990. *Incorporating Environmental and Economic Goals into Nevada's Energy Planning Process*. Tellus Institute, Boston, MA, July.

2. Marginal control costs usually increase with the amount of pollution control. For example, if a pollution source tries to reduce its emissions with a control device that removes 80% of the pollution, 20% is still emitted. The source could purchase a second device that is the same as the first to further reduce these emissions. The second device would control 80% of the remaining 20%, or 16% of the uncontrolled emissions. Thus, the first device would reduce emissions by 80%, while the second device, which cost the same as the first, would only reduce emissions by 16%. Therefore, each unit emission reduction for the second device costs more than for the first device.

3. One could take the next highest cost option for pollution reduction, presumably explicitly or implicitly rejected by society, as a current upper bound.

4. California Energy Commission, 1990. Committee Order for Final Policy Analysis, Docket No. 88-ER-8, March 27.

5. Statens Naturvardsverk (Swedish Environmental Protection Agency), 1990. 1990 Major Review of Strategies and Policies for Air Pollution Abatement, August.

6. Bernow, S.B., and Marron, D.B., 1990. *Valuation of Environmental Externalities for Energy Planning and Operations*, May 1990 update, Tellus Institute.

7. See also the Appendix for a discussion of this problem (p. 1A-I-8).

8. U.S. EPA, 1990. *Health Effects Assessment Summary Tables - Third Quarter FY - 1990*, July.

9. *Ibid.*

10. *Ibid.*

11. Chernick, P., and Caverhill, E., 1991. Joint Testimony on Behalf of Boston Gas Company, Docket No. D.P.U. 91-131, Oct. 4.

12. Bernow, S.B., et al., 1991, *Valuation of Environmental Externalities for Electric Utility Resource Planning in Wisconsin*, prepared for Citizens for a Better Environment by Tellus Institute, November.

Table 1.1 Carcinogens: Potency Factors and Isophorone Equivalents

Pollutant	Cancer Potency	Isophorone Equivalents
Acenapthene		
Acenapthylene		
Acetone		
Acetophenone		
Acrylonitrile	5.40E-01	138
Aluminum		
Ammonia		
Anthracene		
Antimony		
Arsenic	5.00E+01	12821
Barium		
Benzene	2.90E-02	7
Benzo(a)anthracene		
Benzo(a)pyrene		
3,4-Benzofluoranthene		
Benzo(k)flouranthene		
Benzo(ghi)perylene		
Benzoic acid		
Beryllium	4.30E+00	1103
Biphenyl		
Bis(2-ethylhexyl) phthalate	1.40E-02	4
1,3-Butadiene	1.80E+00	462
2-Butanone		
Butyl benzyl phthalate		
Cadmium	6.10E+00	1564
Carbon disulfide		
Carbon tetrachloride	1.30E-01	33
Chlorine		
Chlorobenzene		
Chloroethane		
Chloroform	6.10E-03	2
p-Chloro-m-cresol		
2-Chlorophenol		
Chloroprene		
Chromium		
Chrysene		
Copper		
Coke oven emissions		
p-Cresol		
Cyanide		

Table 1.1 Carcinogens: Potency Factors and Isophorone Equivalents (cont.)

Pollutant	Cancer Potency	Isophorone Equivalents
2,4-D		
4,4-DDT	9.70E-06	2.49E-03
Dibenzo(a,h)anthracene		
1,4-Dichlorobenzene	2.40E-02	6
Dichlorobromomethane		
1,1-Dichloroethane		
1,2-Dichloroethane	9.10E-02	23
1,1-Dichloroethylene	6.00E-01	154
1,2-trans-dichloroethylene		
2,4-Dichlorophenol		
1,2-Dichloropropane	6.80E-02	17
1,3-Dichloropropene	1.80E-01	46
Diethyl phthalate		
2,4-Dimethylphenol		
Dimethyl phthalate		
Di-n-butyl phthalate		
4,6-Dintro-o-cresol		
2,4-Dinitrotoluene	6.80E-01	174
2,6-Dinitrotoluene	6.80E-01	174
1,2-Diphenylhydrazine	8.00E-01	205
Endosulfane sulfate		
Ethylbenzene		
Ethylchloride		
Ethylene oxide	3.50E-01	90
Fluoranthene		
Fluorene		
Fluoride		
Hexachlorobenzene	1.60E+00	410
2-Hexanone		
Hydrogen chloride		
Hydrogen fluoride		
Hydrogen sulfide		
Indeno(1,2,3-cd)pyrene		
Iron		
Isophorone	3.90E-03	1
Lindane		
Lead		
Magnesium		
Manganese		
Mercury		

Table 1.1 Carcinogens: Potency Factors and Isophorone Equivalents (cont.)

Pollutant	Cancer Potency	Isophorone Equivalents
Methane		
Methylene chloride	7.50E-03	2
4-Methyl-2-pentanone		
Napthalene		
Nickel	8.40E-01	215
Nitrobenzene		
PAHs	1.15E+01	2949
Parachloronitrocresol		
Pentachlorophenol		
Phenanthrene		
Phenol		
Propylene	2.40E-01	62
Pyrene		
Selenium		
Silver		
Sodium hydroxide		
Styrene	3.00E-02	8
Sulfides		
2,3,7,8-TCDD	1.50E+05	38461538
2,3,7,8-TCDF		
Tetrachloroethylene	5.10E-02	13
Thallium		
Thiocyanates		
Tin		
Toluene		
1,1,1-Trichloroethane	5.70E-02	15
Trichloroethylene	1.10E-02	3
Trichlorofluoromethane		
2,4,6-Trichlorophenol	1.10E-02	3
1,2,3-Trichloropropane		
Triethanol		
Vanadium		
Vinyl chloride	2.30E+00	590
Xylenes		
Zinc		

Table 1.2 Noncarcinogens: Reference Dose, 1/RD, and Xylene Equivalents

Pollutant	Reference Dose, oral	1/RD	Xylene equivalents
Acenapthene	6.00E-02	17	33
Acenapthylene			
Acetone	1.00E-01	10	20
Acetophenone	1.00E-01	10	20
Acrylonitrile			
Aluminum			
Ammonia	9.71E-01	1	2
Anthracene	3.00E-01	3	7
Antimony	4.00E-04	2500	5000
Arsenic	1.00E-03	1000	2000
Barium	5.00E-02	20	40
Benzene			
Benzo(a)anthracene			
Benzo(a)pyrene			
3,4-Benzofluoranthene			
Benzo(k)flouranthene			
Benzo(ghi)perylene			
Benzoic acid	4.00E+00	0.25	1
Beryllium	5.00E-03	200	400
Biphenyl	5.00E-02	20	40
Bis(2-ethylhexyl) phthalate	2.00E-02	50	100
1,3-Butadiene			
2-Butanone			
Butyl benzyl phthalate	2.00E-01	5	10
Cadmium	5.00E-04	2000	4000
Carbon disulfide	1.00E-01	10	20
Carbon tetrachloride	7.00E-04	1429	2857
Chlorine			
Chlorobenzene	2.00E-02	50	100
Chloroethane			
Chloroform	1.00E-02	100	200
p-Chloro-m-cresol	2.00E+00	0.5	1
2-Chlorophenol	5.00E-03	200	400
Chloroprene	2.00E-02	50	100
Chromium	1.00E+00	1	2
Chrysene			
Copper	3.71E-02	27	54
Coke oven emissions			
p-Cresol	5.00E-02	20	40
Cyanide	2.00E-02	50	100

Table 1.2 Noncarcinogens: Reference Dose, 1/RD, and Xylene Equivalents (cont.)

Pollutant	Reference Dose, oral	1/RD	Xylene equivalents
2,4-D	1.00E-02	100	200
4,4-DDT	5.00E-04	2000	4000
Dibenzo(a,h)anthracene			
1,4-Dichlorobenzene	7.00E-01	1	3
Dichlorobromomethane			
1,1-Dichloroethane	1.00E-01	10	20
1,2-Dichloroethane			
1,1-Dichloroethylene	9.00E-03	111	222
1,2-trans-dichloroethylene	2.00E-02	50	100
2,4-Dichlorophenol	3.00E-03	333	667
1,2-Dichloropropane			
1,3-Dichloropropene	3.00E-04	3333	6667
Diethyl phthalate	8.00E-01	1	3
2,4-Dimethylphenol	2.00E-02	50	100
Dimethyl phthalate	1.00E+00	1	2
Di-n-butyl phthalate			
4,6-Dinitro-o-cresol			
2,4-Dinitrotoluene			
2,6-Dinitrotoluene			
1,2-Diphenylhydrazine			
Endosulfane sulfate			
Ethylbenzene	1.00E-01	10	20
Ethylchloride			
Ethylene oxide			
Fluoranthene	4.00E-02	25	50
Fluorene	4.00E-02	25	50
Fluoride	6.00E-02	17	33
Hexachlorobenzene	8.00E-04	1250	2500
2-Hexanone			
Hydrogen chloride			
Hydrogen fluoride			
Hydrogen sulfide	3.00E-03	333	667
Indeno(1,2,3-cd)pyrene			
Iron			
Isophorone	2.00E-01	5	10
Lindane	3.00E-04	3333	6667
Lead	1.40E-03	714	1429
Magnesium			
Manganese	2.00E-01	5	10
Mercury	3.00E-04	3333	6667

Table 1.2 Noncarcinogens: Reference Dose, 1/RD, and Xylene Equivalents (cont.)

Pollutant	Reference Dose, oral	1/RD	Xylene equivalents
Methane			
Methylene chloride	6.00E-02	17	33
4-Methyl-2-pentanone			
Napthalene	4.00E-03	250	500
Nickel	2.00E-02	50	100
Nitrobenzene	5.00E-04	2000	4000
PAHs			
Parachloronitrocresol			
Pentachlorophenol	3.00E-02	33	67
Phenanthrene			
Phenol	6.00E-01	2	3
Propylene			
Pyrene	3.00E-02	33	67
Selenium	3.00E-03	333	667
Silver	3.00E-03	333	667
Sodium hydroxide			
Styrene	2.00E-01	5	10
Sulfides			
2,3,7,8-TCDD			
2,3,7,8-TCDF			
Tetrachloroethylene	1.00E-02	100	200
Thallium	7.00E-05	14286	28571
Thiocyanates			
Tin	6.00E-01	2	3
Toluene	3.00E-01	3	7
1,1,1-Trichloroethane	9.00E-02	11	22
Trichloroethylene			
Trichlorofluoromethane	3.00E-01	3	7
2,4,6-Trichlorophenol			
1,2,3-Trichloropropane	6.00E-03	167	333
Triethanol			
Vanadium	7.00E-03	143	286
Vinyl chloride			
Xylenes	2.00E+00	0.5	1
Zinc	2.00E-01	5	10

Table 1.3 Health Effects Ranking

Pollutant	Carcinogens Isophorone Equivalents	Noncarcinogens Xylene Equivalents	Combined Ranking [1]
Acenapthene		33	33
Acenapthylene			
Acetone		20	20
Acetophenone		20	20
Acrylonitrile	138		415
Aluminum			
Ammonia		2	2
Anthracene		7	7
Antimony		5000	5,000
Arsenic	12821	2000	20,231
Barium		40	40
Benzene	7		22
Benzo(a)anthracene			
Benzo(a)pyrene			
3,4-Benzofluoranthene			
Benzo(k)flouranthene			
Benzo(ghi)perylene			
Benzoic acid		0.5	0.5
Beryllium	1103	400	1,854
Biphenyl		40	40
Bis(2-ethylhexyl) phthalate	4	100	55
1,3-Butadiene	462		1,385
2-Butanone			
Butyl benzyl phthalate		10	10
Cadmium	1564	4000	4,346
Carbon disulfide		20	20
Carbon tetrachloride	33	2857	1,479
Chlorine			
Chlorobenzene		100	100
Chloroethane			
Chloroform	2	200	102
p-Chloro-m-cresol		1	1
2-Chlorophenol		400	400
Chloroprene		100	100
Chromium		2	2
Chrysene			
Copper		54	54
Coke oven emissions			
p-Cresol		40	40
Cyanide		100	100

Table 1.3 Health Effects Ranking (cont.)

Pollutant	Carcinogens Isophorone Equivalents	Noncarcinogens Xylene Equivalents	Combined Ranking [1]
2,4-D		200	200
4,4-DDT	2.49E-03	4000	2,000
Dibenzo(a,h)anthracene			
1,4-Dichlorobenzene	6	3	11
Dichlorobromomethane			
1,1-Dichloroethane		20	20
1,2-Dichloroethane	23		70
1,1-Dichloroethylene	154	222	342
1,2-trans-dichloroethylene		100	100
2,4-Dichlorophenol		667	667
1,2-Dichloropropane	17		52
1,3-Dichloropropene	46	6667	3,403
Diethyl phthalate		3	3
2,4-Dimethylphenol		100	100
Dimethyl phthalate		2	2
Di-n-butyl phthalate			
4,6-Dinitro-o-cresol			
2,4-Dinitrotoluene	174		523
2,6-Dinitrotoluene	174		523
1,2-Diphenylhydrazine	205		615
Endosulfane sulfate			
Ethylbenzene		20	20
Ethylchloride			
Ethylene oxide	90		269
Fluoranthene		50	50
Fluorene		50	50
Fluoride		33	33
Hexachlorobenzene	410	2500	1,865
2-Hexanone			
Hydrogen chloride			
Hydrogen fluoride			
Hydrogen sulfide		667	667
Indeno(1,2,3-cd)pyrene			
Iron			
Isophorone	1	10	7
Lindane		6667	6,667
Lead		1429	1,429
Magnesium			
Manganese		10	10
Mercury		6667	6,667

Table 1.3 Health Effects Ranking (cont.)

Pollutant	Carcinogens Isophorone Equivalents	Noncarcinogens Xylene Equivalents	Combined Ranking [1]
Methane			
Methylene chloride	2	33	20
4-Methyl-2-pentanone			
Napthalene		500	500
Nickel	215	100	373
Nitrobenzene		4000	4,000
PAHs	2949		8,846
Parachloronitrocresol			
Pentachlorophenol		67	67
Phenanthrene			
Phenol		3	3
Propylene	62		185
Pyrene		67	67
Selenium		667	667
Silver		667	667
Sodium hydroxide			
Styrene	8	10	17
Sulfides			
2,3,7,8-TCDD	38461538		115,384,615
2,3,7,8-TCDF			
Tetrachloroethylene	13	200	120
Thallium		28571	28,571
Thiocyanates			
Tin		3	3
Toluene		7	7
1,1,1-Trichloroethane	15	22	33
Trichloroethylene	3		8
Trichlorofluoromethane		7	7
2,4,6-Trichlorophenol	3		8
1,2,3-Trichloropropane		333	333
Triethanol			
Vanadium		286	286
Vinyl chloride	590		1,769
Xylenes		1	1
Zinc		10	10

Notes:
[1] The Combined Ranking assumes that 1 Isophorone Equivalent = 3 * Xylene Equivalent.

Table 1.4 Pollutant Prices

POLLUTANT	Pollutant Price ($/pound)
CO	$0.42
NOx	$3.63
Particulates	$5.85
SOx	$5.87
VOCs	$2.50
Acenapthene	$37
Acenapthylene	
Acetone	$22
Acetophenone	$22
Acrylonitrile	$465
Aluminum	
Ammonia	$2
Anthracene	$7
Antimony	$5,600
Arsenic	$22,658
Barium	$45
Benzene	$25
Benzo(a)anthracene	
Benzo(a)pyrene	
3,4-Benzofluoranthene	
Benzo(k)flouranthene	
Benzo(ghi)perylene	
Benzoic acid	$1
Beryllium	$2,076
Biphenyl	$45
Bis(2-ethylhexyl) phthalate	$62
1,3-Butadiene	$1,551
2-Butanone	
Butyl benzyl phthalate	$11
Cadmium	$4,868
Carbon disulfide	$22
Carbon tetrachloride	$1,656
Chlorine	$6
Chlorobenzene	$112
Chloroethane	
Chloroform	$115
p-Chloro-m-cresol	$1
2-Chlorophenol	$448

Table 1.4 Pollutant Prices (cont.)

POLLUTANT	Pollutant Price ($/pound)
Chloroprene	$112
Chromium	$2
Chrysene	
Copper	$60
Coke oven emissions	$25
p-Cresol	$45
Cyanide	$112
2,4-D	$224
4,4-DDT	$2,240
Dibenzo(a,h)anthracene	
1,4-Dichlorobenzene	$12
Dichlorobromomethane	
1,1-Dichloroethane	$22
1,2-Dichloroethane	$78
1,1-Dichloroethylene	$383
1,2-trans-dichloroethylene	$112
2,4-Dichlorophenol	$747
1,2-Dichloropropane	$59
1,3-Dichloropropane	$3,811
Diethyl phthalate	$3
2,4-Dimethylphenol	$112
Dimethyl phthalate	$2
Di-n-butyl phthalate	
4,6-Dinitro-o-cresol	
2,4-Dinitrotoluene	$566
2,6-Dinitrotoluene	$586
1,2-Diphenylhydrazine	$689
Endosulfane sulfate	
Ethylbenzene	$22
Ethylchloride	
Ethylene oxide	$302
Fluoranthene	$56
Fluorene	$56
Fluoride	$37
Hexachlorobenzene	$2,089
2-Hexanone	
Hydrogen chloride	$6
Hydrogen fluoride	
Hydrogen sulfide	$35
Indeno(1,2,3-cd)pyrene	
Iron	
Isophorone	$7
Lindane	$7,467

Table 1.4 Pollutant Prices (cont.)

POLLUTANT	Pollutant Price ($/pound)
Lead	$1,600
Magnesium	
Manganese	$11
Mercury	$7,467
Methane	$0.04
Methylene chloride	$22
4-Methyl-2-pentanone	
Napthalene	$560
Nickel	$418
Nitrobenzene	$4,480
PAHs	$9,908
Parachloronitrocresol	
Pentachlorophenol	$75
Phenanthrene	
Phenol	$4
Propylene	$207
Pyrene	$75
Selenium	$747
Silver	$747
Sodium hydroxide	
Styrene	$19
Sulfides	$35
2,3,7,8-TCDD	$129,230,769
2,3,7,8-TCDF	
Tetrachloroethylene	$134
Thallium	$32,000
Thiocyanates	
Tin	$4
Toluene	$7
1,1,1-Trichloroethane	$37
Trichloroethylene	$9
Trichlorofluoromethane	$7
2,4,6-Trichlorophenol	$9
1,2,3-Trichloropropane	$373
Triethanol	
Vanadium	$320
Vinyl chloride	$1,982
Xylenes	$1
Zinc	$11

APPENDIX I - EVALUATION OF ENVIRONMENTAL DAMAGE

There are essentially three methods by which environmental costs can be valued: the first is the damage cost approach, which attempts to trace the actual physical environmental impacts and to value the physical damage associated with them. The second approach, on which the overwhelmingly largest part of the economics literature focuses, attempts to elicit consumers' preferences, either directly, by presenting them with questionnaires, or indirectly, by observing consumers' behavior in the market. The third is the control or abatement cost approach. Below, we discuss each approach and present our choice of methodology.

It should be noted that the names which we have assigned to these three methods do not enjoy a consistent use in the literature: It seems that in the terminology of academic economics, the "direct valuation approach" refers to methods eliciting consumers' preferences by surveys and questionnaires (in particular: the Contingent Valuation Method), whereas the "indirect valuation approach" refers to the revealed preference approach (in particular, the Hedonic Property Price Method, the Hedonic Wage Method, and the Travel Cost Method). Some papers written in the context of public utility regulation use the term "direct valuation" to refer to the damage cost approach.[1]

The damage cost approach

When we speak about the environmental degradation caused by pollution, we have many specific impacts in mind: The contamination of drinking water with hazardous materials, which poses severe health threats to humans, animals and plants; the pollution of the air, which, apart from impacts on human health, causes damage to forests, crops, and buildings, and so forth. Many of these impacts cause a monetary cost to someone: patients and the public health system have to incur expenditures to treat diseases related to pollution, such as allergies and asthma; farmers are faced with the loss of crops, fishermen with the loss of catch, and so forth.

Of course, the damages caused by pollution far exceed these monetary losses: the general impairment of the quality of life, the physical and mental discomfort to people, the loss of natural environment which is not used commercially - all these do not normally receive a monetary valuation by the market. Many of these impacts are very hard or even impossible to evaluate objectively. What is a human life worth? The sum of its potential earnings? How to value the loss of a species, or of a habitat for rare species?

Even for those impacts which have direct monetary consequences, such as health expenditures and crop loss, it is a very complex endeavor to establish a quantitative causal relationship between the amount of pollutant emitted and the amount of damage caused. There are two approaches by which one could try to establish such a relationship, both of which are problematic: the "bottom-up" and the "top-down" approach. The bottom-up approach focuses on the different paths, spatial and temporal, that an individual pollutant takes from the point of emission to the contact with the medium to which it causes damages, evaluates the damages, and sums up the individual figures thus found. The top-down approach looks at the total emissions and the total damage, economy-wide.

The bottom-up approach

There are five stages that have to be studied in the attempt to trace the impact caused by a pollutant. These are the emission of the pollutant, the dispersal of the quantity emitted, the exposure of the medium to the pollutant, response of the medium (this is the physical damage caused) and valuation of the damage determined in the previous step. Each of these steps has to be quantified. It is probably straightforward to quantify the amount of pollutant emitted. The different paths a pollutant can take are, however, more difficult to determine. They depend on site-specific criteria and weather conditions. Exposure-Response assessments (also called "Dose-Response" studies) come to very different conclusions, because they cannot carry out controlled experiments. Many different factors contribute to the occurrence of particular diseases. It is difficult enough to relate the occurrence of, say, cancer, to the exposure to a specific amount of a pollutant in the laboratory. It is much harder to do so under conditions of an uncontrolled experiment (where other factors are not controlled for). This is not to say that we do not know that certain substances are highly carcinogenic. It is only to say that there is a great degree of uncertainty as to what the exact quantitative relationships are.

An additional source of uncertainty arises from the interaction of different pollutants. In combination, the impacts they cause are often more than the sum of the impacts they would cause in isolation.

These are only some of the difficulties posed by the damage cost approach. There are many more. We refer the interested reader to the literature.[2]

The top-down approach

This approach looks at the damage caused in the entire economy and tries to relate it to total emission (of one pollutant or a group of pollutants). While this provides a great simplification in that site specific factors do not need to be considered, many of the problems described for the bottom-up approach are present in exacerbated form: it is extremely difficult to isolate the influence of individual pollutants.

Because the physical processes which this approach attempts to capture are fraught with so much uncertainty, studies trying to assess and value the physical environmental damage of pollution have yielded very different inconsistent.[3]

Individuals' Preferences Approach, or: Direct and Indirect Monetary Valuation Methods

a) Philosophical Foundations of Welfare Economics

The largest part of the academic economics literature seeks to find values for the commodity "environment" by eliciting people's preferences, whether by asking a sample of the population directly (Direct Valuation Methods), or by observing people's actual behavior from which, it is thought, one can infer values which people put on environmental characteristics (Indirect Valuation Methods, or Revealed Preferences Approach).

To an outsider, this may seem a strange route to take. However, it is based on the central assumption of welfare economics (which provides the basis for valuation of the environment): that each person is the best judge of his or her own interests. Also, it is only the welfare of humans that is relevant. Fauna, flora and the inanimate world have no interests or intrinsic value; their only value lies in the enjoyment or utility they provide to humans. In other words, no end can be prescribed to society; there is no binding overall moral end which members of a society strive for.

For the purpose of analysis, economists have distinguished between different types of value that the environment can hold for individuals: These are use value, option value, and existence value. Use value is based on the utility which people derive from the "consumption" of the environment for recreational purposes, such as boating, fishing and other sportive activities. The option value is the use value in the presence of uncertainty: People may not consume the environment at present, but may want to do so in the future. To have the option for future use preserved is assumed to be valued by the consumers. Finally, the existence value is the value which people assign to the environment for "altruistic" reasons (it is interesting that economics calls this motive altruistic, when it is not directed at other humans); it is the utility which they derive from the knowledge of the existence of the environment.

b) Direct Valuation, or: Contingent Valuation Method (CVM)

The Contingent Valuation Method (CVM) assumes hypothetical (contingent) markets. In essence, it consists of experiments in which people are asked to express their valuation for a specific environmental commodity. These experiments can be designed as bidding games, they can consist of filling in questionnaires, and so forth.[4]

To render this approach valid, i.e. to allow that it actually measures what is claimed it measures, several assumptions have to be made, e.g. pertaining to the aggregability of individual preferences.[5] In addition, it is subject to many sources of bias.[6] There is e.g. the strategic bias: Since environmental quality is a public good (i.e. it exhibits jointness of supply, that means: once it is provided, people cannot be excluded from its consumption), people have an incentive to understate their preference (if they are held to pay), counting on the fact that other people will provide for the supply of the good. This is the free-rider problem. Then there are several sources of bias which stem from the fact that individuals are not perfectly rational. It has been observed that people respond to the starting value that is quoted to them (source for the "starting point bias"). Also, the question is whether the hypothetical markets correspond well enough to real markets.

It is also of crucial importance exactly which change in environmental conditions consumers are asked to evaluate: Two concepts are suggested in the literature: Willingness to Pay (WTP) and Willingness to Accept (WTA). Loosely speaking, the former is the amount of money that a consumer would be willing to spend to secure an environmental benefit, and the latter is the compensation that he would demand to accept an environmental cost. However, both concepts can be applied to one and the same change in environmental conditions. Consider e.g. a policy to clean up 90 % of sulfur oxides emissions: WTP then is the maximum amount of money an individual would give away to have 90 % of SO_x emissions abated, while maintaining his or her utility level; and WTA is the amount of money he or she would have to be given to accept the pollution while maintaining the utility level corresponding to the absence of 90 % of the present pollution. Clearly, the two concepts imply a different distribution of property rights. In the first setting, the pollution with SO_x is the reference case, and it is perceived that cleaning up will yield a benefit to the consumer. In the latter setting, a clean environment is the reference case, and pollution is seen to be a cost to the consumer. It seems that in the first case, the polluter is assumed to have a right to pollute, and in the latter case, the parties bearing the pollution have the right to a clean environment.[7]

It has been asserted that economic theory suggests that these two values do not differ much. However, this result is only true for very specific assumptions (which, so it has been argued, are plausible). Empirical studies assessing the magnitude of WTA versus WTP have consistently produced far greater amounts for WTA than for WTP. The estimates for WTA have often exceeded the ones for WTP by a factor of four.[8]

There has been an ongoing discussion about this apparent discrepancy. It was long known that the difference between the two magnitudes is the greater, the greater the income elasticity of demand is.[9] This makes sense intuitively: The WTP is obviously limited by an income constraint. People may care very much for the environment, but they may not be able to afford to spend much on it if their income is small. However, it is not only the price elasticity of income which influences the difference between WTP and WTA. Recently, the very interesting result has been derived that the difference between WTP and WTA depends also on the uniqueness of the good in question. The more unique an environmental good is, the more will WTA exceed WTP.[10] The large difference between WTA and WTP may then be taken to indicate that the uniqueness of environmental features is actually perceived as such by people. This provides a strong argument for conservation.

In this context, it is very interesting to note that federal regulations, in the assessment of damages in the context the Comprehensive Environmental Response, Compensation, and

Liability Act (CERCLA, 1980) actually barred the use of the WTA method.[11] Carson and Navarro state that "... It should be openly acknowledged that there is an important divergence between what Congress wanted to be measured - WTA including existence values - and what the Department of the Interior regulations eventually mandated should be measured - WTP excluding existence values. This divergence occurred because of the admitted difficulty by economists of measuring WTA and existence values, but it is a divergence which leads to an underestimate of damages which is likely to be significant."[12]

c) Indirect Valuation Methods, or Revealed Preference Approach: Hedonic Price Method (HPM) and Travel Cost Method (TCM)

Hedonic price methods: Hedonic property prices. The hedonic price method tries to identify surrogates for the nonexistent market for the environment. Markets which qualify as surrogate markets for the environment are those in which a private good is traded that may bear some relationship to the public environmental good. The notion underlying the concept of hedonic prices is that people derive utility from various attributes of a product. A product has many attributes, some of which can relate to the presence of a public good. A house, e.g., can have different features which individual consumers value differently: it can have a cellar or not, a loft, balconies, a garden, etc. Each of these features commands a price; however, this price is implicit: Individual features of a house are not sold separately. One attribute of the house is the environment in which it is located. In theory, one can construct demand functions that depend on these individual characteristics, and one can derive an implicit price for certain environmental features. That is to say, one can derive the amount of money consumers are willing to spend to obtain one more unit of q, the environmental quality feature (if q is air quality, then "one more unit of q" would refer to "one unit less of pollutant", where the "pollutant" could refer to an index of air pollution). One would expect to observe differentials in housing prices, depending on the quality of the specific environment they are located in.

The derivation of an implicit price for an environmental characteristic from an ideal type demand function is a rather straightforward calculation. To estimate these implicit prices from observable market data, however, requires some strong assumptions and is far from unproblematic: Apart from the usual assumptions about the structure of individual utility functions relating to aggregability, it has to be assumed that people have a wide enough array of choices to make their decision on the basis of all characteristics. This is obviously hardly ever the case. Often, one characteristic overrides all others; proximity to the place of work often takes this role. People mostly have not much choice over where they find work and thus move into an environment that they would not move to otherwise. Another problem is that it is not easy to find a sample with sufficient variation, i.e. enough houses which exhibit different characteristics. The specific environment of houses varies together with other factors, and it is very hard to isolate the influence of one variable when they vary together. And, as stated above, in the absence of a wide array of choices, people are likely to base their decision on other characteristics than only the specific environment.

Hedonic price method: Hedonic wages

The notion of a good embodying many characteristics implies that a job, too, has many characteristics, not only the wage that it pays. One important characteristic is the risk to the health and life of the worker. It is argued that workers will only accept a job with high risk when given a "compensating wage differential". The hedonic wage method consists of relating the size of wage differentials for various jobs to their different risk characteristics. From this relationship, the value which workers ascribe to their lives is inferred. One problem with this method is that it presupposes information about the job characteristics, on the part of the workers and on the part of the researcher. Workers often do not have sufficient information about the risks to health and life which they are exposed to at work. Also, unless a job implies exposure to specific pollutants it is not possible to establish the dislike which workers hold for a specific pollutant. There is also a problem of measurement here. Data on specific pollution at work are not readily available; data usually exist only on the consequences of hazards, such as accidents, morbidity, and mortality. The hedonic wage studies would be of more use, if of any, in damage cost studies, in that they could give an indication of the value which people ascribe to their lives.

The travel cost method

The travel cost method is employed to evaluate the recreational benefits which a specific area holds for consumers. The amount of time and money which people are spending to get to and spent in the area is supposed to indicate the use value which they ascribe to this area.

Apart from various technical problems,[13] the obvious flaw of this approach is that it only targets the value of an area for a very specific narrow use. Surely people value natural resources for more than the amenity. And again, there is no way in which this method would allow us to evaluate the contribution of a single pollutant to environmental degradation.

The control cost approach

This method enjoys increasing popularity in the attempt of utility companies to internalize the environmental cost of energy production.[14] Some states have actually adopted this approach to incorporate environmental cost of electricity production in their energy planning process.[15]

The control cost approach infers the cost that society attributes to pollution from the regulations that it imposes on itself. Complying with standards set for pollutant emission is costly - thus, there must be a perceived benefit to pollution abatement. Two concepts are central to this approach: The marginal cost of pollution abatement, and the marginal benefit of pollution abatement.

The **marginal cost of pollution abatement** is an increasing function of the amount of pollutant being controlled. This does not only imply that to abate more pollution costs more. (The latter would be expressed by a **total** cost function of pollution abatement rising with the amount of pollution being controlled.) Increasing marginal cost also implies that the unit cost of abatement, the cost of abatement per unit of pollutant, rises with more and more pollution being abated. This just reflects economic decision making. To remove the first unit of pollutant, one would choose the cheapest technology available. The most expensive technology would only be employed if the potential of cheaper technologies were exhausted, i.e. if as much pollution as possible were abated with cheaper technologies.

The **marginal benefit of pollutant abatement** is a decreasing function of the amount of pollutant being removed. This does not only mean that the overall benefit is greater, the more pollution is abated, but it also implies that the benefit per unit of pollutant removed is greater, the greater the overall level of pollution is. In other words: The benefit from preventing one more ton of SO_x to enter the atmosphere is smaller, the more SO_x has already been controlled. The negative side of this relationship is that the marginal damage function of pollution is generally increasing, that is, the damage that one unit of pollutant causes is greater, the higher the overall pollution levels. (The capacity of ecosystems to absorb pollution can reach critical points beyond which the damage increases drastically).

These functions may not be strictly monotonic, i.e. they may contain constant portions. It is for example, plausible that the first unit of pollutant (say, the first hundred thousand tons of SO_x) causes as much damage as the tenth, but less than the eleventh. This would imply that the marginal benefit of pollution abatement is approaching constancy after falling initially.

With this constellation of costs and benefits of pollution abatement, the optimal emission standard for a particular pollutant emission is that level of pollutant at which the marginal cost of abatement equals the marginal benefit of abatement. To set such a standard would constitute an efficient allocation of resources to the activity of pollution abatement. To do more would cost society more than the benefits which would result from the implementation of that standard.

The next step of the argument is somewhat of a leap of faith: It is assumed that the way in which regulatory standards are set are a) completely rational, and b) accurately represent society's preferences.

An ideal, rational public decision maker would set the emission standard for the pollutant at the optimal level. Thus, knowing what it costs to remove the last unit of the pollutant to satisfy the regulation, one knows the benefit accruing from removing this unit of pollutant. But the benefit of removing one unit of pollutant is equal to the cost its presence imposes on society (this is approximately true when we are not dealing with large amounts of pollutants).

Another way to depict this is as follows: the emission standard can be expressed as a linear function. The point of intersection of the emission standard with the marginal cost of abatement curve determines the marginal cost of removing that last unit of pollutant to meet the standard. Recall that it is important to get at the marginal cost of compliance: Which is the most expensive pollution control which is administered? This is the price society is willing to pay to have the last unit of pollutant controlled, thus, this is the value that society ascribes to the absence of that unit of pollutant.

Of course there are several problems with this approach. For one, existing legislation and regulations are not perfectly rational, nor do they perfectly reflect society's preferences. What are "society's preferences" anyway? We will deal with these ideological, normative issues later. First we turn to some problems which are more technical in nature.

First, there is no emission standard for each individual pollutant. Some pollutants are not regulated at all, and for others, not standards, but controls are administered.

The latter feature presents the problem of "joint cost of pollution control": Several pollutants causing very different environmental impacts can be captured with one and the same device. How should the cost of that device be allocated to individual pollutants? E.g., a smokestack scrubber may capture some amount of sulfur dioxide as well as some small amount of heavy metals. Does that imply that the cost of the scrubber will be "evenly" divided and ascribed to control costs of SO_x as well as cadmium? No. Recall that it is the marginal cost of control of a specific pollutant which provides the (negative) value of that pollutant to society. If of all the cadmium potentially released into the environment, smokestack scrubbers capture, say, 60 %, but there are other regulations addressing the remaining 40 %, then it is these regulations that are relevant; in effect, it is the regulation removing the "last" unit of cadmium which will provide the value that society places on cadmium removal from the environment. It will be the most stringent regulation, and the costliest to comply with.

In addition, we can only infer a value to that pollutant which the device is intended to capture, i.e. the pollutant to which the regulation is addressed, because it is this pollutant for which the regulation implies a certain value.

Another problem is that there may not exist regulations for all pollutants. A case in point is the emission of greenhouse gases. One could value the costs caused by these emissions through the costs of the measures which would offset the emission of the gases - e.g. afforestation. It seems also legitimate to assume that society holds consistent preferences, and that for some pollutants, regulations addressing different but similar ones can be used: For example, the banning of lead acid batteries from incinerators reveals the regulator's (representing society's) preference that heavy metals should not be emitted. It seems legitimate to assume a regulation banning other heavy metals products of similar toxicity from incinerators.

Rationale for Our Choice to Employ the Control Cost Approach

Clearly, the best method to value external costs is the damage cost approach. It corresponds most closely to what we understand environmental impacts to be. Although there are some damages that hold very different values to different people, there are still considerable costs that can, potentially, be valued objectively because they are costs that affect goods which are traded in the market. The estimates of these costs would establish a lower bound to the dollar value of the externality.

However, to undertake such an estimation is an extremely complex endeavor. Millions of dollars have been spent on studies, and their results are still loaded with much uncertainty. We clearly do not have the resources to engage in this kind of study for disposal fee analysis.

As to the approaches employed by academic economics, we feel too uncomfortable with the kind of assumptions that are required to lend them credibility. In addition, the data limitations and sources of bias have a too great potential to let the researcher miss the subtle relationships posited by theory. It is an approach that rests on highly technical and theoretical notions which, again, are plausible in the realm of economic theory but which may not be legitimate in the real world. Last, not least, they are hard to convey and thus hard to justify to a wider audience.

Thus, we have decided to adopt the Control Cost Approach. Two main considerations have guided our choice:

For one, the Control Cost Approach is the only approach which is feasible to employ and administer with the available resources. Any administrative body would be ill advised to adopt a method for evaluation of externalities which is costly, complex, and fraught with a lot of uncertainty. The control cost approach is being discussed by public utilities as a sensible compromise between what one would want to study and the limitation of resources. Also, Tellus Institute has developed some expertise with this approach. Several studies have been undertaken in-house that employ the control cost approach[16], and the state of Massachusetts has adopted the methodology suggested by Tellus.

The second reason is more normative in nature: We know that regulators are not perfectly rational; nor are they perfect representatives of society's preferences. "Society's preferences" are diverse - individual members of society may hold wildly different values, and very diverse interests are at stake. However, we have to ascribe legitimacy to the political process and assume that it will, with all its imperfections, attain some kind of consensus which is expressed in the regulations which society imposes on itself. Thus, although we may not believe that existing regulation always reflects a fair societal compromise[17], we do, with some qualifications, subscribe to its normative content.

ENDNOTES

1. However, the recent OECD (1989) study which surveys methods of externality evaluation employed by the academic economics profession uses these terms in yet a different way: "Direct valuation techniques" refers to all methods trying to elicit consumers' preferences, be it by contingent valuation or by revealed preferences, and "Indirect Valuation Procedures" refers to what we call the "damage cost approach". Then, later in the text, studies focussed on revealed preferences are referred to as "indirect market studies" (p.38).

   The following more technical terms are used with some consistency: Contingent Valuation Method (CVM), Revealed Preferences Approaches: Hedonic Price Method (HDM) (Hedonic Property Prices, Hedonic Wages), and Travel Cost Method (TCM).

2. An informative discussion of the complex issues arising with the damage cost approach can be found in Chernick and Caverhill (1989).

3. For a review of physical damage cost studies see Ottinger et al. (1990), Chapter V.

4. A brief but comprehensive list can be found in A. Myrick Freeman III (1982). The 1979 monograph by the same author is a classic in the field. For a more recent presentation, see Mitchell and Carson (1989).

5. Per-Olov Johansson (1979), p.52; Mitchell and Carson (1989), p.41.

6. A brief but informative discussion can be found in OECD, 1989, p. 36.

7. For a profound discussion of the history of these concepts, see Mitchell and Carson, p.30.

8. See e.g. OECD (1989), p.39.

9. A consumer's income elasticity of demand for a good is the relative change in his or her purchase of this good in response to a relative income change; in other words: the percentage change in the amount spent on the good, given a 1 % change in income.

10. Randall and Stoll (1980) have found that the difference between WTP and WTA is a function of a parameter which they call the "price flexibility of demand". Hanemann (1989) has identified this parameter as the ratio of an income elasticity divided by a substitution elasticity. If the denominator of this expression becomes small (and goes towards zero), the expression as a whole becomes large (and goes towards infinity). That implies that the more unique an environmental commodity

is, (i.e. the less close substitutes it has), the larger will be the difference between WTP and WTA.

11. Carson and Navarro (1988), p.817.

12. *Ibid.*, p. 830.

13. See e.g. OECD (1989), p.43.

14. See e.g. Chernick and Caverhill (1989), Tellus Institute (1990).

15. The state of Massachusetts has done so, upon a recommendation by Tellus Institute, Boston May (1989).

16. Bernow, S., and Marron, D., 1990. *The Treatment of Environmental Impacts in Electric Resource Evaluation: A Case Study in Vermont.* Tellus Institute, Boston, MA, January 22, 1990, and Bernow, Stephen et al.: *Incorporating Environmental and Economic Goals into Nevada's Energy Planning Process.* Tellus Institute, Boston, MA, July 30.

17. Also, by assuming consistency on the part of the regulator, we will paint a "regulation reference case" that is more stringent than the regulations which are at present in case. Also, we may take recourse to planned legislation and regulation, or policies advocated by large parts of the population.

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APPENDIX II - CHEMICAL HAZARD RANKING METHODS

To date, most cradle-to-grave assessments of packaging materials have documented the levels of various pollutants emitted during the stages of production and then summed emissions within an impact category - air, water, and industrial solid waste - to determine the total impact of production. These schemes inherently assume that one pound of a pollutant is no different than one pound of any other pollutant and that the impacts are therefore additive. Summing pollutants implicitly assumes for example, that one pound of sulfur dioxide has the same impact as one pound of benzene, two pollutants that have very different health effects. Simply summing pollutants does not account for these varying impacts; therefore, a methodology is required that ranks pollutants according to the relative harm that they cause.

One of the goals of Tellus Institute's packaging study is to develop a method that will enable us to incorporate these differing impacts in our analysis of the impacts associated with the production and disposal of packaging materials. We began this task by reviewing existing methods to evaluate the hazard posed by various pollutants. For example, risk assessment methodologies evaluate the hazard posed by different pollutants. The framework for evaluation includes quantifying the release of each pollutant into an environmental media (i.e., air, water, soil), predicting the fate and transportation of the pollutant both within and between media, analyzing the pathways by which humans and other organisms will be exposed to the pollutant, and analyzing the health effect of the pollutant.

Simpler methods to evaluate the hazard posed by a mixture of various chemical pollutants also exist. The majority of these methods have been developed to assist in evaluating and prioritizing chemicals that may require further study or regulation. These methods typically look at a wide range of factors for each chemical including indicators of human health, ecological impacts, yearly production quantities, and potential for release into the environment. Each of these factors is then scored independently, yielding a scoring matrix. Interpreting the matrix can be difficult as it requires judgement, or valuation, of the importance of each factor. The bibliography at the end of this appendix provides a list of such scoring systems.

An example of this scoring methodology is the chemical scoring system developed by EPA Office of Toxic Substances and Oak Ridge National Laboratory (O'Bryan and Ross, 1988). Eleven parameters are scored, several of which pertain to toxicity. These parameters include: oncogenicity, genotoxicity, developmental toxicity, lethal acute toxicity, nonlethal acute toxicity, subchronic/chronic toxicity, aquatic toxicity, bioconcentration, chemical production volume, occupational exposure, consumer exposure, and environmental exposure. Each parameter is scored independently, with scores for most parameters ranging from 0 to 9. Scores are not added, weighted or combined in any manner. Thus, the end result of this methodology is a "scoring profile" consisting of nine individually ranked parameters. Such a system does not enhance comparison of chemicals. Different chemicals can have very different scoring profiles. To decide which chemical is "worse" the user of such a system would have to decide which parameters are more important (i.e., should be weighted more).

Other ranking systems have been developed that solely rely upon indicators of toxicity. A methodology developed by US EPA Office of Health and Environmental Assessment ranks carcinogenic and chronic toxic chemicals based solely upon toxicological information. To evaluate carcinogens, the level of carcinogenic evidence along with the potency factor of the carcinogen is considered. The International Agency for Research on Cancer (IARC) has developed a numerical classification scheme which is used for evaluating strength of carcinogenic evidence. The potency factor in this report is defined as $1/ED_{10}$, where ED_{10} is the dose associated with a lifetime cancer risk of 10%. Potency factors are then grouped into four levels and a matrix is constructed using the potency factor grouping and IARC grouping. This matrix ranks chemicals as posing a high, medium, or low cancer hazard. Therefore, this evaluation is only qualitative.

The evaluation of chronic toxicity in this ranking system relies upon two factors - minimum effective dose (MED), the smallest chronic dose at which a toxic effect is noted, and type of chronic toxic effect. MED values are scored from 1 to 10 with 10 indicating high toxicity. A range of 1 to 10 is also assigned to each chemical based upon severity of chronic effect with 10 established as the most severe effect, death or shortening of lifespan. A composite score is next generated from the product of the MED value and chronic effect value. This composite score does not provide an ordinal ranking.

The two scoring systems reviewed above typify the range of existing systems. These systems assemble acute and chronic toxicity data for chemical substances and use these data to categorize the substances into groups indicative of low to high toxicity. Other factors such as potential for release of the chemical into the environment, environmental persistence, and ability to bioconcentrate can also be ranked. Thus, the final ranking of chemical substances is actually more a qualitative ranking than quantitative ranking.

A quantitative ranking system is more informative than a qualitative ranking as it allows a greater degree of comparison between chemical substances. A quantitative ranking enables a relative comparison between substances to determine how much better or worse one substance is than another. For example, the health effects ranking system presented in this report allows us to conclude that cadmium, which has a combined ranking score of 4,346 (see Table 1.3), is 3 times worse than lead which has a combined ranking score of 1,429. Therefore, due to this advantage of a quantitative ranking system, we chose to use the system presented in this report.

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CHAPTER 2 - METHODOLOGY FOR DETERMINING ENVIRONMENTAL IMPACTS OF PACKAGING MATERIALS DISPOSAL

In a previous report entitled, *The Marginal Cost of Handling Packaging Materials in the New Jersey Solid Waste System*, the conventional cost of handling packaging materials in New Jersey's solid waste system was assessed. In this chapter the environmental cost of packaging disposal is assessed in order to determine the total cost of handling packaging materials in the solid waste system.

The environmental impacts of solid waste and recycling collection and the impacts of three MSW management facilities are examined in this chapter including:

1. MSW landfill;
2. MSW incinerator; and
3. Materials recovery (recycling).

Air emissions and water effluent are quantified for the three types of facilities and air emissions are quantified for garbage and recycling collection of the materials. Environmental impacts are quantified for each packaging material disposed of at each facility type. The method of allocating facility environmental impacts to each material is discussed below. In general, environmental impacts for each material will be based on total pollutant loadings from the facility type and the amount of each material disposed of or recycled. Thus, each material will have per ton pollutant factors for each facility type. Pollutants associated with each material will depend on the specific material and the type of facility.

2.1 LANDFILLS

Allocating environmental impacts to specific materials in landfills is difficult because numerous factors contribute to landfill leachate and gas generation. Data which link environmental impacts directly to specific materials are extremely limited. Therefore, the environmental impacts of each material is derived from total landfill pollutant loadings and the New Jersey waste stream composition. If it is known that a material does not contribute to leachate or gas generation, then no environmental impacts are attributed to it. In estimating pollutant factors for landfill leachate and gas emissions it is important to note that there is a delay between deposition and the generation of leachate and gas.

2.1.1 Landfill Leachate Generation

The purpose of this section is to develop leachate pollution factors for each packaging material disposed of in a MSW landfill. To do this, it is necessary to determine the amount of leachate generated, the composition of that leachate, and the materials contributing to that leachate in New Jersey landfills. This report examines a generic, controlled New Jersey landfill which complies with current New Jersey landfill regulations. Assessing leachate generation, leachate composition, and materials contributing to leachate in New Jersey landfills, however, is difficult without extensive field work. Therefore, in this section leachate generation, composition, and contributors are estimated using the best available data.

Degradation of materials in a landfill is a slow process, as evidenced by reports of the presence in landfills of still-readable newspapers several decades old. However, the goal of this study is not to assess the timing of landfill leachate impacts, but rather to determine how specific waste components influence leachate characteristics.

The development of leachate pollution factors for each packaging material disposed of in a MSW landfill was a four step process:

1. identify the amount of leachate (in gallons) generated in New Jersey landfills;
2. identify the concentration of pollutant (ppm or pounds/gallon) in leachate;
3. convert pollutant concentration to pollutant factors (pounds pollutant/ton MSW); and
4. allocate pounds of pollutant to materials according to composition analysis of the materials and the percentage of that material in the New Jersey waste stream.

First, quantifying leachate generation requires knowledge of the geology, hydrogeology, precipitation, climate, field capacity (water-holding capacity of landfill materials), cover permeability, landfill slope, and cover material of a landfill. To estimate the quantity of leachate generated by a generic, controlled New Jersey landfill, which is in compliance with regulations, the water balance model developed by the U.S. EPA for landfills was used: "HELP" (the "Hydrologic Evaluation of Landfill Performance" model).[1]

Using HELP and making the assumptions outlined in Table 2.1, annual controlled leachate (landfill with a liner and leachate collection system) generation for a generic New Jersey landfill is estimated to be 0.0147 gallons per ton of waste over the lifetime of the landfill. Landfill lifetime includes 25 active years and 30 post-closure years. These leachate generation rates are yearly estimates based on calculations using the HELP model.

Second, to identify pollutant concentrations in leachate it is necessary to have test data on landfill leachate. National data on leachate composition was used as these data are from a published report and represent a large sample size. Listed in Tables 2.2 and 2.3 are leachate composition concentrations (pounds of pollutant/gallon leachate) for inorganic pollutants and organic pollutants.

In the third step the pollutant concentrations were multiplied by the gallons of leachate per ton of waste to arrive at total pollutant loadings per ton of MSW (pounds of pollutants/ton MSW) for controlled landfills (see Table 2.4). In the fourth step these pollutants were allocated to specific materials depending on composition analysis (based upon the ultimate analysis - Table 2.5), reactivity of the material in a landfill, and percentage of the material in the waste stream (See Table 2.6).

All heavy metals -- with the exception of cadmium, lead, mercury,* nickel, and zinc -- were allocated to specific materials based on the ultimate analysis done for eight metals in Table 2.5 and/or percentage of the material landfilled. Cadmium, lead, mercury, nickel, and zinc were allocated to specific materials after large fractions of their total contribution to landfill leachate was allocated to household hazardous wastes. Household hazardous wastes account for 52% of all cadmium,[2] 13% of all lead,[3] 93% of all mercury,[4] 20% of all nickel,[5] and 45% of all zinc.[6] Therefore, for example, 52 percent of all cadmium is apportioned to household hazardous wastes and the remaining 48 percent is allocated across all other materials containing cadmium, based upon the ultimate analysis (see Table 2.5). The household hazardous waste which accounts for the high use of these metals is batteries.[7]

Organic pollutants were allocated to specific materials based on their reactivity in a landfill, contribution to the specific pollutant, and/or percentage of the material in the waste stream. Glass, metals, miscellaneous inorganics, and inert solids do not contribute to organic pollutants from landfills. All pesticides -- 2,4-D; 4,4-DDT; endosulfane sulfate; and lindane -- and 1,1-dichloroethane (used only as a solvent) were allocated to household hazardous wastes. Yard waste may also be a small contributor to pesticides in landfill leachate, but in this report we assumed that household hazardous wastes were responsible for all pesticides found in landfill leachate. All other organics, with the exception of phenol, were allocated to paper, plastics, tires/rubber, miscellaneous organics (non-compostable organics), and household hazardous wastes based on their percentage in the waste stream. Phenols were the only organic chemicals attributed to organic wastes.

The results of allocating pollutants to various waste components is shown in Table 2.7. For ease of presentation, only those waste components that include packaging materials are shown.

2.1.2 Landfill Gas Generation

Landfill gas is produced primarily by the anaerobic decomposition of organic materials. Factors which affect landfill gas generation include: landfill temperature, aeration, moisture content, pH, and waste composition.[8] Because numerous factors affect landfill gas generation -- similar to leachate generation -- it is difficult to allocate specific pollutants in landfill gas to specific materials. This is further complicated by the fact that some gases are a by-product of reactions which occur in the landfills. In addition, discussion

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* Mercury was not identified as a constituent of leachate from landfills meeting RCRA requirements.

with staff of NJ Department of Environmental Protection revealed that the materials under consideration in this study are not major contributors to landfill gas production.[9] Therefore, landfill gas impacts were not determined in this study.

2.2 MSW INCINERATORS

In this section, the emissions associated with solid waste incineration - air emissions and solid waste (i.e, ash) - are presented. The emissions are shown in Table 2.8

2.2.1 Air Emissions Associated with Solid Waste Incineration

As with the combustion of any material, the combustion of solid waste generates air emissions. The quantity of a pollutant released into the air per unit of solid waste incinerated depends on the efficiency of the pollution control device (and on combustion efficiency). We developed emission factors (expressed as pound of pollutant per ton MSW incinerated) representative of newer mass-burn incinerators assuming the facilities are equipped with a scrubber, fabric filter baghouse, and Thermal $DeNO_x$ for air pollution control (see Table 2.8). The next step is apportioning pollutant emissions to waste stream components. The methodology is described below.

Table 2.5 provides an ultimate analysis of various waste stream components. Emissions of the metals documented in this table - arsenic, barium, cadmium, chromium, lead, mercury, selenium, and silver - were apportioned according to the weighted average (i.e., the amount of metal that each component contributes to the total amount of metal) of these metals in the waste stream components (see Table 2.5). Batteries contribute several metals (lead, cadmium, mercury, nickel and zinc) to the solid waste stream. The percentages of these metals contributed by batteries were apportioned to household hazardous waste (see Section 2.1.1); the remaining percentages were apportioned according to the ultimate analysis. As no information could be found about waste stream sources of antimony, nickel, tin, and vanadium, emissions of these metals were apportioned evenly to each waste stream component. Wastes classified simply as "metals" are responsible for 30% of the copper in the waste stream and miscellaneous inorganics are responsible for another 40%. Thus, 30% of the copper emissions were attributed to the metals category, 40% was attributed to miscellaneous inorganics, and the remaining 30% was attributed to the remaining waste stream components.

Criteria air emissions include carbon monoxide, nitrogen oxides, particulates, sulfur oxides, and VOCs. Carbon monoxide and particulates emissions are related to the amount of waste incinerated; therefore these emissions were apportioned evenly across waste stream components. Sulfur oxide emissions were attributed to waste stream components containing sulfur according to the weighted average sulfur content. Emissions of VOCs arise from the combustible portion of the waste stream. These emissions were evenly apportioned to those categories.

Nitrogen oxide emissions can arise from two sources - nitrogen present in the MSW and nitrogen present in the combustion air. A search of the literature did not elucidate which of these two pathways are predominant. Therefore, we assumed 50% of the emissions arise from nitrogen in the fuel. Correspondingly, 50% of the emissions were attributed to waste stream components containing nitrogen, in proportion to their contributions to the total weight of nitrogen in the waste stream. Turning to the other source, nitrogen which is present in the combustion air is also converted to nitrogen oxides. As the amount of combustion air is proportional to the amount of solid waste incinerated, the remaining 50% of the emissions was attributed evenly to each waste stream component.

Hydrogen chloride and hydrogen fluoride are associated with the presence of chlorine and fluorine in waste stream components. Hydrogen chloride emissions were allocated based upon the content of chlorine in each waste stream component. Sources of fluorine in waste include plastic, teflon coated metals, and floor and wall panel facings.[10] Hydrogen fluoride emissions were therefore apportioned to plastics, metals, bulky items, and textiles.

PAHs and PCDDs/PCDFs are generated from precursors in the waste stream. Precursors for PAH formation include paper, plastics, organics, and household hazardous waste. Emissions of PAHs were apportioned evenly among these components. The chemistry behind PCDD/PCDF formation is somewhat more complex. Precursors are formed from aromatic organics and chlorine sources. Emissions were apportioned evenly among waste components that yield aromatic organic compounds during combustion and chlorine including paper, plastics, yard waste, wood waste, rubber, textiles, miscellaneous inorganics (which contain chlorine) and household hazardous waste.

In Table 2.9, the pounds of pollutants per ton of packaging material are presented.

2.2.2 Leachate Associated with Solid Waste Incinerator Ash

Incineration of MSW generates two types of solid waste: bottom ash, which is the residue formed from the combustion of MSW; and fly ash, which is the particulates formed in the furnace and carried with the flue gases that then enter and are collected in the air pollution control devices. By weight, the resulting ash is 25-35% of the incoming waste. Thus, for example, a 1,000 ton per day incinerator will generate 250 to 350 tons per day of ash. Approximately 90% of this ash is bottom ash and 10% is fly ash.

Bottom ash and fly ash differ markedly in their compositions. Bottom ash is large inert, incombustible residue. It mainly consists of large particles of broken glass, metals, ceramics, and any other heavy incombustible residues which are not removed by the incinerator flue gas. The metals in the bottom ash are usually visible and recoverable. The remaining incombustible portion is granular in nature and usually has lower metal concentrations than those found in fly ash.

Fly ash, on the other hand, consists of lighter particulates. The particle size of the fly ash is much finer than that of the bottom ash. Metal concentrations in the fly ash are generally higher than those of the bottom ash. This can be attributed to the fly ash's relatively small particle size and hence proportionately larger surface area.

While on a weight basis bottom ash exceeds fly ash, it is the fly ash that contains most of the environmental contaminants of concern. In the air pollution control device, volatile contaminants including heavy metals, dioxins, dibenzofurans, and other organics condense on these particulates.

The main environmental concern associated with ash is its impacts when landfilled. Once landfilled, pollutants adhered to the particulates can be released into leachate. A recent study measured pollutant concentrations in leachate from five ash landfills.[11] The results of that study, summarized in Table 2.8, show that the amount of pollutants entering the environment from ash is minor as compared to air emissions.

2.3 MATERIALS RECOVERY

The impacts of materials recovery facilities arise primarily from the air emissions of mechanical machinery and particulates from processing operations, such as glass crushers. Most of the emissions are minimal and local in nature, representing more of an in-facility issue than external environmental problem.

Only one source of data was found for recycling facility air emissions. In a report by the Center for the Biology of Natural Systems (CBNS), *Development and Pilot Test of an Intensive Municipal Solid Waste Recycling System for the Town of East Hampton*, results are presented from a two day sampling of a materials recovery facility in Groton, Connecticut. This facility accepts mixed glass, aluminum, and tin containers which are separated and processed through a combination of manual and automatic systems.

The data from this report are not necessarily representative of all recycling facilities. In addition, the CBNS study is an analysis of *ambient* conditions, not actual emission rates, because measurements were made from the exhaust fan in the building. Converting ambient measurements to environmental emissions is not a straightforward process and further reduces the suitability of the data to this project.

Pollutant levels measured in the Groton, Connecticut facility are listed in Tables 2.10 and 2.11. The tables present data collected both during active sorting and when no sorting occurred. While emissions differed for organics and microorganisms during sorting, no change in the particulate emissions and heavy metals was found. Consequently, particulates and heavy metals are only listed in Table 2.10, which reports emissions when no sorting occurred.

Emissions within a recycling facility can emanate from a number of sources: emissions from collection vehicles; the unloading of materials on the tipping floor; emissions from front-end loaders; particulates from glass crushers and other sorting and processing equipment; and emissions from automated sorting and processing machinery. Some of these emissions result from sorting activities, though many occur from other activities, such as the dumping of materials. Therefore, particulates or heavy metals in the air may result from these other activities even though they appear to result from sorting.

Seven materials were responsible for the majority of volatile organic compounds (VOCs) detected: silicone oil (a lubricant); isobutane (used in liquid petroleum gas fuel); trichlorofluoromethane (a refrigerant); 1,1,1-trichloroethane (a cleaning solvent); acetone (an industrial solvent); toluene (an industrial solvent); ethyl benzene (a component of gasoline); and xylene (a solvent and component of gasoline). Several organic compounds increased in concentration during active sorting. Of these materials, only isobutane showed a clear increase during both sampling days. A number of other materials showed increased concentrations only during the first day of sampling, when 65% of sampling was performed. These include trichloroethane, trichloroethylene, benzene, tetrachloroethylene, and possibly hexane and toluene. The average concentration of VOCs measured is about 3,256 ug/m^3. This, however, is likely a minimum concentration. During the two days of air testing at the facility, the sampling device became saturated. As a result, not all of the pollutants were adsorbed on the collecting material used in the sampling devices and the concentrations may therefore be underestimated.

While the test results are shown in Tables 2.10 and 2.11, they have not been incorporated into the disposal impacts associated with packaging materials, due to the collection and sampling problems which surround the results.

2.4 SOLID WASTE AND RECYCLABLE COLLECTION

Air emissions resulting from the collection of garbage and recyclables are considered in this section; water effluents, of course, are not generated by trucks. The principal data source for transportation air emission factors is the U.S. EPA report, *Compilation of Air Pollutant Emission Factors II: Mobile Sources*. Additional information was obtained from studies published by the California Air Resources Board.[12,13] Air emissions factors were found for HC, CO, NO_x total VOCs, benzene, ethyl benzene, toluene, and xylenes.

Emission factors in the U.S. EPA *Compilation* report are based upon pounds of pollutants emitted per ton-mile. These factors are available for HC, CO, NO_x and total VOCs; they are converted to a volume-measure (i.e., pollutants per cubic yard-mile) that is based upon standard recycling and garbage truck capacity. The volume-based measure is more appropriate for assessing collection impacts because materials fill up trucks by volume, not by weight.

Emissions factors for benzene, ethyl benzene, toluene and xylenes are based on the percentage composition of all individual VOCs emitted from diesel exhaust.

Air impacts are estimated for each material, based upon in-truck volumes (different for recycling and garbage trucks because of compaction) and assumed truck collection miles. Some adjustment to emission factors is required because these factors assume normal transport of goods at high average speeds, whereas waste collection involves large amounts of idle time spent collecting materials, combined with higher average speed during transport to the waste facility.

Per ton emission levels from recycling and garbage collection vehicles are presented in Table 2.12. From these figures, emissions per ton for each material are determined. These results are presented in Table 2.13 for recycling collection and in Table 2.14 for garbage collection. Data on HC, CO, NO_x total VOCs, benzene, ethyl benzene, toluene, and xylenes are shown in these tables. The emission factors have been converted to emission levels per ton based upon the following assumptions:

3.0 pounds waste generated per person per day;
2.6 people per household;
15% of material recycled by weight;
a recycling collection rate of 80 households per hour; and
a garbage collection rate of 60 households per hour.

The assumed recycling rate does not necessarily reflect the current New Jersey recycling rate as the purpose of determining emissions associated with recyclables collection is not to determine the total emissions associated with collection of New Jersey's recyclable materials, but rather to determine the *per-ton* emissions associated with collecting one ton of each material.

Emissions from garbage and recycling trucks are assumed to be identical. However, in reality, emissions are higher from garbage trucks due to compaction cycles and slightly larger engine requirements. For both trucks, emission factors for "heavy duty diesel vehicles" are used from the *Compilation of Air Pollutant Emission Factors*. These emission factors assume operating conditions of standard trucks transporting goods, not of waste collection vehicles with frequent stops, starts, and compaction cycles. To account for this, emission factors for idle time are used, and adjustments for slower traveling speed are made to the standard emission factors.

2.5 ENVIRONMENTAL DISPOSAL COSTS

In Table 2.15 pollutants per ton of material associated with garbage collection and recyclables collection are multiplied by the pollutant prices to yield environmental dollar cost per ton of packaging material.

For environmental costs of disposal, we multiplied the physical emissions of materials in the disposal facilities by the pollutant prices to obtain a dollar value for each pollutant per ton of material. These individual emission costs are then summed to determine the total environmental cost per ton of each material type in a given facility.

Table 2.15 calculates the weighted environmental cost per ton of packaging material. This result was obtained from the costs for each material at a given facility and weighted according to the percent of packaging material handled at each facility. For example, 18% of old corrugated cardboard (OCC) is recycled, 41% is landfilled, and 41% is incinerated. Thus, 18% of the total environmental cost for OCC is the cost associated with recycling, 41% of the total cost comes from the OCC landfill cost, and 41% is from the OCC incineration cost. This calculation was performed for each packaging material. The quantity of packaging material handled by each facility was taken from our previous report, *The Marginal Cost of Handling Packaging Materials in the New Jersey Solid Waste Stream*. The percentage originally attributed to transfer stations in the aforementioned report is distributed such that 85% of the material is sent to landfills and 15% is sent to incinerators.

2.6 ENDNOTES FOR CHAPTER 2

1. U.S. EPA, 1984. *Hydrologic Evaluation of Landfill (HELP) Model, Volumes I and II* Cincinnati, Ohio.

2. Franklin Associates, 1989. *Characterization of Products Containing Lead and Cadmium in Municipal Solid Waste in the United States, 1970 to 2000*, Prairie Village, KS.

3. *Ibid.*, p. 8.

4. Rosseaux, P., Navarro, A., and Vermande, P., 1989. "Heavy metal distribution in household waste," *BioCycle*, September, p. 83.

5. *Ibid.*

6. *Ibid.*

7. Franklin Assoc., *op. cit.*

8. Senior, E. and Kasali, G.B., 1990. "Landfill Gas," in *Microbiology of Landfill Sites*, edited by Eric Senior, CRC Press: Boca Raton, Florida, p. 119.

9. Conversation with Mike Winka, NJ DEPE, August, 1991.

10. Draft Environmental Impact Report for the East Bridgewater (MA) Integrated Waste Disposal System, Volume I: Impact Report, prepared by CSI, 1990.

11. US EPA, 1990. *Characterization of Municipal Waste Combustion Ash, Ash Extracts, and Leachates*, prepared by NUS Corp., EPA 530-SW-90-029A, March.

12. California Air Resources Board, 1989. *Technical Guidance Document to the Criteria and Guidelines Regulation for AB-2588* [Air Toxics "Hot Spots" Information and Assessment Act of 1987], August.

13. California Air Resources Board, 1989. *Identification of Volatile Organic Compound Species Profiles*, ARB Speciation Manual, August.

Table 2.1 New Jersey Landfill Assumptions and Leachate Generation for Controlled Landfill

Leachate generation (cu ft/ yr)	228
Acres	50
Square Feet	2,178,000
Depth (feet)	70
Volume (cubic ft)	152,460,000
MSW density (tons/cu yd)	0.5
MSW density (tons/cu ft)	0.02
Total delivered weight (tons)	2,896,740
Years Open	25
Tons delivered/year	115,870
Leachate generated (cu ft/ton MSW)	1.97E-03
Leachate generated (gal/ton MSW)	1.47E-02
Leachate generated (lbs/ton MSW) [1]	1.23E-01

Notes:
[1] Leachate generated (gal/ton MSW) * 8.3453 lbs/gal.

Sources: U.S. EPA, "HELP Model," 1984; and Tellus Institute.
Bruce Witkowski, Engineering Bureau, New Jersey DEP.

Table 2.2 MSW Leachate Constituents-- Inorganic Pollutants -- from Landfills Meeting RCRA Requirements

Inorganic Pollutant	Average Concentration (ppm)	Average Concentration (lbs/gal)	Median Concentration (ppm)	Maximum Concentration (ppm)
Aluminum	2.04E+00	1.70E-05	2.00E+00	5.80E+00
Antimony	n/d	n/d	n/d	n/d
Arsenic	1.10E-02	9.18E-08	9.00E-03	2.30E-02
Barium	7.31E-01	6.10E-06	4.80E-01	1.70E+00
Beryllium	n/d	n/d	n/d	n/d
Cadmium	2.20E-03	1.84E-08	2.00E-03	7.00E-03
Chromium (total)	8.30E-03	6.93E-08	6.25E-03	3.90E-02
Cobalt	n/d	n/d	n/d	n/d
Copper	n/d	n/d	n/d	n/d
Iron	8.01E+01	6.69E-04	1.94E+01	2.68E+02
Lead	1.70E-02	1.42E-07	3.00E-03	6.10E-02
Manganese	3.06E+00	2.56E-05	1.21E+02	4.24E+02
Magnesium	2.00E+02	1.67E-03	1.51E+00	8.87E+00
Mercury	n/d	n/d	n/d	n/d
Nickel	6.79E-02	5.67E-07	6.50E-02	1.60E-01
Selenium	8.60E-04	7.18E-09	n/d	6.00E-03
Silver	n/d	n/d	n/d	n/d
Thallium	n/d	n/d	n/d	n/d
Tin	n/d	n/d	n/d	n/d
Vanadium	1.60E-02	1.34E-07	1.70E-02	2.40E-02
Zinc	7.29E-01	6.08E-06	3.15E-01	2.59E+00

Source: U.S. EPA, "Characterization of Leachates from Municipal Waste Disposal Sites and Co-Disposal Sites," 1987, pp. 4-8 - 4-10.

Table 2.3 MSW Leachate Constituents -- Organic Pollutants -- from Landfills Meeting RCRA Requirements

Organic Pollutant	Average Concentration (ppm)	Average Concentration (lbs/gal)	Median Concentration (ppm)	Maximum Concentration (ppm)
Acetone	1.97E+00	1.64E-05	1.13E+00	4.60E+00
2-Butanone [1]	3.56E+00	2.97E-05	9.70E-01	1.20E+01
p-Cresol [2]	1.33E+00	1.11E-05	2.65E-02	5.10E+00
2,4-D	1.49E-02	1.25E-07	n/d	1.20E-01
4,4-DDT	7.36E-05	6.14E-10	5.30E-05	1.60E-04
1,1-Dichloroethane	2.90E-04	2.42E-09	n/d	4.00E-03
t-1,2-Dichloroethene	2.43E-03	2.03E-08	n/d	1.60E-02
Diethyl phthalate	2.29E-03	1.91E-08	n/d	3.20E-02
Endrin	n/d	n/d	n/d	n/d
Endosulfan sulfate	2.00E-05	1.67E-10	n/d	2.80E-01
Ethyl benzene	n/d	n/d	n/d	n/d
bis(2-Ethylhexyl)phthalate	2.60E-03	2.17E-08	n/d	1.90E-02
2-Hexanone [3]	1.69E-01	1.41E-06	2.55E-02	6.90E-01
Lindane	2.90E-06	2.42E-11	n/d	2.30E-05
4-Methyl-2-pentanone [4]	7.36E-02	6.14E-07	n/d	5.70E-01
Methylene chloride	1.12E-01	9.35E-07	9.70E-01	3.60E-01
Phenol	5.12E-01	4.27E-06	2.05E-02	2.10E+00
1,1,3-Trichloropropane	1.64E-02	1.37E-07	n/d	2.30E-01
Toluene	3.06E-01	2.55E-06	n/d	1.10E+00
Xylenes, total	n/d	n/d	n/d	n/d

[1] = also known as methyl ethyl ketone
[2] = also known as 4-methyl phenol
[3] = also known as methyl butyl ketone
[4] = also known as methyl isobutyl ketone

Source: U.S. EPA, "Characterization of Leachates from Municipal Waste Disposal Sites and Co-Disposal Sites," 1987, pp. 4-14 - 4-15.

Table 2.4 Pounds of Pollutants Per Ton MSW from a Controlled Landfill

INORGANIC POLLUTANTS	Lbs Pollutants/ Ton MSW
Aluminum	2.51E-07
Antimony	n/d
Arsenic	1.35E-09
Barium	8.98E-08
Beryllium	n/d
Cadmium	2.70E-10
Chromium (total)	1.02E-09
Cobalt	n/d
Copper	n/d
Iron	9.84E-06
Lead	2.09E-09
Manganese	3.76E-07
Magnesium	2.46E-05
Mercury	n/d
Nickel	8.34E-09
Selenium	1.06E-10
Silver	n/d
Thallium	n/d
Tin	n/d
Vanadium	1.97E-09
Zinc	8.95E-08
ORGANIC POLLUTANTS	
Acetone	2.41E-07
2-Butanone	4.37E-07
p-Cresol	1.63E-07
2,4-D	1.83E-09
4,4-DDT	9.04E-12
1,1-Dichloroethane	3.56E-11
trans-1,2-Dichloroethylene	2.98E-10
Diethyl phthalate	2.81E-10
Endrin	n/d
Endosulfane sulfate	2.46E-12
Ethyl benzene	n/d
bis(2-Ethylhexyl)phthalate	3.19E-10
2-Hexanone	2.08E-08
Lindane	3.56E-13
4-Methyl-2-pentanone	9.04E-09
Methylene chloride	1.38E-08
Phenol	6.29E-08
Toluene	2.01E-09
1,2,3-Trichloropropane	3.76E-08
Xylenes	n/d

Note: n/d = not detected

TABLE 2.5 Ultimate Analysis for Residential MSW

Parameter	Unit	Paper	Plastics	Organics [1]	Wood	Textiles	Rubber	Ceramics	Glass	Metal	Inorganics
Arsenic	ppm	3.80	3.10	10.53	3.97	7.60	4.57	4.00	5.07	31.47	2.30
Barium	ppm	27.07	41.27	110.40	34.80	24.13	20.80	113.80	108.87	24.87	73.00
Cadmium	ppm	4.83	1.69	5.84	0.78	1.94	1.52	0.90	0.89	1.33	1.38
Chromium	ppm	8.82	17.85	34.37	7.46	395.17	66.63	12.92	120.93	45.91	38.72
Lead	ppm	28.80	58.57	532.43	72.17	15.03	16.37	767.33	32.40	2066.83	384.67
Mercury	ppm	0.68	0.74	0.63	0.66	0.47	0.41	0.06	0.05	0.13	1.14
Selenium	ppm	7.17	1.83	1.87	1.47	4.40	41.40	1.37	1.40	1.70	8.37
Silver	ppm	0.88	0.93	0.77	0.97	1.50	0.51	0.87	0.47	0.77	0.97
Carbon	%	34.57	45.17	19.40	42.73	46.27	37.87	6.63	1.15		12.77
Hydrogen	%	7.30	7.60	7.47	6.10	6.00	4.57	1.73	1.13		2.70
Nitrogen	%	0.22	0.11	0.47	0.51	2.38	0.24	0.15	0.15		1.74
Oxygen	%	50.80	49.53	56.00	48.80	42.43	16.83	0.00	0.00		14.73
Sulfur	%	0.12	0.15	0.34	0.06	0.16	0.55	0.05	0.08		2.03
Chlorine	%	0.30	1.26	0.15	0.13	0.37	2.79	0.06	0.07		2.30

[1] Brush, grass, food waste, and miscellaneous organic wastes.

Source: SCS Engineers, "NYC Solid Waste 'Ultimate Analysis,'" 1990.

Table 2.6 New Jersey Waste Composition

Paper	
Newspaper	12.00%
Old Corrugated Cardboard	5.00%
Other	28.00%
Plastics	
HDPE	2.00%
PET	0.55%
Film	1.80%
Other	6.00%
Glass	6.71%
Metals	
Aluminum	1.00%
Other Metals	3.00%

Source: Tellus Institute, "The Marginal Cost of Handling Packaging Materials in the New Jersey Solid Waste System", p.22.

Table 2.7 Controlled Landfill Emission Factors (pounds/ton)

	PAPER	PLASTICS	GLASS	METALS
INORGANIC POLLUTANTS				
Arsenic	8.18E-10	6.67E-10	1.09E-09	6.77E-09
Barium	4.56E-08	6.95E-08	1.83E-07	4.19E-08
Cadmium	1.61E-10	5.62E-11	2.96E-11	4.42E-11
Chromium (total)	3.61E-10	7.30E-10	4.95E-09	1.88E-09
Lead	2.18E-10	4.42E-10	2.45E-10	1.56E-08
Manganese	3.76E-07	3.76E-07	3.76E-07	3.76E-07
Nickel	6.67E-09	6.67E-09	6.67E-09	6.67E-09
Selenium	1.06E-10	1.06E-10	1.06E-10	1.06E-10
Vanadium	1.97E-09	1.97E-09	1.97E-09	1.97E-09
Zinc	4.92E-08	4.92E-08	4.92E-08	4.92E-08
ORGANIC POLLUTANTS				
Acetone	3.94E-07	3.94E-07		
2-Butanone (methyl ethyl ketone)	7.13E-07	7.13E-07		
p-Cresol	2.66E-07	2.66E-07		
trans-1,2-Dichloroethylene	4.87E-10	4.87E-10		
Diethyl phthalate	4.58E-10	4.58E-10		
bis(2-Ethylhexyl)phthalate	5.21E-10	5.21E-10		
4-Methyl-2-pentanone (methyl isobutyl ketone)	1.47E-08	1.47E-08		
Methylene chloride	2.24E-08	2.24E-08		
Phenol	7.37E-08	7.37E-08		
Toluene	6.13E-08	6.13E-08		
1,2,3-Trichloropropane	3.28E-09	3.28E-09		

Table 2.8 Emission Factors for Solid Waste Incinerators (lbs/ton MSW)

	Air Emissions	Leachate	Total
CRITERIA AIR POLLUTANTS			
CO	5.33E-01		5.33E-01
NOx	1.88E+00		1.88E+00
Particulates	1.32E-01		1.32E-01
SOx	1.15E-01		1.15E-01
VOCs	4.36E-02		4.36E-02
INORGANIC POLLUTANTS			
Antimony	1.16E-05		1.16E-05
Arsenic	3.60E-06	6.59E-10	3.60E-06
Beryllium	6.55E-07		6.55E-07
Cadmium	1.31E-05	3.68E-11	1.31E-05
Chromium (total)	1.52E-05	4.72E-11	1.52E-05
Copper	2.08E-04	5.04E-11	2.08E-04
Lead	1.33E-04	1.41E-10	1.33E-04
Manganese	3.60E-03	1.15E-07	3.60E-03
Mercury	1.76E-03		1.76E-03
Nickel	3.49E-05		3.49E-05
Selenium	9.90E-06	5.36E-10	9.90E-06
Tin	1.15E-04		1.15E-04
Vanadium	1.48E-06		1.48E-06
Zinc	5.08E-04	8.54E-10	5.08E-04
ORGANIC POLLUTANTS			
PAHs (total)	1.71E-04		1.71E-04
PCDD/PCDF (total)	1.91E-09	5.90E-16	1.91E-09
MISCELLANEOUS			
Hydrogen chloride	1.41E-01		1.41E-01
Hydrogen fluoride	6.93E-04		6.93E-04

Sources: Tellus Institute 1991, "Disposal Cost Fee Study", February.
US EPA, 1990. "Characterization of Municipal Waste Combustion Ash, Ash Extracts, and Leachates, March.

Table 2.9 Incineration Emission Factors (pounds/ton)

	PAPER	PLASTICS	hdpe	pet	pvc	other plastic	GLASS	METALS
CRITERIA AIR POLLUTANTS								
CO	8.88E-02	8.88E-02	8.88E-02	8.88E-02	8.88E-02	8.88E-02	8.88E-02	8.88E-02
NOx	2.47E-01	2.14E-01	2.14E-01	2.14E-01	2.14E-01	2.14E-01	2.01E-01	1.57E-01
Particulates	2.20E-02	2.20E-02	2.20E-02	2.20E-02	2.20E-02	2.20E-02	2.20E-02	2.20E-02
SOx	4.98E-03	5.30E-03	5.30E-03	5.30E-03	5.30E-03	5.30E-03	3.64E-03	
VOCs	1.09E-02	1.09E-02	1.09E-02	1.09E-02	1.09E-02	1.09E-02		
INORGANIC POLLUTANTS								
Antimony	1.94E-06	1.94E-06	1.94E-06	1.94E-06	1.94E-06	1.94E-06	1.94E-06	1.94E-06
Arsenic	2.62E-07	8.71E-08	8.71E-08	8.71E-08	8.71E-08	8.71E-08	8.76E-08	5.66E-07
Beryllium	1.09E-07	1.09E-07	1.09E-07	1.09E-07	1.09E-07	1.09E-07	1.09E-07	1.09E-07
Cadmium	7.10E-07	1.63E-06	1.63E-06	1.63E-06	1.63E-06	1.63E-06	2.50E-07	2.19E-07
Chromium (total)	1.45E-07	1.27E-07	1.27E-07	1.27E-07	1.27E-07	1.27E-07	9.58E-07	6.00E-06
Copper	9.37E-06	9.37E-06	9.37E-06	9.37E-06	9.37E-06	9.37E-06	9.37E-06	6.25E-05
Lead	1.37E-06	1.41E-06	1.41E-06	1.41E-06	1.41E-06	1.41E-06	1.25E-05	4.59E-05
Manganese	5.99E-04	5.99E-04	5.99E-04	5.99E-04	5.99E-04	5.99E-04	5.99E-04	5.99E-04
Mercury	9.78E-06	8.85E-06	8.85E-06	8.65E-06	8.85E-06	8.85E-06	3.83E-06	7.05E-06
Nickel	4.65E-06	4.65E-06	4.65E-06	4.65E-06	4.65E-06	4.65E-06	4.65E-06	4.65E-06
Selenium	1.10E-06	3.57E-07	3.57E-07	3.57E-07	3.57E-07	3.57E-07	2.71E-07	6.31E-07
Tin	1.92E-05	1.92E-05	1.92E-05	1.92E-05	1.92E-05	1.92E-05	1.9165E-05	1.92E-05
Vanadium	2.46E-07	2.46E-07	2.46E-07	2.46E-07	2.46E-07	2.46E-07	2.4645E-07	2.46E-07
Zinc	4.65E-05	4.65E-05	4.65E-05	4.65E-05	4.65E-05	4.65E-05	4.6536E-05	4.65E-05
ORGANIC POLLUTANTS								
PAHs (total)	3.42E-05	3.42E-05	3.42E-05	3.42E-05	3.42E-05	3.42E-05		
PCDD/PCDF (total)	2.39E-10	2.39E-10			1.19E-10	1.19E-10		
MISCELLANEOUS								
Hydrogen chloride	5.65E-03	2.08E-02			1.04E-02	1.04E-02	8.39E-03	
Hydrogen fluoride		1.73E-04	1.73E-04	1.73E-04	1.73E-04	1.73E-04		1.73E-04

Table 2.10 Recycling Facility Environmental Impacts - Inactive (No Sorting)

	Emission Concentration (ug/CM)	Emission Rate (lb/hr)	Emissions per Ton (lb)
Particulates (1)	3,000,000	0.064	0.0128
Heavy Metals (1)			
Cadmium	0.4	8.53E-09	1.71E-09
Chromium	1.4	2.99E-08	5.97E-09
Lead	2.3	4.91E-08	9.81E-09
Nickel	4.95	1.06E-07	2.11E-08
Arsenic	n/d		
Mercury	0.23	4.91E-09	9.81E-10
Organics (2)			
Acetone	125	2.67E-06	5.33E-07
Benzene	5	1.07E-07	2.13E-08
Carbon disulfide	5	1.07E-07	2.13E-08
Carene	16	3.41E-07	6.83E-08
Chloroform	2	4.27E-08	8.53E-09
Cyclohexane	5	1.07E-07	2.13E-08
Diethyl Ether	5	1.07E-07	2.13E-08
Ethyl Acetate	10	2.13E-07	4.27E-08
Ethyl Benzene	29	6.19E-07	1.24E-07
Hexane	4	8.53E-08	1.71E-08
Isobutane	116	2.47E-06	4.95E-07
Methyl chloride	39	8.32E-07	1.66E-07
Mehtyl Cyclohexane	2	4.27E-08	8.53E-09
Methylpentane	3	6.40E-08	1.28E-08
Silicone Oil	252	5.38E-06	1.08E-06
Pentane	5	1.07E-07	2.13E-08
Tetrachloroethylene	17	3.63E-07	7.25E-08
1,1,1-trichloroethane	44	9.39E-07	1.88E-07
Trichloroflouromethane	143	3.05E-06	6.10E-07
Toluene	88	1.88E-06	3.75E-07
Xylenes	707	1.51E-05	3.02E-06

	Bacterial Colonies/CM	Colonies/HR	Colonies/ton
Air Microorganisms			
Inactivity	125	1,209	242

(1) Active sorting appears to have no noticable impact on emission levels.
(2) For most pollutants, represents underestimate since detectors were oversaturated.

Source: Center for the Biology of Natural Systems. "Development and Pilot Test of an Intensive Municipal Solid Waste Recycling System for the Town of East Hampton: Volume I", prepared for the New York State Energy Research and Development Authority. February 1990.

Table 2.11 Recycling Facility Environmental Impacts - Active Sorting

	Emission Concentration (ug/CM)	Emission Rate (lb/hr)	Emissions per Ton (lb)
Organics (2)			
Acetone	137	2.92E-06	5.85E-07
Benzene	7	1.49E-07	2.99E-08
Carbon disulfide	5	1.07E-07	2.13E-08
Carene	2	4.27E-08	8.53E-09
Chloroform	2	4.27E-08	8.53E-09
Cyclohexane	5	1.07E-07	2.13E-08
Diethyl Ether	5	1.07E-07	2.13E-08
Ethyl Acetate	22	4.69E-07	9.39E-08
Ethyl Benzene	14	2.99E-07	5.97E-08
Hexane	7	1.49E-07	2.99E-08
Isobutane	1500	3.20E-05	6.40E-06
Methyl chloride	47	1.00E-06	2.01E-07
Mehtyl Cyclohexane	3	6.40E-08	1.28E-08
Methylpentane	3	6.40E-08	1.28E-08
Silicone Oil	467	9.96E-06	1.99E-06
Pentane	31	6.61E-07	1.32E-07
Tetrachloroethylene	12	2.56E-07	5.12E-08
1,1,1-trichloroethane	103	2.20E-06	4.39E-07
Trichloroflouromethane	142	3.03E-06	6.06E-07
Toluene	98	2.09E-06	4.18E-07
Xylenes	91	1.94E-06	3.88E-07

	Bacterial Colonies/CM	Colonies/HR	Colonies/ton
Air Microorganisms	3700	35,797	7,159

(2) For most pollutants, represents underestimate since detectors were oversaturated.

Source: Center for the Biology of Natural Systems. "Development and Pilot Test of an Intensive Municipal Solid Waste Recycling System for the Town of East Hampton: Volume I", prepared for the New York State Energy Research and Development Authority. February 1990.

Table 2.12 **Air Emissions from Recycling and Garbage Collection**

	Recycling Collection		Garbage Collection	
	Emission Factor (g/hour)	Emissions (lb/ton)	Emission Factor (g/hour)	Emissions (lb/ton)
CRITERIA AIR POLLUTANTS				
CO	102.12	6.87E-01	102.12	1.62E-01
NOx	144.49	9.73E-01	144.49	2.29E-01
SOx	20.64	1.39E-01	20.64	3.27E-02
VOCs	34.71	2.34E-01	34.71	5.50E-02
ORGANIC POLLUTANTS				
Benzene		4.18E-03		9.80E-04
Ethyl Benzene		1.40E-04		3.00E-05
Toluene		4.20E-03		9.90E-04
Xylenes		1.50E-03		3.50E-04
Collection Rate (tons/hr)		3.28E-01		1.39E+00

Assumes 3.0 lb/person/day, 2.6 people/household, 15% recycled, 80 households/hour for recycling and 60 hh/hr for garbage.

Source: U.S. EPA, "Compilation of Air Pollutant Emission Factors, Volume II: Mobile Sources", Fourth edition, September 1985.

Table 2.13 Recycling Collection Emission Factors (pounds/ton)

	Old Corrugated Cardboard	Mixed Paper	Other Paper	HDPE	PET	Film	Other Plastic
CRITERIA AIR POLLUTANTS							
CO	3.94E-01	5.92E-01	8.87E-01	2.54E+00	2.96E+00	3.55E+00	2.54E+00
NOx	5.55E-01	8.32E-01	1.25E+00	3.57E+00	4.16E+00	4.99E+00	3.57E+00
SOx	7.92E-02	1.19E-01	1.78E-01	5.09E-01	5.94E-01	7.13E-01	5.09E-01
VOCs	1.39E-01	2.09E-01	3.13E-01	8.95E-01	1.04E+00	1.25E+00	8.95E-01
ORGANIC POLLUTANTS							
Benzene	2.49E-03	3.74E-03	5.61E-03	1.60E-02	1.87E-02	2.24E-02	1.60E-02
Ethyl benzene	8.35E-05	1.25E-04	1.88E-04	5.37E-04	6.26E-04	7.52E-04	5.37E-04
Toluene	2.51E-03	3.76E-03	5.64E-03	1.61E-02	1.88E-02	2.26E-02	1.61E-02
Xylenes	8.91E-04	1.34E-03	2.00E-03	5.73E-03	6.68E-03	8.02E-03	5.73E-03

Table 2.13 Recycling Collection Emission Factors (pounds/ton) (cont.)

	Recycled Glass	Non-recycled Glass	Aluminum Cans	Ferrous
CRITERIA AIR POLLUTANTS				
CO	1.48E-01	1.48E-01	1.48E+00	4.44E-01
NOx	2.08E-01	2.08E-01	2.08E+00	6.24E-01
SOx	2.97E-02	2.97E-02	2.97E-01	8.91E-02
VOCs	5.22E-02	5.22E-02	5.22E-01	1.57E-01
ORGANIC POLLUTANTS				
Benzene	9.35E-04	9.35E-04	9.35E-03	2.80E-03
Ethyl benzene	3.13E-05	3.13E-05	3.13E-04	9.40E-05
Toluene	9.40E-04	9.40E-04	9.40E-03	2.82E-03
Xylenes	3.34E-04	3.34E-04	3.34E-03	1.00E-03

Table 2.14 Garbage Collection Emission Factors (pounds/ton)

POLLUTANTS	Old Corrugated Cardboard	Mixed Paper	Other Paper	HDPE	PET	Film	Other Plastic
CRITERIA AIR POLLUTANTS							
CO	1.55E-01	1.30E-01	1.41E-01	3.28E-01	3.35E-01	1.91E-01	3.66E-01
NOx	2.18E-01	1.83E-01	1.98E-01	4.62E-01	4.71E-01	2.68E-01	5.14E-01
SOx	3.12E-02	2.61E-02	2.83E-02	6.60E-02	6.73E-02	3.83E-02	7.35E-02
VOCs	5.48E-02	4.59E-02	4.97E-02	1.16E-01	1.18E-01	6.74E-02	1.29E-01
ORGANIC POLLUTANTS							
Benzene	9.80E-04	8.22E-04	8.90E-04	2.07E-03	2.12E-03	1.21E-03	2.31E-03
Ethyl benzene	3.29E-05	2.75E-05	2.98E-05	6.95E-05	7.09E-05	4.04E-05	7.75E-05
Toluene	9.86E-04	8.26E-04	8.95E-04	2.09E-03	2.13E-03	1.21E-03	2.32E-03
Xylenes	3.50E-04	2.94E-04	3.18E-04	7.42E-04	7.57E-04	4.31E-04	8.26E-04

Table 2.14 Garbage Collection Emission Factors (pounds/ton) (cont.)

POLLUTANTS	Recycled Glass	Non-recycled Glass	Aluminum Cans	Ferrous
CRITERIA AIR POLLUTANTS				
CO	3.59E-02	3.59E-02	3.90E-01	1.54E-01
NOx	5.05E-02	5.05E-02	5.49E-01	2.16E-01
SOx	7.21E-03	7.21E-03	7.84E-02	3.09E-02
VOCs	1.27E-02	1.27E-02	1.38E-01	5.43E-02
ORGANIC POLLUTANTS				
Benzene	2.27E-04	2.27E-04	2.46E-03	9.72E-04
Ethyl benzene	7.60E-06	7.60E-06	8.26E-05	3.26E-05
Toluene	2.28E-04	2.28E-04	2.48E-03	9.78E-04
Xylenes	8.11E-05	8.11E-05	8.81E-04	3.48E-04

Table 2.15 Environmental Impacts of Disposal: Cost Per Ton of Material

Material	Recycling Collection	% [1]	Controlled Landfill Collection	Leachate	Total LF	% [1]	Incineration Collection	Air Emissions	Total Incin	% [1]
Paper										
OCC [2]	$3.07	18%	$1.21	$0.00	$1.21	41%	$1.21	$1.63	$2.84	41%
Other Paper	$6.92		$1.10	$0.00	$1.10	50%	$1.10	$1.63	$2.73	51%
Plastic										
HDPE	$19.77	4%	$2.56	$0.00	$2.56	47%	$2.56	$1.44	$4.00	49%
PET	$23.06	8%	$2.61	$0.00	$2.61	45%	$2.61	$1.44	$4.05	48%
PVC	$19.77		$2.85	$0.00	$2.85	49%	$2.85	$1.52	$4.37	51%
Other Plastic	$19.77		$2.85	$0.00	$2.85	49%	$2.85	$1.52	$4.37	51%
Glass	$1.15	41%	$0.28	$0.00	$0.28	26%	$0.28	$1.04	$1.32	34%
Metals										
Aluminum cans	$11.53	25%	$3.04	$0.00	$3.04	34%	$3.04	$0.90	$3.94	40%
Ferrous	$3.46	18%	$1.20	$0.00	$1.20	38%	$1.20	$0.90	$2.10	44%

Notes:

[1] These values represent the % of each material handled at each facility and are taken from "The Marginal Cost of Handling Packaging Materials in the New Jersey Solid Waste System", p75, Table 24. The percentage handled by transfer stations in the above mentioned report was distributed here so that 85% of the material handled by transfer stations goes to landfills and 15% goes to incinerators).

[2] Old corrugated cardboard.

Table 2.15 Environmental Impacts of Disposal: Cost Per Ton of Material (cont.)

Material	Weighted Average ($/ton material)
Paper	
OCC [2]	$2.22
Other Paper	$1.94
Plastic	
HDPE	$3.96
PET	$4.97
PVC	$3.62
Other Plastic	$3.62
Glass	$0.99
Metals	
Aluminum cans	$5.49
Ferrous	$2.01

CHAPTER 3 - ASSESSING THE FULL COST OF PACKAGING PRODUCTION AND DISPOSAL

In this chapter, we quantify the full external cost of packaging production and disposal, with the disposal costs based upon solid waste management in New Jersey. The full external cost consists of all costs of production and disposal, other than the current market cost of production, i.e. all costs which are "external" to the decisions of packaging buyers and sellers. There are two principal categories of external costs: 1) the environmental cost of production as discussed in this chapter and summarized for each packaging material in Table 3.1; and 2) the conventional and environmental costs of disposal as presented in Chapter 2 and summarized in Table 3.2. Table 3.3 compares the total environmental production cost and the total environmental disposal cost per unit weight of each material. These costs are also compared in Chapter 4 on a per package basis.

3.1 PRODUCTION COSTS

3.1.1 Methodology

To determine the environmental cost of producing packaging materials, we used the pollutant prices presented in Chapter 1 and the quantity of pollutant emitted per ton of packaging as presented in a previous report in this study, *Inventory of Material and Energy Use and Air and Water Emissions from the Production of Packaging Materials*. In the previous report, both the uncontrolled and controlled emissions associated with each stage of packaging material production were determined. In this report we have only used the controlled emission factors. Each of these pollutant emission factors (expressed as pounds of pollutant/ton of packaging material) was multiplied by the pollutant price (expressed as dollars/pound of pollutant) and summed to yield the cost per ton of material produced.

The environmental cost associated with industrial solid waste has not been included in the production cost. Unlike air and water emissions, solid waste is not regulated on the basis of the quantity produced. Rather, its disposal is regulated only after it is generated - and then only if it is classified as hazardous, or it is disposed of off-site. Large quantities of non-hazardous wastes, disposed of on-site, are not regulated, and therefore are not consistently reported. As a result, generation rates are difficult to determine.

We did explore various options to incorporate industrial solid waste into the inventory. An EPA document, *Solid Waste Disposal in the United States*,[1] reports total quantity of industrial solid waste by major industries (2-digit SIC codes) for 1985. Information at the 2-digit SIC code level is too aggregate to be used in this study where we have examined industries at the 4-digit SIC level. The Toxic Release Inventory (TRI) database provides the quantity of toxic chemicals disposed of on-site and off-site by manufacturers and users of these chemicals on a plant-by-plant basis. These data are quantified as the weight of pollutant released by a source each year. Without corresponding facility production data, the environmental release cannot be expressed as weight of pollutant per unit weight of product. A study correlating TRI pollutant emissions with production data could potentially yield the necessary unit emission factors (which could then be handled in the same manner as other emission factors covered in this report). However, such a study would be a massive undertaking, far beyond the scope of the current project.

In order for industrial solid waste data to be analogous to air and water emissions, we are not actually interested in the amount of solid waste generated. Industrial solid waste generation is captured in the price of a good, like the conventional costs for air and water pollution control. Rather, we are interested in "controlled" emissions associated with solid waste - the amount that escapes treatment. Thus, we would need information on the disposal path for industrial solid waste, and the environmental impacts of the disposal path, for example, leachate from landfills. Such data would be difficult to determine.

While we were unable to quantify industrial solid waste generation, our data search was not futile - it shows the need for establishing a database for solid waste generation that is comparable to data available for air and water emissions.

3.1.2 Costs Associated with Packaging Production

The environmental cost to produce one ton of each packaging material is summarized in Table 3.1. These prices reflect the environmental cost of producing one ton of each packaging material. Alternatively, these prices can be interpreted as the amount of packaging material that has an associated dollar value of environmental impacts from production. For example, bleached kraft paperboard has an environmental production cost of $330 per ton of material. Thus, 2.6 ounces of bleached kraft paperboard production is associated with an environmental cost of $1. As the amount of each material needed to make a package differs, comparison at this stage can be misleading. (See Chapter 4 for translation into per package costs.) In this chapter, we examine the pollutants that dominate the cost for each material and compare the production and disposal costs for materials.

The environmental production costs in Table 3.1 are presented in two categories. The first column is the cost attributed to emissions of criteria air pollutants, methane, chlorine, and hydrogen chloride. The price calculations for these pollutants are independent of the health effects ranking system. (As discussed in Chapter 1, the control costs for these pollutants were inferred from regulations that specifically address these pollutants.) The second column is the cost attributed to emissions of toxic and carcinogenic compounds; these pollutants are priced using the health effects ranking system. The last column of the table is the percent of the total environmental production cost attributed to toxic and carcinogenic pollutants. Table 3.1 shows that, for plastics, the toxic and carcinogenic compounds comprise over one-third the total environmental production cost. For paper packaging materials, these compounds are responsible for less than one-third of the total environmental production cost. Both virgin and recycled steel production costs are dominated by toxic and carcinogenic pollutants. Both virgin and recycled glass and aluminum production costs are dominated by the criteria air pollutants, methane, chlorine, and hydrogen chloride.

For glass, recycling eliminates the need for raw material acquisition and handling. As a result the criteria air pollutants associated with this stage are eliminated, thereby decreasing the cost attributed to these pollutants. In addition, as the use of cullet decreases the amount of energy required for glass production, emissions of criteria pollutants associated with this category are also decreased.

Recycling of aluminum provides an 80% decrease in the environmental production cost attributed to criteria air pollutants and a much larger decrease - 99% - in the toxic and carcinogenic cost category. Recycling of aluminum eliminates the need for raw material mining and two processes that emit pollutants - alumina production (an intermediate product), and aluminum production. In addition, the production of aluminum is an energy intensive operation, resulting in high emissions attributable to process energy use. Thus emissions are lower both due to the lower amount of energy needed to recycle aluminum as opposed to virgin production, and due to the fact that melting aluminum for recycling emits less pollutants (especially in the toxic/carcinogenic category) than virgin production.

While the environmental production cost for recycled aluminum and glass are lower than the virgin production costs, corrugating medium made from wastepaper has a higher production cost than that from virgin material. The recycling of wastepaper results in water pollution due to inks and coatings found on wastepaper. Trace metals such as cadmium, zinc, aluminum, and chromium are used in some inks. These materials are liberated in the pulping process and can therefore be found in the water effluent. In addition, the production of virgin corrugating medium has the lowest environmental cost of all virgin paper materials; as a result, even though recycled corrugating medium has a lower environmental cost than many other virgin paper materials, its cost is still higher than its virgin material counterpart.

Recycled steel has a slightly lower environmental production cost than virgin steel. Recycled steel has traditionally been an essential raw material of the steel making process. In basic oxygen furnaces (BOF), 20%-35% scrap is technically required to be added to the hot metal from the blast furnace. While much of this requirement is met by scrap generated at the steel mill, this home scrap can be supplemented with steel can scrap. Up to 40% scrap can be used in the BOF. While theoretically all of the scrap used in the BOF could be from steel cans, this scrap would not displace the current use of in-house scrap. Assuming 28% scrap is used in "virgin" production, then only 12% steel can scrap can then be used in "recycled" production. This additional 12% does not provide a large off-set of virgin steel production impacts. In addition, steel cans are detinned prior to use in the BOF; this step causes environmental loadings. These factors result in similar environmental production costs for virgin and recycled steel. While electric arc furnaces (EAFs), another furnace type used for steel production can use 100% can scrap, the impact of these furnaces was not modeled in this study. EAFs do not produce steel with the properties required for can sheet; instead EAFs produce steel products such as reinforcing bar for construction, rod, nails, or wire.

The one environmental production cost that stands out in Table 3.1 is the cost associated with PVC. This high cost is due to air emissions of vinyl chloride. Air emission standards have been promulgated to lower vinyl chloride emissions from PVC manufacturing[2] - in fact, vinyl chloride is one of the few air toxics that is regulated. However, vinyl chloride is still emitted during PVC manufacture and this chemical is a known human carcinogen. Vinyl chloride therefore has a high price due to a high health effects ranking.

Appendix I of this chapter presents the emissions from the production of each packaging material (expressed as pounds of pollutant/ton of material) and the cost attributed to that pollutant (dollars/ton of material).* Comparing the cost of each pollutant to the total cost attributed to a material's production reveals an interesting fact - a few pollutants often dominate the total environmental cost of a material.

For all plastics except PET and PVC, naphthalene presents the highest environmental production cost. On average, naphthalene accounts for over 33% of the total environmental production costs associated with production of HDPE, LDPE, polypropylene, and polystyrene. Air emissions from petroleum refining is one source of naphthalene for these plastics. Monomer production also emits this pollutant into water. For PET production, naphthalene is emitted into water from several other production stages - ethylene, paraxylene, ethylene oxide/ethylene glycol, and PET production. Polystyrene production also emits naphthalene from several production stages including ethylene, benzene and ethylbenzene/styrene production. Thus, the environmental costs of production for these plastics could be reduced if naphthalene emissions from petroleum refining and monomer production was reduced.

PET and PVC are the only plastics that do not have an environmental production cost dominated by naphthalene. Instead, antimony dominates this cost for PET and vinyl chloride (or VC) dominates this cost for PVC. VC is emitted into the air and water during monomer production and PVC production. As discussed previously, VC has a high cost due to a high health effects ranking.

For paper and paperboard products that include kraft pulping as a production stage (bleached kraft paperboard, unbleached coated folding boxboard, linerboard, and unbleached kraft paper), particulate emissions from this production step present the major environmental production cost. Due to particulate emissions, these types of paper products

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* The environmental cost (dollars/ton of material) for some pollutants in Appendix I appear as $0.00 due to rounding of numbers. Thus, costs for these pollutant are less than 1¢/ton of packaging material.

have higher per ton environmental costs than other types of paper and paperboard. Thus, if emissions from kraft pulping were reduced, the environmental costs associated with those paper products would also be reduced.

Virgin corrugating medium and the three products made from wastepaper - folding boxboard, linerboard and corrugating medium - have environmental production costs that are dominated by sulfur oxides and nitrogen oxides (50% and 25% respectively). Air emissions associated with energy production is the source of these emissions. The environmental cost attributable to sulfur oxides and nitrogen oxides tends to be higher for recycled paper products than virgin paper products. Production of recycled paper materials requires the purchase of electricity as opposed to waste products that are used to provide energy for virgin paper products. This fuel switching increases the per ton of material emissions of sulfur oxides and nitrogen oxides.

Emissions of nitrogen oxides, sulfur oxides, and VOCs are the predominant environmental costs associated with virgin glass production. As discussed previously, this is due to air emissions from raw materials acquisition. Due to the fact that recycling glass eliminates the need for raw materials and decreases the amount of energy needed to produce one ton of material, the total environmental cost for recycled glass production is lower.

The pollutants that contribute most to the overall environmental production cost for virgin aluminum are nitrogen oxides, particulates, and sulfur oxides. These three pollutants are emitted into the air from the Bayer process, the production stage that converts bauxite (the raw material for aluminum production) into alumina and from the Hall-Heroult process, the production stage that converts alumina to aluminum. Particulates are emitted during these two production processes while the energy required for these production stages is the source of nitrogen oxides and sulfur oxides.

The environmental production cost for recycled aluminum is much lower than virgin aluminum. As stated previously, this is due to the fact that recycled aluminum production eliminates two processing steps (alumina and aluminum production) and less energy is required to produce one ton of recycled aluminum. For recycled aluminum, the two major pollutants are particulates from demagging (a production step unique to recycled aluminum production that reduces the level of magnesium in the used beverage cans) and sulfur oxides and nitrogen oxides from process energy.

For both virgin and recycled steel, four pollutants account for almost 80% of the environmental production cost - lead, sulfur oxides, particulates, and coke oven emissions. The source of coke oven emissions is fugitive air emissions from the ovens. Lead is emitted into the water from iron ore sinter production, pig iron production, and steel production in the basic oxygen furnace; it is also emitted into air from every production process due to process energy use. Similarly, sulfur oxides are associated with process energy use, especially during pig iron production. Particulate emissions are associated with coke production, pig iron production and basic oxygen furnace steel production. Very little reduction in environmental impacts is gained from recycled steel production. This is due to the fact that the basic oxygen furnace, which is the only furnace type that can be used for can stock, uses very little recycled steel.

3.2 DISPOSAL COSTS

Table 3.2 presents the packaging disposal costs. The first column of this table presents the conventional costs of disposal, as calculated in our report entitled *The Marginal Cost of Handling Packaging Materials in the New Jersey Solid Waste Stream*. The conventional waste management cost is the monetary cost of collection, transport, processing, and disposal of waste. For collection and transport, this is the purchase and maintenance of trucks as well as the collection workers' wages and the fuel consumed to operate the trucks. For processing and disposal, this is the cost of constructing, operating, and closing different waste disposal facilities. The second column in this table summarizes the environmental costs of disposal, as presented in Table 2.15.

As shown in Table 3.2, the disposal costs for each material are dominated by conventional costs. Aluminum has the lowest conventional disposal cost due to the fact that aluminum commands a high revenue at the MRF.

Perhaps the most outstanding fact shown in Table 3.2 is the total cost of disposal for almost all packaging materials is small when compared to the environmental costs of packaging material production. Thus, when considering the total impacts of packaging, the impacts are dominated by production rather than disposal impacts. However, these impacts may occur at different geographical locations. While many of the disposal impacts are realized in New Jersey, many of the production impacts occur outside the state.

The next table, Table 3.3 presents the full cost of packaging production. This cost is presented both for a ton of material and for an ounce of packaging material. These costs form the basis for the case studies of specific products presented in Chapter 4.

3.3 ENDNOTES FOR CHAPTER 3

1. U.S. EPA, Office of Solid Waste, 1988. *Solid Waste Disposal in the United States*, EPA/530-SW-88-011), Oct.

2. U.S. EPA, Office of Air Quality Planning and Standards, 1982. *Vinyl Chloride - A Review of National Emission Standards*, EPA-450/3-82-003, February.

Table 3.1 Environmental Cost of Packaging Material Production

Materials	Criteria Air Pollutants, Methane, Chlorine, and Hydrogen Chloride ($/ton material)	Toxic and Carcinogenic Pollutants ($/ton material)	TOTAL Environmental Production ($/ton material)	% of Total Env. Production from Toxics/ Carcinogens
PLASTIC				
HDPE	$170	$122	$292	42%
LDPE	$210	$134	$344	39%
PET	$261	$593	$854	69%
PP	$157	$210	$367	57%
PS	$189	$196	$385	51%
PVC	$188	$4,864	$5,053	96%
PAPER				
Bleached Kraft Paperboard	$229	$101	$330	31%
Unbleached Coated Folding Boxboard	$187	$82	$269	30%
Linerboard	$193	$80	$273	29%
Corrugating Medium	$77	$6	$83	8%
Unbleached Kraft Paper	$193	$84	$277	30%
Folding Boxboard from wastepaper	$120	$14	$135	11%
Linerboard from wastepaper	$121	$15	$135	11%
Corrugating Medium from wastepaper	$162	$21	$183	12%
Virgin Glass	$83	$3	$85	3%
Recycled Glass	$54	$0	$55	0%
Virgin Aluminum	$1,511	$423	$1,933	22%
Recycled Aluminum	$312	$1	$313	0%
Virgin Steel	$74	$156	$230	68%
Recycled Steel	$74	$147	$222	66%

Table 3.2 Full Costs of Packaging Material Disposal

Materials	Conventional Disposal ($/ton material)	Environmental Disposal ($/ton material)	TOTAL DISPOSAL ($/ton material)
PLASTIC			
HDPE	$242	$4	$245
LDPE	$232	$4	$236
PET	$250	$5	$255
PP	$232	$4	$236
PS	$232	$4	$236
PVC	$232	$4	$236
PAPER			
Bleached Kraft Paperboard	$110	$2	$112
Unbleached Coated Folding Boxboard	$110	$2	$112
Linerboard	$118	$2	$120
Corrugating Medium	$118	$2	$120
Unbleached Kraft Paper	$110	$2	$112
Folding Boxboard from wastepaper	$110	$2	$112
Linerboard from wastepaper	$118	$2	$120
Corrugating Medium from wastepaper	$118	$2	$120
Virgin Glass	$71	$1	$72
Recycled Glass	$71	$1	$72
Virgin Aluminum	$24	$5	$29
Recycled Aluminum	$24	$5	$29
Virgin Steel	$134	$2	$136
Recycled Steel	$134	$2	$136

Table 3.3 Full Costs of Packaging Material Production and Disposal

Materials	FULL COST ($/ton material)	FULL COST ($/ounce material)
PLASTIC		
HDPE	$537	$0.017
LDPE	$580	$0.018
PET	$1,108	$0.035
PP	$602	$0.019
PS	$620	$0.019
PVC	$5,288	$0.165
PAPER		
Bleached Kraft Paperboard	$443	$0.014
Unbleached Coated Folding Boxboard	$382	$0.012
Linerboard	$394	$0.012
Corrugating Medium	$204	$0.006
Unbleached Kraft Paper	$390	$0.012
Folding Boxboard from wastepaper	$247	$0.008
Linerboard from wastepaper	$256	$0.008
Corrugating Medium from wastepaper	$303	$0.009
Virgin Glass	$157	$0.005
Recycled Glass	$127	$0.004
Virgin Aluminum	$1,963	$0.061
Recycled Aluminum	$342	$0.011
Virgin Steel	$366	$0.011
Recycled Steel	$358	$0.011

APPENDIX - VALUATION OF EMISSIONS FROM PRODUCTION OF PACKAGING MATERIALS

POLLUTANTS	Pollutant Price	HDPE (lbs poll/ ton material)	HDPE ($/ton mat)	LDPE (lbs poll/ ton material)	LDPE ($/ton mat)
CO	$0.42	7.04E-01	$0.29	7.14E-01	$0.30
NOx	$3.63	8.22E+00	$29.84	1.18E+01	$42.83
Particulates	$5.85	2.81E+00	$16.44	3.86E+00	$22.58
SOx	$5.87	2.10E+01	$123.27	2.45E+01	$143.82
VOCs	$2.50	1.18E-01	$0.30	1.30E-01	$0.33
Acenapthene	$37.33	9.22E-07	$0.00	9.49E-07	$0.00
Acenapthylene		2.91E-07		8.27E+01	
Acetophenone	$22.40				
Acrylonitrile	$465.23	6.50E-04	$0.30	6.69E-04	$0.31
Aluminum		2.81E-02			
Ammonia	$2.31	2.14E-07	$0.00	2.25E-07	$0.00
Anthracene	$7.47	1.66E-05	$0.00	1.71E-05	$0.00
Antimony	$5,600.00				
Arsenic	$22,658.46	3.15E-13	$0.00	3.30E-13	$0.00
Benzene	$24.98	1.75E-01	$4.37	1.95E-01	$4.87
Benzo(a)anthracene		1.14E-06		9.49E-07	
Benzo(a)pyrene		9.97E-07		1.03E-06	
3,4-Benzofluoranthene				6.50E-07	
Benzo(k)flouranthene		6.31E-07		6.50E-07	
Benzo(ghi)perylene		3.79E-06		3.90E-06	
Benzoic acid	$0.56				
Beryllium	$2,076.31	1.26E-12	$0.00	1.32E-12	$0.00
Biphenyl	$44.80	3.35E-03	$0.15	3.52E-03	$0.16
Bis(2-ethylhexyl) phthalate	$62.03	9.22E-07	$0.00	9.49E-07	$0.00
1,3-Butadiene	$1,550.77				
2-Butanone					
Butyl benzyl phthalate	$11.20				
Cadmium	$4,867.69	6.31E-07	$0.00	6.50E-07	$0.00
Carbon disulfide	$22.40				
Carbon tetrachloride	$1,656.00				
Chlorine	$5.87				
Chlorobenzene	$112.00	2.91E-07	$0.00	2.99E-07	$0.00
Chloroethane					
Chloroform	$114.63	9.22E-07	$0.00	9.49E-07	$0.00
p-Chloro-m-cresol	$1.12	9.44E-12	$0.00	9.91E-12	$0.00
2-Chlorophenol	$448.00	8.15E-06	$0.00	8.39E-06	$0.00

POLLUTANTS	Pollutant Price	HDPE (lbs poll/ ton material)	HDPE ($/ton mat)	LDPE (lbs poll/ ton material)	LDPE ($/ton mat)
Chloroprene	$112.00				
Chromium	$2.24	2.54E-05	$0.00	3.03E-02	$0.07
Chrysene		6.31E-07		6.50E-07	
Copper	$60.31	5.52E-05	$0.00	1.47E-03	$0.09
Coke oven emissions	$24.98				
Cyanide	$112.00	6.60E-06	$0.00	6.80E-06	$0.00
Dibenzo(a,h)anthracene					
1,4-Dichlorobenzene	$11.94				
Dichlorobromomethane		6.31E-07		6.50E-07	
1,1-Dichloroethane	$22.40				
1,2-Dichloroethane	$78.40			4.32E-04	$0.03
1,1-Dichloroethylene	$382.91				
1,2-trans-Dichloroethylene	$112.00				
2,4-Dichlorophenol	$746.67	6.29E-12	$0.00	6.61E-12	$0.00
1,2-Dichloropropane	$58.58				
1,3-Dichloropropene	$3,810.87				
Diethyl phthalate	$2.80	4.72E-11	$0.00	4.96E-11	$0.00
2,4-Dimethylphenol	$112.00	1.43E-04	$0.02	1.47E-04	$0.02
Dimethyl phthalate	$2.24	3.15E-12	$0.00	1.02E-02	$0.02
Di-n-butyl phthalate		1.26E-12		1.62E-03	
4,6-Dinitro-o-cresol					
2,4-Dinitrotoluene	$585.85				
2,6-Dinitrotoluene	$585.85				
1,2-Diphenylhydrazine	$689.23				
Ethylbenzene	$22.40	3.11E-01	$6.97	3.27E-01	$7.32
Ethylchloride					
Ethylene oxide	$301.54				
Fluoranthene	$56.00	9.22E-07	$0.00	9.49E-07	$0.00
Fluorene	$56.00	1.21E-06	$0.00	1.25E-06	$0.00
Fluoride	$37.33				
Hexachlorobenzene	$2,089.23				
Hydrogen chloride	$5.87				
Hydrogen sulfide	$34.72				
Indeno(1,2,3-cd)pyrene		6.31E-07		6.50E-07	
Isophorone	$7.28				
Lindane	$7,466.67				
Lead	$1,600.00	4.22E-03	$6.75	7.62E-03	$12.19
Manganese	$11.20				
Mercury	$7,466.67	6.31E-07	$0.00	3.66E-05	$0.27

POLLUTANTS	Pollutant Price	HDPE (lbs poll/ ton material)	HDPE ($/ton mat)	LDPE (lbs poll/ ton material)	LDPE ($/ton mat)
Methane	$0.04	1.04E-01	$0.00	9.38E-02	$0.00
Methylene chloride	$21.90	9.22E-07	$0.00	9.49E-07	$0.00
Napthalene	$560.00	1.84E-01	$103.04	1.94E-01	$108.64
Nickel	$417.85	1.07E-10	$0.00	1.12E-10	$0.00
Nitrobenzene	$4,480.00	6.31E-07	$0.00	6.50E-07	$0.00
Parachloronitrocresol					
Pentachlorophenol	$74.67				
Phenanthrene		4.85E-06		4.99E-06	
Phenol	$3.73	2.43E-03	$0.01	4.26E-03	$0.02
Propylene	$206.77				
Pyrene	$74.67	9.22E-07	$0.00	9.49E-07	$0.00
Selenium	$746.67	1.26E-06	$0.00	1.30E-06	$0.00
Silver	$746.67	1.26E-12	$0.00	1.32E-12	$0.00
Sodium hydroxide					
Styrene	$18.52				
Sulfides	$34.72	1.89E-08	$0.00	1.98E-08	$0.00
2378-TCDD	$129,230,769.23				
2378-TCDF					
Tetrachloroethylene	$133.97				
Thallium	$32,000.00	7.87E-12	$0.00	8.26E-12	$0.00
Thiocyanates					
Toluene	$7.47	6.68E-03	$0.05	2.14E-02	$0.16
1,1,1-Trichloroethane	$37.00				
Trichloroethylene	$9.48				
Trichlorofluoromethane	$7.47				
2,4,6-Trichlorophenol	$9.48				
Triethanol					
Vanadium	$320.00				
Vinyl chloride	$1,981.54				
Xylenes	$1.12	2.41E-02	$0.03	2.53E-02	$0.03
Zinc	$11.20	7.56E-05	$0.00	7.78E-05	$0.00
TOTAL			**$291.85**		**$344.07**
($/ounce mat)			$0.009		$0.011

Notes: Blank entries imply zero values; $0.00 entries imply positive values rounded to zero.

POLLUTANTS	PP (lbs poll/ ton material)	PP ($/ton mat)	PET (lbs poll/ ton material)	PET ($/ton mat)
CO	6.24E-01	$0.26	1.43E+00	$0.60
NOx	6.86E+00	$24.90	1.20E+01	$43.56
Particulates	2.44E+00	$14.27	4.03E+00	$23.58
SOx	1.99E+01	$116.81	3.24E+01	$190.19
VOCs	1.56E-01	$0.39	1.27E+00	$3.18
Acenapthene	3.98E-03	$0.15	3.50E-06	$0.00
Acenapthylene	1.67E-07		3.20E-06	
Acetophenone				
Acrylonitrile	9.64E-04	$0.45	3.41E-03	$1.59
Aluminum				
Ammonia	3.93E-07	$0.00	3.18E-07	$0.00
Anthracene	1.69E-05	$0.00	7.64E-06	$0.00
Antimony			6.86E-02	$384.16
Arsenic	5.78E-13	$0.00	4.68E-13	$0.00
Benzene	4.21E-03	$0.11	8.37E-02	$2.09
Benzo(a)anthracene	1.10E-06		4.25E-07	
Benzo(a)pyrene	1.24E-06		5.37E-07	
3,4-Benzofluoranthene	9.36E-07		2.91E-07	
Benzo(k)flouranthene	9.36E-07		2.91E-07	
Benzo(ghi)perylene	5.62E-06		1.74E-06	
Benzoic acid				
Beryllium	2.31E-12	$0.00	1.87E-12	$0.00
Biphenyl	6.16E-03	$0.28	4.98E-03	$0.22
Bis(2-ethylhexyl) phthalate	8.88E-05	$0.01	4.25E-07	$0.00
1,3-Butadiene				
2-Butanone				
Butyl benzyl phthalate				
Cadmium	9.36E-07	$0.00	2.91E-07	$0.00
Carbon disulfide				
Carbon tetrachloride				
Chlorine				
Chlorobenzene	1.67E-07	$0.00	1.34E-07	$0.00
Chloroethane				
Chloroform	3.08E-04	$0.04	3.95E-05	$0.00
p-Chloro-m-cresol	1.73E-11	$0.00	1.40E-11	$0.00
2-Chlorophenol	1.14E-05	$0.01	7.70E-05	$0.03

POLLUTANTS	PP (lbs poll/ ton material)	PP ($/ton mat)	PET (lbs poll/ ton material)	PET ($/ton mat)
Chloroprene				
Chromium	1.30E-02	$0.03	6.83E-04	$0.00
Chrysene	9.36E-07		2.91E-07	
Copper	6.50E-05	$0.00	2.59E-04	$0.02
Coke oven emissions				
Cyanide	9.53E-06	$0.00	7.67E-05	$0.01
Dibenzo(a,h)anthracene				
1,4-Dichlorobenzene				
Dichlorobromomethane	9.36E-07		2.91E-07	
1,1-Dichloroethane				
1,2-Dichloroethane			8.01E-05	$0.01
1,1-Dichloroethylene				
1,2-trans-Dichloroethylene				
2,4-Dichlorophenol	1.16E-11	$0.00	9.35E-12	$0.00
1,2-Dichloropropane				
1,3-Dichloropropene				
Diethyl phthalate	8.67E-11	$0.00	7.01E-11	$0.00
2,4-Dimethylphenol	2.12E-04	$0.02	3.54E-04	$0.04
Dimethyl phthalate	5.78E-12	$0.00	2.59E-05	$0.00
Di-n-butyl phthalate	2.31E-12		6.57E-05	
4,6-Dinitro-o-cresol				
2,4-Dinitrotoluene				
2,6-Dinitrotoluene				
1,2-Diphenylhydrazine				
Ethylbenzene	5.84E-01	$13.07	4.68E-01	$10.47
Ethylchloride				
Ethylene oxide			9.70E-02	$29.25
Fluoranthene	1.10E-06	$0.00	4.25E-07	$0.00
Fluorene	1.27E-06	$0.00	3.63E-06	$0.00
Fluoride				
Hexachlorobenzene				
Hydrogen chloride				
Hydrogen sulfide				
Indeno(1,2,3-cd)pyrene	9.36E-07		2.91E-07	
Isophorone				
Lindane				
Lead	3.17E-03	$5.07	5.28E-03	$8.45
Manganese				
Mercury	9.36E-07	$0.01	1.46E-06	$0.01

POLLUTANTS	PP (lbs poll/ ton material)	PP ($/ton mat)	PET (lbs poll/ ton material)	PET ($/ton mat)
Methane	9.56E-02	$0.00	2.02E-01	$0.01
Methylene chloride	1.10E-06	$0.00	7.77E-05	$0.00
Napthalene	3.39E-01	$189.84	2.75E-01	$154.04
Nickel	1.97E-10	$0.00	1.59E-10	$0.00
Nitrobenzene	9.36E-07	$0.00	2.91E-07	$0.00
Parachloronitrocresol				
Pentachlorophenol				
Phenanthrene	5.08E-06		2.23E-06	
Phenol	2.41E-03	$0.01	5.44E-01	$2.03
Propylene				
Pyrene	1.10E-06	$0.00	4.25E-07	$0.00
Selenium	1.87E-06	$0.00	5.82E-07	$0.00
Silver	2.31E-12	$0.00	1.87E-12	$0.00
Sodium hydroxide				
Styrene				
Sulfides	3.47E-08	$0.00	2.81E-06	$0.00
2378-TCDD				
2378-TCDF				
Tetrachloroethylene			3.07E-06	$0.00
Thallium	1.44E-11	$0.00	1.17E-11	$0.00
Thiocyanates				
Toluene	2.15E-02	$0.16	1.04E-02	$0.08
1,1,1-Trichloroethane	1.16E-02	$0.43		
Trichloroethylene				
Trichlorofluoromethane				
2,4,6-Trichlorophenol				
Triethanol			2.60E-02	
Vanadium				
Vinyl chloride			3.07E-06	$0.01
Xylenes	4.43E-02	$0.05	3.58E-02	$0.04
Zinc	1.36E-02	$0.15	9.10E-04	$0.01
TOTAL		**$366.53**		**$853.67**
($/ounce mat)		$0.011		$0.027

POLLUTANTS	PS (lbs poll/ ton material)	PS ($/ton mat)	PVC (lbs poll/ ton material)	PVC ($/ton mat)
CO	1.03E+00	$0.43	6.36E-01	$0.27
NOx	8.82E+00	$32.02	1.05E+01	$38.12
Particulates	2.95E+00	$17.26	3.36E+00	$19.66
SOx	2.36E+01	$138.53	2.02E+01	$118.57
VOCs	1.57E-01	$0.39	1.16E-01	$0.29
Acenapthene	3.31E-06	$0.00	4.16E-07	$0.00
Acenapthylene	4.73E-07		1.31E-07	
Acetophenone				
Acrylonitrile	2.16E-04	$0.10	2.93E-04	$0.14
Aluminum				
Ammonia	2.69E-07	$0.00	1.94E-07	$0.00
Anthracene	7.17E-05	$0.00	7.48E-06	$0.00
Antimony				
Arsenic	3.96E-13	$0.00	2.86E-13	$0.00
Benzene	1.93E+00	$48.31	7.97E-02	$1.99
Benzo(a)anthracene	9.56E-06		2.85E-07	
Benzo(a)pyrene	1.36E-06		4.84E-07	
3,4-Benzofluoranthene	8.35E-07		2.85E-07	
Benzo(k)flouranthene	1.21E-06		2.85E-07	
Benzo(ghi)perylene	5.42E-06		2.45E-06	
Benzoic acid				
Beryllium	1.58E-12	$0.00	1.14E-12	$0.00
Biphenyl	4.22E-03	$0.19	3.04E-03	$0.14
Bis(2-ethylhexyl) phthalate	5.43E-04	$0.03	1.27E-04	$0.01
1,3-Butadiene			4.20E-04	$0.65
2-Butanone				
Butyl benzyl phthalate	3.19E-05	$0.00		
Cadmium	1.73E-07	$0.00	7.14E-12	$0.00
Carbon disulfide				
Carbon tetrachloride	1.45E-07	$0.00	6.61E-01	$1,094.28
Chlorine			1.92E+00	$11.27
Chlorobenzene	1.10E-06	$0.00		
Chloroethane	1.35E-06		3.30E-05	
Chloroform	2.75E-05	$0.00	9.43E-01	$108.11
p-Chloro-m-cresol	1.19E-11	$0.00	8.57E-12	$0.00
2-Chlorophenol	5.71E-05	$0.03	4.16E-07	$0.00

POLLUTANTS	PS (lbs poll/ ton material)	PS ($/ton mat)	PVC (lbs poll/ ton material)	PVC ($/ton mat)
Chloroprene			8.00E-05	$0.01
Chromium	7.82E-04	$0.00	4.37E-04	$0.00
Chrysene	1.73E-07		3.68E-06	
Copper	2.06E-05	$0.00	5.77E-03	$0.35
Coke oven emissions				
Cyanide	5.01E-06	$0.00	5.27E-05	$0.01
Dibenzo(a,h)anthracene	1.45E-07			
1,4-Dichlorobenzene	1.45E-07	$0.00		
Dichlorobromomethane	1.50E-06			
1,1-Dichloroethane	2.56E-06	$0.00		
1,2-Dichloroethane	1.61E-04	$0.01	4.30E+00	$337.12
1,1-Dichloroethylene	4.20E-07	$0.00		
1,2-trans-Dichloroethylene				
2,4-Dichlorophenol	1.53E-06	$0.00	5.71E-12	$0.00
1,2-Dichloropropane	4.00E-04	$0.02		
1,3-Dichloropropene	1.45E-07	$0.00	2.05E-04	$0.78
Diethyl phthalate	5.94E-11	$0.00	4.28E-11	$0.00
2,4-Dimethylphenol	1.45E-07	$0.00		
Dimethyl phthalate	5.38E-04	$0.00	2.86E-12	$0.00
Di-n-butyl phthalate	3.91E-05		3.54E-04	
4,6-Dinitro-o-cresol				
2,4-Dinitrotoluene				
2,6-Dinitrotoluene				
1,2-Diphenylhydrazine	1.09E-05	$0.01		
Ethylbenzene	3.93E-01	$8.80	2.83E-01	$6.34
Ethylchloride			9.80E-03	
Ethylene oxide				
Fluoranthene	3.44E-06	$0.00		
Fluorene	5.82E-04	$0.03	4.16E-07	$0.00
Fluoride				
Hexachlorobenzene				
Hydrogen chloride			4.20E-04	$0.00
Hydrogen sulfide				
Indeno(1,2,3-cd)pyrene	1.54E-06		5.47E-07	
Isophorone				
Lindane	9.36E-06	$0.07		
Lead	3.96E-03	$6.33	9.87E-03	$15.79
Manganese				
Mercury	3.59E-07	$0.00	2.57E-11	$0.00

POLLUTANTS	PS (lbs poll/ ton material)	PS ($/ton mat)	PVC (lbs poll/ ton material)	PVC ($/ton mat)
Methane	1.45E-01	$0.01	7.62E-02	$0.00
Methylene chloride	4.13E-06	$0.00	1.92E-02	$0.42
Napthalene	2.32E-01	$130.10	1.67E-01	$93.52
Nickel	1.49E-05	$0.01	4.43E-03	$1.85
Nitrobenzene	1.73E-07	$0.00	1.88E-06	$0.01
Parachloronitrocresol				
Pentachlorophenol			5.37E-04	$0.04
Phenanthrene	5.77E-04		5.71E-12	
Phenol	3.45E-03	$0.01	2.19E-06	$0.00
Propylene			1.40E-04	$0.03
Pyrene	1.96E-05	$0.00	1.09E-03	$0.08
Selenium	3.47E-07	$0.00	4.91E-10	$0.00
Silver	1.58E-12	$0.00	1.14E-12	$0.00
Sodium hydroxide				
Styrene	1.04E-01	$1.93		
Sulfides	2.38E-08	$0.00	1.71E-08	$0.00
2378-TCDD				
2378-TCDF				
Tetrachloroethylene			3.73E-03	$0.50
Thallium	9.90E-12	$0.00	5.69E-07	$0.02
Thiocyanates				
Toluene	1.04E-02	$0.08	6.05E-03	$0.05
1,1,1-Trichloroethane			7.68E-05	$0.00
Trichloroethylene			2.45E-01	$2.32
Trichlorofluoromethane	4.88E-05	$0.00		
2,4,6-Trichlorophenol	4.38E-07	$0.00		
Triethanol				
Vanadium				
Vinyl chloride			1.61E+00	$3,199.79
Xylenes	3.04E-02	$0.03	2.19E-02	$0.02
Zinc	6.96E-04	$0.01	1.24E-04	$0.00
TOTAL		**$384.76**		**$5,052.53**
($/ounce mat)		$0.012		$0.158

POLLUTANTS	Bleached Kraft Paperboard (lbs poll/ ton material)	Bleached Kraft Paperboard ($/ton mat)	Unbleached Coated Folding Boxboard (lbs poll/ ton mat)	Unbleached Coated Folding Boxboard ($/ton mat)
CO	2.15E+01	$8.98	1.83E+01	$7.64
NOx	4.95E+00	$17.97	3.15E+00	$11.43
Particulates	2.40E+01	$140.40	2.10E+01	$122.85
SOx	9.98E+00	$58.58	7.56E+00	$44.38
VOCs	5.90E-01	$1.48	4.61E-01	$1.15
Acenapthene				
Acenapthylene				
Acetophenone				
Acrylonitrile				
Aluminum				
Ammonia				
Anthracene				
Antimony				
Arsenic				
Benzene	2.95E-04	$0.01		
Benzo(a)anthracene				
Benzo(a)pyrene				
3,4-Benzofluoranthene				
Benzo(k)flouranthene				
Benzo(ghi)perylene				
Benzoic acid				
Beryllium				
Biphenyl				
Bis(2-ethylhexyl) phthalate	8.85E-04	$0.05	1.79E-04	$0.01
1,3-Butadiene				
2-Butanone				
Butyl benzyl phthalate				
Cadmium				
Carbon disulfide				
Carbon tetrachloride				
Chlorine	2.54E-01	$1.49		
Chlorobenzene				
Chloroethane				
Chloroform	1.77E-03	$0.20		
p-Chloro-m-cresol				
2-Chlorophenol				

POLLUTANTS	Bleached Kraft Paperboard (lbs poll/ ton material)	Bleached Kraft Paperboard ($/ton mat)	Unbleached Coated Folding Boxboard (lbs poll/ ton mat)	Unbleached Coated Folding Boxboard ($/ton mat)
Chloroprene				
Chromium	1.62E-02	$0.04	2.15E-03	$0.00
Chrysene				
Copper	5.84E-03	$0.35	1.62E-03	$0.10
Coke oven emissions				
Cyanide				
Dibenzo(a,h)anthracene				
1,4-Dichlorobenzene				
Dichlorobromomethane				
1,1-Dichloroethane				
1,2-Dichloroethane				
1,1-Dichloroethylene				
1,2-trans-Dichloroethylene				
2,4-Dichlorophenol	2.95E-04	$0.22		
1,2-Dichloropropane				
1,3-Dichloropropene				
Diethyl phthalate				
2,4-Dimethylphenol				
Dimethyl phthalate				
Di-n-butyl phthalate	1.18E-03			
4,6-Dinitro-o-cresol				
2,4-Dinitrotoluene				
2,6-Dinitrotoluene				
1,2-Diphenylhydrazine				
Ethylbenzene	2.95E-04	$0.01		
Ethylchloride				
Ethylene oxide				
Fluoranthene				
Fluorene				
Fluoride				
Hexachlorobenzene				
Hydrogen chloride				
Hydrogen sulfide	2.41E+00	$83.68	2.13E+00	$73.95
Indeno(1,2,3-cd)pyrene				
Isophorone				
Lindane				
Lead	8.54E-03	$13.66	4.43E-03	$7.09
Manganese				
Mercury	7.38E-06	$0.06	4.49E-06	$0.03

POLLUTANTS	Bleached Kraft Paperboard (lbs poll/ ton material)	Bleached Kraft Paperboard ($/ton mat)	Unbleached Coated Folding Boxboard (lbs poll/ ton mat)	Unbleached Coated Folding Boxboard ($/ton mat)
Methane	1.76E-01	$0.01	1.37E-01	$0.01
Methylene chloride	5.90E-04	$0.01	7.18E-04	$0.02
Napthalene				
Nickel	4.17E-03	$1.74	8.97E-04	$0.37
Nitrobenzene				
Parachloronitrocresol				
Pentachlorophenol	5.61E-03	$0.42		
Phenanthrene				
Phenol	1.48E-03	$0.01		
Propylene				
Pyrene				
Selenium				
Silver				
Sodium hydroxide				
Styrene				
Sulfides				
2378-TCDD	5.03E-09	$0.65		
2378-TCDF	2.24E-07			
Tetrachloroethylene				
Thallium				
Thiocyanates				
Toluene				
1,1,1-Trichloroethane				
Trichloroethylene				
Trichlorofluoromethane				
2,4,6-Trichlorophenol	2.95E-04	$0.00		
Triethanol				
Vanadium				
Vinyl chloride				
Xylenes				
Zinc	3.25E-02	$0.36	1.45E-02	$0.16
TOTAL		$330.38		$269.21
($/ounce mat)		$0.010		$0.008

POLLUTANTS	Linerboard (lbs poll/ ton material)	Linerboard ($/ton mat)	Corrugating Medium (lbs poll/ ton material)	Corrugating Medium ($/ton mat)
CO	1.89E+01	$7.89	4.76E+00	$1.99
NOx	3.14E+00	$11.40	5.59E+00	$20.29
Particulates	2.18E+01	$127.53	1.57E+00	$9.18
SOx	7.66E+00	$44.96	7.62E+00	$44.73
VOCs	4.75E-01	$1.19	3.27E-01	$0.82
Acenapthene				
Acenapthylene				
Acetophenone				
Acrylonitrile				
Aluminum				
Ammonia				
Anthracene				
Antimony				
Arsenic				
Benzene			1.08E-04	$0.00
Benzo(a)anthracene				
Benzo(a)pyrene				
3,4-Benzofluoranthene				
Benzo(k)flouranthene				
Benzo(ghi)perylene				
Benzoic acid				
Beryllium				
Biphenyl				
Bis(2-ethylhexyl) phthalate	2.84E-04	$0.02	8.08E-04	$0.05
1,3-Butadiene				
2-Butanone				
Butyl benzyl phthalate				
Cadmium				
Carbon disulfide				
Carbon tetrachloride				
Chlorine				
Chlorobenzene				
Chloroethane				
Chloroform				
p-Chloro-m-cresol				
2-Chlorophenol				

POLLUTANTS	Linerboard (lbs poll/ ton material)	Linerboard ($/ton mat)	Corrugating Medium (lbs poll/ ton material)	Corrugating Medium ($/ton mat)
Chloroprene				
Chromium	6.62E-04	$0.00	1.02E-03	$0.00
Chrysene				
Copper	4.73E-04	$0.03	1.35E-03	$0.08
Coke oven emissions				
Cyanide				
Dibenzo(a,h)anthracene				
1,4-Dichlorobenzene				
Dichlorobromomethane				
1,1-Dichloroethane				
1,2-Dichloroethane				
1,1-Dichloroethylene				
1,2-trans-Dichloroethylene				
2,4-Dichlorophenol				
1,2-Dichloropropane				
1,3-Dichloropropene				
Diethyl phthalate				
2,4-Dimethylphenol				
Dimethyl phthalate				
Di-n-butyl phthalate	9.45E-05			
4,6-Dinitro-o-cresol				
2,4-Dinitrotoluene				
2,6-Dinitrotoluene				
1,2-Diphenylhydrazine				
Ethylbenzene			5.39E-05	$0.00
Ethylchloride				
Ethylene oxide				
Fluoranthene				
Fluorene				
Fluoride				
Hexachlorobenzene				
Hydrogen chloride				
Hydrogen sulfide	2.21E+00	$76.73		
Indeno(1,2,3-cd)pyrene				
Isophorone				
Lindane				
Lead	2.01E-03	$3.22	3.69E-03	$5.90
Manganese				
Mercury	2.36E-06	$0.02	1.35E-06	$0.01

POLLUTANTS	Linerboard (lbs poll/ ton material)	Linerboard ($/ton mat)	Corrugating Medium (lbs poll/ ton material)	Corrugating Medium ($/ton mat)
Methane	1.38E-01	$0.01	9.02E-02	$0.00
Methylene chloride			2.69E-04	$0.01
Napthalene				
Nickel	4.73E-04	$0.20	5.39E-04	$0.23
Nitrobenzene				
Parachloronitrocresol				
Pentachlorophenol			5.39E-05	$0.00
Phenanthrene				
Phenol	2.84E-04	$0.00	7.54E-04	$0.00
Propylene				
Pyrene				
Selenium				
Silver				
Sodium hydroxide				
Styrene				
Sulfides				
2378-TCDD				
2378-TCDF				
Tetrachloroethylene				
Thallium				
Thiocyanates				
Toluene			1.08E-04	$0.00
1,1,1-Trichloroethane				
Trichloroethylene				
Trichlorofluoromethane				
2,4,6-Trichlorophenol				
Triethanol				
Vanadium				
Vinyl chloride				
Xylenes			1.08E-04	$0.00
Zinc	6.33E-03	$0.07	3.72E-03	$0.04
TOTAL		**$273.27**		**$83.35**
($/ounce mat)		$0.009		$0.003

POLLUTANTS	Unbleached Kraft Paper (lbs poll/ ton mat)	Unbleached Kraft Paper ($/ton mat)	Folding Boxboard from Wastepaper (lbs poll/ ton material)	Folding Boxboard from Wastepaper ($/ton mat)
CO	1.89E+01	$7.89	7.33E-01	$0.31
NOx	3.14E+00	$11.40	8.53E+00	$30.96
Particulates	2.18E+01	$127.53	1.75E+00	$10.24
SOx	7.66E+00	$44.96	1.33E+01	$78.07
VOCs	4.75E-01	$1.19	3.42E-01	$0.86
Acenapthene				
Acenapthylene				
Acetophenone				
Acrylonitrile				
Aluminum				
Ammonia				
Anthracene				
Antimony				
Arsenic				
Benzene			1.33E-05	$0.00
Benzo(a)anthracene				
Benzo(a)pyrene				
3,4-Benzofluoranthene				
Benzo(k)flouranthene				
Benzo(ghi)perylene				
Benzoic acid				
Beryllium				
Biphenyl				
Bis(2-ethylhexyl) phthalate	1.79E-04	$0.01	1.25E-03	$0.08
1,3-Butadiene				
2-Butanone				
Butyl benzyl phthalate			8.40E-04	$0.01
Cadmium				
Carbon disulfide				
Carbon tetrachloride				
Chlorine				
Chlorobenzene				
Chloroethane				
Chloroform			5.34E-05	$0.01
p-Chloro-m-cresol				
2-Chlorophenol				

POLLUTANTS	Unbleached Kraft Paper (lbs poll/ ton mat)	Unbleached Kraft Paper ($/ton mat)	Folding Boxboard from Wastepaper (lbs poll/ ton material)	Folding Boxboard from Wastepaper ($/ton mat)
Chloroprene				
Chromium	2.15E-03	$0.00	2.31E-03	$0.01
Chrysene				
Copper	1.62E-03	$0.10	2.23E-03	$0.13
Coke oven emissions				
Cyanide			4.27E-04	$0.05
Dibenzo(a,h)anthracene				
1,4-Dichlorobenzene				
Dichlorobromomethane			1.33E-05	
1,1-Dichloroethane				
1,2-Dichloroethane				
1,1-Dichloroethylene				
1,2-trans-Dichloroethylene				
2,4-Dichlorophenol				
1,2-Dichloropropane				
1,3-Dichloropropene				
Diethyl phthalate			4.59E-03	$0.01
2,4-Dimethylphenol				
Dimethyl phthalate				
Di-n-butyl phthalate			5.87E-04	
4,6-Dinitro-o-cresol				
2,4-Dinitrotoluene				
2,6-Dinitrotoluene				
1,2-Diphenylhydrazine				
Ethylbenzene				
Ethylchloride				
Ethylene oxide				
Fluoranthene				
Fluorene				
Fluoride				
Hexachlorobenzene				
Hydrogen chloride				
Hydrogen sulfide	2.21E+00	$76.73		
Indeno(1,2,3-cd)pyrene				
Isophorone				
Lindane				
Lead	4.41E-03	$7.06	7.42E-03	$11.87
Manganese				
Mercury	4.49E-06	$0.03	1.27E-05	$0.09

POLLUTANTS	Unbleached Kraft Paper (lbs poll/ ton mat)	Unbleached Kraft Paper ($/ton mat)	Folding Boxboard from Wastepaper (lbs poll/ ton material)	Folding Boxboard from Wastepaper ($/ton mat)
Methane	1.38E-01	$0.01	6.40E-02	$0.00
Methylene chloride	7.18E-04	$0.02	6.80E-04	$0.01
Napthalene				
Nickel	8.97E-04	$0.37	9.34E-04	$0.39
Nitrobenzene				
Parachloronitrocresol				
Pentachlorophenol			1.80E-02	$1.19
Phenanthrene				
Phenol			5.71E-03	$0.02
Propylene				
Pyrene				
Selenium				
Silver				
Sodium hydroxide				
Styrene				
Sulfides				
2378-TCDD				
2378-TCDF				
Tetrachloroethylene				
Thallium				
Thiocyanates				
Toluene			6.67E-05	$0.00
1,1,1-Trichloroethane			4.00E-05	$0.00
Trichloroethylene				
Trichlorofluoromethane				
2,4,6-Trichlorophenol			5.75E-03	$0.05
Triethanol				
Vanadium				
Vinyl chloride				
Xylenes			7.67E-05	$0.00
Zinc	1.45E-02	$0.16	2.15E-02	$0.24
TOTAL		**$277.47**		**$134.62**
($/ounce mat)		$0.009		$0.004

POLLUTANTS	Linerboard from Wastepaper (lbs poll/ ton mat)	Linerboard from Wastepaper ($/ton mat)	Corrugating Medium from Wastepaper (lbs poll/ ton material)	Corrugating Medium from Wastepaper ($/ton mat)
CO	7.54E-01	$0.31	1.14E+00	$0.48
NOx	8.78E+00	$31.87	1.34E+01	$48.64
Particulates	1.73E+00	$10.12	1.93E+00	$11.29
SOx	1.33E+01	$78.07	1.71E+01	$100.38
VOCs	1.72E-01	$0.43	3.07E-01	$0.77
Acenapthene				
Acenapthylene				
Acetophenone				
Acrylonitrile				
Aluminum				
Ammonia				
Anthracene				
Antimony				
Arsenic				
Benzene	1.33E-05	$0.00	1.33E-05	$0.00
Benzo(a)anthracene				
Benzo(a)pyrene				
3,4-Benzofluoranthene				
Benzo(k)flouranthene				
Benzo(ghi)perylene				
Benzoic acid				
Beryllium				
Biphenyl				
Bis(2-ethylhexyl) phthalate	1.25E-03	$0.08	1.25E-03	$0.08
1,3-Butadiene				
2-Butanone				
Butyl benzyl phthalate	8.40E-04	$0.01	8.40E-04	$0.01
Cadmium				
Carbon disulfide				
Carbon tetrachloride				
Chlorine				
Chlorobenzene				
Chloroethane				
Chloroform	5.34E-05	$0.01	5.34E-05	$0.01
p-Chloro-m-cresol				
2-Chlorophenol				

POLLUTANTS	Linerboard from Wastepaper (lbs poll/ ton mat)	Linerboard from Wastepaper ($/ton mat)	Corrugating Medium from Wastepaper (lbs poll/ ton material)	Corrugating Medium from Wastepaper ($/ton mat)
Chloroprene				
Chromium	2.31E-03	$0.01	2.31E-03	$0.01
Chrysene				
Copper	2.23E-03	$0.13	2.23E-03	$0.13
Coke oven emissions				
Cyanide	4.27E-04	$0.05	4.27E-04	$0.05
Dibenzo(a,h)anthracene				
1,4-Dichlorobenzene				
Dichlorobromomethane	1.33E-05		1.33E-05	
1,1-Dichloroethane				
1,2-Dichloroethane				
1,1-Dichloroethylene				
1,2-trans-Dichloroethylene				
2,4-Dichlorophenol				
1,2-Dichloropropane				
1,3-Dichloropropene				
Diethyl phthalate	4.59E-03	$0.01	4.59E-03	$0.01
2,4-Dimethylphenol				
Dimethyl phthalate				
Di-n-butyl phthalate	5.87E-04		5.87E-04	
4,6-Dinitro-o-cresol				
2,4-Dinitrotoluene				
2,6-Dinitrotoluene				
1,2-Diphenylhydrazine				
Ethylbenzene				
Ethylchloride				
Ethylene oxide				
Fluoranthene				
Fluorene				
Fluoride				
Hexachlorobenzene				
Hydrogen chloride				
Hydrogen sulfide				
Indeno(1,2,3-cd)pyrene				
Isophorone				
Lindane				
Lead	7.68E-03	$12.29	1.17E-02	$18.77
Manganese				
Mercury	1.27E-05	$0.09	1.27E-05	$0.09

POLLUTANTS	Linerboard from Wastepaper (lbs poll/ ton mat)	Linerboard from Wastepaper ($/ton mat)	Corrugating Medium from Wastepaper (lbs poll/ ton material)	Corrugating Medium from Wastepaper ($/ton mat)
Methane	6.34E-02	$0.00	7.30E-02	$0.00
Methylene chloride	6.80E-04	$0.01	6.80E-04	$0.01
Napthalene				
Nickel	9.34E-04	$0.39	9.34E-04	$0.39
Nitrobenzene				
Parachloronitrocresol				
Pentachlorophenol	1.60E-02	$1.19	1.60E-02	$1.19
Phenanthrene				
Phenol	5.71E-03	$0.02	5.71E-03	$0.02
Propylene				
Pyrene				
Selenium				
Silver				
Sodium hydroxide				
Styrene				
Sulfides				
2378-TCDD				
2378-TCDF				
Tetrachloroethylene				
Thallium				
Thiocyanates				
Toluene	6.67E-05	$0.00	6.67E-05	$0.00
1,1,1-Trichloroethane	4.00E-05	$0.00	4.00E-05	$0.00
Trichloroethylene				
Trichlorofluoromethane				
2,4,6-Trichlorophenol	5.75E-03	$0.05	5.75E-03	$0.05
Triethanol				
Vanadium				
Vinyl chloride				
Xylenes	7.67E-05	$0.00	7.67E-05	$0.00
Zinc	2.15E-02	$0.24	2.15E-02	$0.24
TOTAL		**$135.41**		**$182.63**
($/ounce mat)		$0.004		$0.006

POLLUTANTS	Virgin GLASS (lbs poll/ ton material)	Virgin GLASS ($/ton mat)	Recycled GLASS (lbs poll/ ton mat)	Recycled GLASS ($/ton mat)
CO	6.60E-01	$0.28	3.00E-01	$0.13
NOx	8.73E+00	$31.69	5.41E+00	$19.64
Particulates	6.20E-01	$3.63	2.00E-01	$1.17
SOx	4.19E+00	$24.60	1.93E+00	$11.33
VOCs	8.97E+00	$22.43	8.88E+00	$22.20
Acenapthene				
Acenapthylene				
Acetophenone				
Acrylonitrile				
Aluminum				
Ammonia				
Anthracene				
Antimony				
Arsenic				
Benzene				
Benzo(a)anthracene				
Benzo(a)pyrene				
3,4-Benzofluoranthene				
Benzo(k)flouranthene				
Benzo(ghi)perylene				
Benzoic acid				
Beryllium				
Biphenyl				
Bis(2-ethylhexyl) phthalate				
1,3-Butadiene				
2-Butanone				
Butyl benzyl phthalate				
Cadmium				
Carbon disulfide				
Carbon tetrachloride				
Chlorine				
Chlorobenzene				
Chloroethane				
Chloroform				
p-Chloro-m-cresol				
2-Chlorophenol				

POLLUTANTS	Virgin GLASS (lbs poll/ ton material)	Virgin GLASS ($/ton mat)	Recycled GLASS (lbs poll/ ton mat)	Recycled GLASS ($/ton mat)
Chloroprene				
Chromium				
Chrysene				
Copper				
Coke oven emissions				
Cyanide				
Dibenzo(a,h)anthracene				
1,4-Dichlorobenzene				
Dichlorobromomethane				
1,1-Dichloroethane				
1,2-Dichloroethane				
1,1-Dichloroethylene				
1,2-trans-Dichloroethylene				
2,4-Dichlorophenol				
1,2-Dichloropropane				
1,3-Dichloropropene				
Diethyl phthalate				
2,4-Dimethylphenol				
Dimethyl phthalate				
Di-n-butyl phthalate				
4,6-Dinitro-o-cresol				
2,4-Dinitrotoluene				
2,6-Dinitrotoluene				
1,2-Diphenylhydrazine				
Ethylbenzene				
Ethylchloride				
Ethylene oxide				
Fluoranthene				
Fluorene				
Fluoride				
Hexachlorobenzene				
Hydrogen chloride				
Hydrogen sulfide				
Indeno(1,2,3-cd)pyrene				
Isophorone				
Lindane				
Lead	1.67E-03	$2.67	1.69E-04	$0.27
Manganese				
Mercury				

POLLUTANTS	Virgin GLASS (lbs poll/ ton material)	Virgin GLASS ($/ton mat)	Recycled GLASS (lbs poll/ ton mat)	Recycled GLASS ($/ton mat)
Methane	3.00E-02	$0.00	1.00E-02	$0.00
Methylene chloride				
Napthalene				
Nickel				
Nitrobenzene				
Parachloronitrocresol				
Pentachlorophenol				
Phenanthrene				
Phenol				
Propylene				
Pyrene				
Selenium				
Silver				
Sodium hydroxide				
Styrene				
Sulfides				
2378-TCDD				
2378-TCDF				
Tetrachloroethylene				
Thallium				
Thiocyanates				
Toluene				
1,1,1-Trichloroethane				
Trichloroethylene				
Trichlorofluoromethane				
2,4,6-Trichlorophenol				
Triethanol				
Vanadium				
Vinyl chloride				
Xylenes				
Zinc				
TOTAL		**$85.29**		**$54.73**
($/ounce mat)		$0.003		$0.002

POLLUTANTS	Virgin ALUMINUM (lbs poll/ ton material)	Virgin ALUMINUM ($/ton mat)	Recycled ALUMINUM (lbs poll/ ton material)	Recycled ALUMINUM ($/ton mat)
CO	8.60E+00	$3.59	3.10E-01	$0.13
NOx	9.63E+01	$349.57	1.50E+00	$5.45
Particulates	1.03E+02	$602.55	5.14E+01	$300.69
SOx	9.31E+01	$546.50	9.20E-01	$5.40
VOCs	2.63E+00	$6.58	5.00E-02	$0.13
Acenapthene	2.37E-11	$0.00		
Acenapthylene				
Acetophenone				
Acrylonitrile				
Aluminum	8.47E-03			
Ammonia	1.46E-07	$0.00		
Anthracene				
Antimony	2.69E-03	$15.06		
Arsenic	2.15E-13	$0.00		
Benzene	5.17E-11	$0.00		
Benzo(a)anthracene				
Benzo(a)pyrene	4.83E-05			
3,4-Benzofluoranthene				
Benzo(k)flouranthene				
Benzo(ghi)perylene				
Benzoic acid				
Beryllium	8.61E-13	$0.00		
Biphenyl				
Bis(2-ethylhexyl) phthalate				
1,3-Butadiene				
2-Butanone				
Butyl benzyl phthalate				
Cadmium	5.38E-12	$0.00		
Carbon disulfide				
Carbon tetrachloride				
Chlorine	3.07E-01	$1.80		
Chlorobenzene				
Chloroethane				
Chloroform	6.67E-11	$0.00		
p-Chloro-m-cresol	6.46E-12	$0.00		
2-Chlorophenol				

POLLUTANTS	Virgin ALUMINUM (lbs poll/ ton material)	Virgin ALUMINUM ($/ton mat)	Recycled ALUMINUM (lbs poll/ ton material)	Recycled ALUMINUM ($/ton mat)
Chloroprene				
Chromium	2.49E-09	$0.00		
Chrysene	4.31E-13			
Copper	1.07E-03	$0.06		
Coke oven emissions				
Cyanide	9.79E-10	$0.00		
Dibenzo(a,h)anthracene				
1,4-Dichlorobenzene				
Dichlorobromomethane				
1,1-Dichloroethane				
1,2-Dichloroethane				
1,1-Dichloroethylene				
1,2-trans-Dichloroethylene				
2,4-Dichlorophenol	4.31E-12	$0.00		
1,2-Dichloropropane				
1,3-Dichloropropene				
Diethyl phthalate	3.23E-11	$0.00		
2,4-Dimethylphenol				
Dimethyl phthalate	2.15E-12	$0.00		
Di-n-butyl phthalate	8.61E-13			
4,6-Dinitro-o-cresol				
2,4-Dinitrotoluene				
2,6-Dinitrotoluene				
1,2-Diphenylhydrazine				
Ethylbenzene				
Ethylchloride				
Ethylene oxide				
Fluoranthene				
Fluorene				
Fluoride	7.45E+00	$278.07		
Hexachlorobenzene				
Hydrogen chloride				
Hydrogen sulfide				
Indeno(1,2,3-cd)pyrene				
Isophorone				
Lindane				
Lead	8.04E-02	$128.68	9.22E-04	$1.48
Manganese				
Mercury	1.94E-11	$0.00		

POLLUTANTS	Virgin ALUMINUM (lbs poll/ ton material)	Virgin ALUMINUM ($/ton mat)	Recycled ALUMINUM (lbs poll/ ton material)	Recycled ALUMINUM ($/ton mat)
Methane	3.36E-01	$0.01	2.00E-02	$0.00
Methylene chloride				
Napthalene				
Nickel	1.96E-03	$0.82		
Nitrobenzene				
Parachloronitrocresol				
Pentachlorophenol				
Phenanthrene	4.31E-12			
Phenol	3.87E-10	$0.00		
Propylene				
Pyrene	2.15E-12	$0.00		
Selenium	3.70E-10	$0.00		
Silver	8.61E-13	$0.00		
Sodium hydroxide				
Styrene				
Sulfides	1.29E-08	$0.00		
2378-TCDD				
2378-TCDF				
Tetrachloroethylene				
Thallium	6.89E-11	$0.00		
Thiocyanates				
Toluene	2.17E-10	$0.00		
1,1,1-Trichloroethane				
Trichloroethylene				
Trichlorofluoromethane				
2,4,6-Trichlorophenol				
Triethanol				
Vanadium				
Vinyl chloride				
Xylenes				
Zinc	2.25E-09	$0.00		
TOTAL		**$1,933.29**		**$313.27**
($/ounce mat)		$0.060		$0.010

POLLUTANTS	Virgin STEEL (lbs poll/ ton material)	Virgin STEEL ($/ton mat)	Recycled STEEL (lbs poll/ ton material)	Recycled STEEL ($/ton mat)
CO	3.74E-01	$0.16	3.57E-01	$0.15
NOx	2.42E+00	$8.78	2.35E+00	$8.53
Particulates	7.62E+00	$44.58	7.11E+00	$41.59
SOx	3.46E+00	$20.31	4.11E+00	$24.13
VOCs	4.50E-02	$0.11	3.86E-02	$0.10
Acenapthene				
Acenapthylene	1.09E-04		9.04E-05	
Acetophenone				
Acrylonitrile	1.10E-04	$0.05	9.15E-05	$0.04
Aluminum				
Ammonia	7.17E-01	$1.65	5.96E-01	$1.37
Anthracene				
Antimony	2.11E-04	$1.18	1.75E-04	$0.98
Arsenic	1.98E-04	$4.49	1.69E-04	$3.83
Benzene	1.00E-01	$2.51	8.35E-02	$2.09
Benzo(a)anthracene	3.76E-06		3.13E-06	
Benzo(a)pyrene				
3,4-Benzofluoranthene			3.18E-06	
Benzo(k)flouranthene				
Benzo(ghi)perylene				
Benzoic acid				
Beryllium				
Biphenyl				
Bis(2-ethylhexyl) phthalate				
1,3-Butadiene				
2-Butanone				
Butyl benzyl phthalate				
Cadmium	1.95E-04	$0.95	1.88E-04	$0.92
Carbon disulfide				
Carbon tetrachloride				
Chlorine				
Chlorobenzene				
Chloroethane				
Chloroform	4.06E-04	$0.05	3.97E-04	$0.05
p-Chloro-m-cresol				
2-Chlorophenol				

POLLUTANTS	Virgin STEEL (lbs poll/ ton material)	Virgin STEEL ($/ton mat)	Recycled STEEL (lbs poll/ ton material)	Recycled STEEL ($/ton mat)
Chloroprene				
Chromium	8.39E-03	$0.02	8.39E-03	$0.02
Chrysene	3.59E-05		2.98E-05	
Copper	1.40E-03	$0.08	1.36E-03	$0.08
Coke oven emissions	7.84E-01	$19.59	6.52E-01	$16.29
Cyanide	1.71E-02	$1.92	1.42E-02	$1.59
Dibenzo(a,h)anthracene				
1,4-Dichlorobenzene				
Dichlorobromomethane				
1,1-Dichloroethane				
1,2-Dichloroethane				
1,1-Dichloroethylene				
1,2-trans-Dichloroethylene				
2,4-Dichlorophenol	1.23E-05	$0.01	1.02E-05	$0.01
1,2-Dichloropropane				
1,3-Dichloropropene				
Diethyl phthalate				
2,4-Dimethylphenol	7.47E-04	$0.08	6.21E-04	$0.07
Dimethyl phthalate				
Di-n-butyl phthalate				
4,6-Dinitro-o-cresol	2.09E-07		1.74E-07	
2,4-Dinitrotoluene	2.26E-03	$1.32	1.88E-03	$1.10
2,6-Dinitrotoluene	6.07E-04	$0.36	5.04E-04	$0.30
1,2-Diphenylhydrazine				
Ethylbenzene	1.47E-04	$0.00	1.22E-04	$0.00
Ethylchloride				
Ethylene oxide				
Fluoranthene	8.86E-05	$0.00	7.36E-05	$0.00
Fluorene	1.27E-04	$0.01	1.05E-04	$0.01
Fluoride	1.73E-01	$6.46	1.46E-01	$5.45
Hexachlorobenzene				
Hydrogen chloride				
Hydrogen sulfide				
Indeno(1,2,3-cd)pyrene				
Isophorone	5.23E-06	$0.00	4.35E-06	$0.00
Lindane				
Lead	5.98E-02	$95.71	5.93E-02	$94.91
Manganese	9.27E-02	$1.04	8.10E-02	$0.91
Mercury				

POLLUTANTS	Virgin STEEL (lbs poll/ ton material)	Virgin STEEL ($/ton mat)	Recycled STEEL (lbs poll/ ton material)	Recycled STEEL ($/ton mat)
Methane	1.95E-02	$0.00	1.89E-02	$0.00
Methylene chloride				
Napthalene	6.38E-03	$3.57	5.30E-03	$2.97
Nickel	3.87E-03	$1.62	3.65E-03	$1.61
Nitrobenzene				
Parachloronitrocresol	1.14E-05		9.48E-06	
Pentachlorophenol	2.09E-06	$0.00	1.74E-06	$0.00
Phenanthrene				
Phenol	1.51E-02	$0.06	1.26E-02	$0.05
Propylene				
Pyrene	7.52E-05	$0.01	6.25E-05	$0.00
Selenium	2.12E-04	$0.16	1.81E-04	$0.14
Silver	2.70E-04	$0.20	2.60E-04	$0.19
Sodium hydroxide				
Styrene				
Sulfides	1.12E-01	$3.89	9.31E-02	$3.23
2378-TCDD				
2378-TCDF				
Tetrachloroethylene				
Thallium	1.10E-04	$3.52	1.10E-04	$3.52
Thiocyanates	1.82E-01		1.51E-01	
Toluene	2.59E-01	$1.93	2.15E-01	$1.61
1,1,1-Trichloroethane				
Trichloroethylene				
Trichlorofluoromethane				
2,4,6-Trichlorophenol				
Triethanol				
Vanadium	3.00E-04	$0.10	3.00E-04	$0.10
Vinyl chloride				
Xylenes	7.95E-04	$0.00	6.61E-04	$0.00
Zinc	3.29E-01	$3.69	3.27E-01	$3.66
TOTAL		**$230.16**		**$221.59**
($/ounce mat)		$0.007		$0.007

CHAPTER 4 - CASE STUDIES

4.1 INTRODUCTION

Based on discussion with New Jersey DEPE staff, we examined five products for the purpose of comparing the production and disposal impacts of their respective packaging materials. The products selected for study were soft drinks, juice, fast-food hamburgers, microwave dinners, and hardware. For each product we purchased, cleaned, and weighed the packaging used for the product. Next, we calculated the environmental impacts of material production and disposal based upon the information presented in the previous chapters.

Where possible, we included at least one package made from each material used in mass-marketed versions of the product in the U.S. By including as many different materials as possible in the most common size, we could examine the effect of varying the material while holding the package size constant. We also examined different package sizes to study the variation based on size. If a package could be made from recycled material, we studied that option as well. However, current regulations prohibit use of recycled paper or plastic for many packages which come into contact with food or beverages.

It is important to note that the production impacts calculated in this chapter only include the impacts of packaging material production; forming and filling processes have not been modeled. In addition, secondary packaging is not included in the impacts. This includes, for example, the six pack rings used for canned soda, or the plastic wrap surrounding six packs of aseptic packages. Finally, shipping containers are not included in this analysis.

Spreadsheets were developed to enhance the comparison of production and disposal impacts of packaging materials in each case study. Each product category is presented separately, with a complete description of each item purchased, and the preparation of each item prior to weighing.

4.2 SOFT DRINKS

We examined a range of package sizes and packaging materials for this study. The sizes ranged from 10 ounces to 2 liters, and the only sizes where direct comparisons were possible across materials were in the 1 pint (16 oz.) and 12 oz. sizes. The products purchased for the study are listed in Table 4.1.

In each case, the product was removed from its container, and the container was then rinsed thoroughly and allowed to dry, as were their respective lids. Following this step, each container was carefully weighed; the lid and (if easily removable) the label were weighed separately. In the event that the label was not easily removable and would have weighed less than 0.05 oz., the label was weighed as part of the bottle. The results of the weighing are shown in Table 4.1.

Table 4.2 shows the combined impacts of production and disposal. These impacts are given on a per package basis and on a per ounce of delivered soft drink basis (expressed as ¢/ounce). As this latter denominator allows comparison across packages, we have used this unit to compare containers.

On a per ounce basis, the value of production and disposal impacts ranges from 0.05¢ to 0.39¢, recycled aluminum having the lowest per ounce cost and virgin glass having the highest cost. PET is the only material that can be examined in several sizes - 2 liter, 1 liter, 1 pint, and 12 ounces. As expected, the production and disposal impacts increase with decreasing size due to the fact that the weight of material per ounce of product *increases* with decreasing size.

For the smaller soft drink containers there is greater variation in the types of packaging materials used. We can therefore explore the effect of packaging material and production and disposal impacts. Two types of one pint packages were included in this study - the glass bottle and PET bottle. The recycled glass bottle and PET bottle have similar costs while the virgin glass bottle has a higher production and disposal impact value.

The smallest soft drink containers in this study are the 12 ounce aluminum can, the 12 ounce PET bottle, and 10 ounce glass bottle. For container production from virgin materials, the glass bottle has the highest valued production and disposal impacts. The production and disposal values for PET and virgin aluminum are essentially similar. Including all containers in this analysis (both virgin and recycled-content), recycled aluminum has the lowest impacts; PET, virgin aluminum, and recycled glass have the next lowest impacts, while virgin glass is in the highest impact range.

This case study shows that while the 2 liter PET bottle (note that PET is the only material used to make larger sized containers) has the lowest per fluid ounce production and disposal impacts, this cost increases with decreasing size. However, even in the smaller sized soft drink containers, PET has one of the lowest production and disposal costs.

It is important to note that refillable glass bottles were not evaluated in this study. In another study, the production and disposal of this type of container was shown to produce less pollution than its counterparts.[1]

4.3 JUICE

The second product selected for analysis of its packaging was juice. Juice comes in a large variety of sizes, with each size packaged in a number of different materials. For the purposes of comparison, we examined juice in various sizes ranging from one half gallon to

6 ounces. Direct comparisons across materials were possible for the half gallon and one pint sizes. The products included in the analysis are summarized in Table 4.3

As with the soft drink samples, the products were removed from the packaging and the packaging was cleaned thoroughly and dried. Labels were removed in the cases where this was possible. If the labels were lighter than 0.05 oz., they were weighed as part of the bottle. The results of the weighing are listed in Table 4.3. For the aseptic package, we weighed the total package; but since it is a three-layer composite that cannot easily be separated into single-material layers, we contacted the manufacturer to determine the weights of these layers.[2]

The valuation of production and disposal impacts are shown in Tables 4.4 through 4.6. The impacts in these tables are given on a per package and per ounce of delivered juice basis.

For the larger sized juice containers shown in Table 4.4, the virgin glass and PET bottles have the highest per ounce valuation of production and disposal impacts. Recycled glass, virgin steel, and recycled steel have the next highest costs. Recycled and virgin steel have similar impacts due to the fact that the type of furnace used to produce steel for cans can only use a low percentage of recycled steel.[3] The paperboard carton and HDPE bottle have the lowest production and disposal impacts - less than 0.06¢ per ounce of product.

Table 4.5 values the production and disposal impacts for one pint juice containers. The results are similar to those obtained for the larger juice containers - virgin glass has the highest impacts, recycled glass has intermediate impacts, and the paperboard carton and HDPE container have the lowest impacts.

The last category of juice containers in this study is the single serving container. The impacts associated with these containers are shown in Table 4.6. While each container is not the same exact size, the impact of these containers can be compared on a per ounce of serving basis. The virgin aluminum and virgin glass containers have the highest valued impacts. Recycled glass, virgin steel, and recycled steel have intermediate impacts while the aseptic package and recycled aluminum can have the lowest impacts. It is interesting to note that the aseptic package, which has been banned in Maine and received poor publicity in general, does not have very high valued impacts; only recycled aluminum has lower per ounce impacts. However, the potentially complex package fabrication processes have not been included in our analysis.

Glass is the only container which can be compared across all package sizes. As expected, in general the impacts per ounce of juice increase with decreasing package size. Virgin glass has one of the highest environmental production and disposal costs in each size category. While this impact is reduced for recycled glass, the impacts of recycled glass tend to be between the lowest and highest environmental costs.

4.4 FAST-FOOD HAMBURGERS

The types of packaging used to deliver the fast-food hamburger to the public tend to take two forms, the "clamshell," and the paper wrapper. The clamshell is either a formed polystyrene or folded bleached coated paperboard container which encases the hamburger much as a clamshell encases a clam. Traditionally, McDonald's has used the polystyrene clamshell and Burger King has used the paperboard clamshell. Given the substantial concern over the production and disposal impacts of polystyrene, McDonald's has eliminated the clamshell packaging and is now using a simple coated paper wrapper.

In order to compare the full impact cost of each package, we decided to evaluate the packaging requirements for the same size burger at both Burger King and McDonald's, the two leading chains. The products, weights, and material content of the packages are listed in Table 4.7.

McDonald's paper wrapper presented us with an analytical dilemma: while we did not specifically model the type of paper used in these wrappers, we wanted to include the new wrapper in our study. The new paper wrapper is essentially identical to unbleached kraft paperboard, the major difference being that paperboard is thicker than paper. However, since the impacts are calculated on a weight basis, this difference in thickness (and therefore difference between weights of materials) is accounted for.

The results of this analysis, as shown in Table 4.8, support McDonald's move to the paper wrapper as being more environmentally responsible. This switch provides a decrease from 0.4¢ to 0.25¢, in environmental impacts of production and disposal. It is interesting to note that the coated bleached kraft paperboard clamshell has a higher production and disposal cost than that of the polystyrene clamshell. Polystyrene fares well when compared to the paper clamshell due both to its lighter weight, and to the fact that the bleaching step required to make bleached paperboard results in high air and water emissions. However, it should be noted that a major environmental concern in regard to polystyrene production, namely ozone depletion caused by ozone-depleting blowing agents, is not accounted for in our estimate of the environmental cost of production.

4.5 MICROWAVE DINNERS

We examined five different microwave packaging options for meals which contained roughly the same quantity of food. The five meals examined are summarized in Table 4.9. In preparation for the weighing of the packaging, all of the food was removed and the packaging was washed and dried thoroughly. It is difficult to make comparisons across these packaging options since each meal is likely to contain many different packaging materials which are designed to work in combination with each other and for which switching to alternative materials may not be suitable.

As can be seen in Table 4.10, the packages with the lowest production and disposal impacts are the simpler packages. These packages do not have all three package components - meal box, meal tray, and meal cover. The packages that have all three components have the higher production and disposal impacts. Thus, this case study shows the merits of source reduction as a way to minimize overall impacts.

4.6 HARDWARE

The final product for which we analyzed packaging was for hardware. It used to be the case that all nails, screws, nuts and bolts, hinges, and clasps were all sold in bulk without requiring a package. Now, most of these items are only offered for bulk sale in a few sizes and most often appear on store shelves in specially designed packaging. Hardware packaging, unlike the previous products' packaging we examined, often is designed so that the purchaser may see, and in some cases remove, the item in order to ensure that it is the appropriate size for the purchaser's needs.

For the purpose of this study, we examined the packaging of all-purpose screws. Three types of screw packages were evaluated, as well as two bag options for the bulk purchase of a similar quantity of screws. The products, packaging type, materials and weights are listed in Table 4.11.

The results of the analysis, as shown in Table 4.12, points out that there are clear environmental advantages to not utilizing PVC in the packaging of hardware. The evaluated impacts of production and disposal for the two packages using PVC were 1.8¢ for the peel-back 16 screw pack, and 7.4¢ for the PVC container holding 100 screws, as opposed to a range of 0.19¢ to 0.30¢ for other packaging options (excluding bulk purchases).

With respect to the difference between bulk purchases versus packaged purchases, there is no significant advantage to bulk purchases if the PVC packaging options are omitted. Naturally, the bulk purchase packaging has even lower production and disposal valued impacts with respect to packaged purchases as the quantities purchased increase. For example, one could just as easily place 1000 screws in the paper bag incurring the same full cost, whereas one would need a much larger box to contain 1000 screws with the packaged purchase option.

4.7 CONCLUSIONS

Table 4.13 summarizes the findings of the five case studies - the environmental cost associated with the production and disposal of various packages required for a unit of various products. For soft drinks packaged in containers manufactured from virgin materials, PET has the lowest environmental cost (based upon cost per ounce of product). Recycled glass has environmental costs similar to PET, while recycled aluminum has the lowest environmental cost.

Several packaging sizes were examined for juice packaging - 1/2 gallon, one pint, and single serving (ranking from 11.5 to 6 ounces). For the two larger serving sizes, paperboard cartons and HDPE bottles have the lowest environmental production and disposal costs. These two types of packaging materials are not used for the smaller sized packages. For single serving packages, the recycled aluminum can and aseptic package have the lowest environmental costs while the virgin aluminum can has the highest environmental cost. Aluminum spans both ends of the price spectrum due to the fact that virgin aluminum production emits many pollutants while recycling of aluminum, which entails melting and reforming the aluminum, eliminates pollutant-producing manufacturing stages.

The third product examined in this study was fast food hamburger packaging. This case study showed that the new paper wrapper being used by McDonald's to wrap its quarter pound hamburger incurs a lower environmental cost of production and disposal than the polystyrene clamshell the hamburger chain recently abandoned.

Microwave dinners were an interesting category of packaging to examine - this product has many package components. The case study shows the benefits of minimizing the amount of packaging used as those dinners with the lowest environmental production and disposal costs are the ones that use less layers of packaging. The highly packaged dinners have a meal box, meal tray, and meal cover while the simpler dinners have no meal tray.

The final case study examined was hardware. The important feature of this case study is that it was the only case study which examined PVC. As shown in Table 4.13, there are clear environmental advantages to not utilizing PVC in hardware packaging. This case study argues for replacing PVC with other packaging materials.

To the best of our knowledge, this study is the first to incorporate a methodology to evaluate various pollutants associated with packaging material production and disposal. As such, we expect that the methodology employed in this study will evolve over the next several years, especially due to the U.S. Environmental Protection Agency's work in lifecycle assessment methodology.

4.8 ENDNOTES FOR CHAPTER 4

1. Sellers, V.R. and Sellers, J.D., 1989. *Comparative Energy and Environmental Impacts for Soft Drink Delivery Systems*, prepared for The National Assoc. for Plastic Container Recovery by Franklin Assoc., March.

2. Edward A. Klein, Tetra Pak Inc., March 1991, personal communication.

3. This is discussed in our previous report *Inventory of Material and Energy Use & Air and Water Emissions from the Production of Packaging Materials*, Tellus Institute.

Table 4.1 Products Included in Soft Drink Case Study

	Container		Lid		Label	
Item	**Weight (oz)**	**Material**	**Weight (oz)**	**Material**	**Weight (oz)**	**Material**
2 liter	1.85/0.75	PET/HDPE	0.10	Aluminum	**	
1 liter	1.45/0.30	PET/HDPE	0.10	Aluminum	**	
16 ounces	8.45	Glass	0.10	Aluminum	0.10	Paper
16 ounces	1.10	PET	0.10	PP	**	
12 ounces	0.60	Aluminum	NA	NA		
12 ounces	0.90	PET	0.10	PP	**	
10 ounces	7.65	Glass	0.10	Steel	0.05	Paper

Note:
NA = not applicable
PP = polypropylene
** Labels weighed with container.

Table 4.2 Soft Drink Packaging : Valuation of Emissions from Production and Disposal

		2 Liter PET Bottle Alum Lid		1 Liter PET Bottle Alum Lid		1 Pint Glass Bottle Paper Label Alum Lid			
								Recycled	Recycled
Materials	cents/oz	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit
HDPE	1.68	0.75	1.26	0.30	0.50				
LDPE	1.81								
PET	3.46	1.85	6.41	1.45	5.02				
PP	1.88								
PS	1.94								
PVC	16.53								
Bleached Kraft Paperboard	1.38					0.10	0.14	0.10	0.14
Unbleached Coated Folded Boxboard	1.19								
Linerboard	1.23								
Corrugated Medium	0.64								
Unbleached Kraft Paperboard	1.22								
Folding boxboard from Wastepaper	0.77								
Linerboard from Wastepaper	0.80								
Corrugated Medium from Wastepaper	0.95								
Glass	0.49					8.45	4.15		
Recycled Glass	0.40							8.45	3.35
Aluminum	6.13	0.10	0.61	0.10	0.61	0.10	0.61	0.10	0.61
Recycled Aluminum	1.07								
Steel	1.15								
Recycled Steel	1.12								
Total		2.70	8.28	1.85	5.64	8.65	4.90	8.65	4.10
Total value of emissions (cents) per ounce of product			0.12		0.17		0.31		0.26

Table 4.2 Soft Drink Packaging : Valuation of Emissions from Production and Disposal (cont.)

		1Pint PET Bottle Plastic Lid (PP)		12 oz Alum Can				12 oz. PET Bottle Plastio Lid (PP)	
Materials	cents/oz	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit	Recycled oz/unit	Recycled cents/mat/unit	oz/unit	cents/mat/unit
HDPE	1.68								
LDPE	1.81								
PET	3.46	1.10	3.81					0.90	3.12
PP	1.88	0.10	0.19					0.10	0.19
PS	1.94								
PVC	16.53								
Bleached Kraft Paperboard	1.38								
Unbleached Coated Folded Boxboard	1.19								
Linerboard	1.23								
Corrugated Medium	0.64								
Unbleached Kraft Paperboard	1.22								
Folding boxboard from Wastepaper	0.77								
Linerboard from Wastepaper	0.60								
Corrugated Medium from Wastepaper	0.95								
Glass	0.49								
Recycled Glass	0.40								
Aluminum	6.13			0.60	3.68				
Recycled Aluminum	1.07					0.60	0.64		
Steel	1.15								
Recycled Steel	1.12								
Total		1.20	4.00	0.60	3.68	0.60	0.64	1.00	3.31
Total value of emissions (cents) per ounce of product			0.25		0.31		0.05		0.28

Table 4.2 Soft Drink Packaging : Valuation of Emissions from Production and Disposal (cont.)

		10 oz. Glass Bottle Paper Label Steel Lid			
				Recycled	Recycled
Materials	cents/oz	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit
HDPE	1.68				
LDPE	1.81				
PET	3.46				
PP	1.88				
PS	1.94				
PVC	16.53				
Bleached Kraft Paperboard	1.38	0.05	0.07	0.05	0.07
Unbleached Coated Folded Boxboard	1.19				
Linerboard	1.23				
Corrugated Medium	0.64				
Unbleached Kraft Paperboard	1.22				
Folding boxboard from Wastepaper	0.77				
Linerboard from Wastepaper	0.80				
Corrugated Medium from Wastepaper	0.95				
Glass	0.49	7.65	3.76		
Recycled Glass	0.40			7.65	3.03
Aluminum	6.13				
Recycled Aluminum	1.07				
Steel	1.15	0.10	0.11	0.10	0.11
Recycled Steel	1.12				
Total		7.80	3.94	7.80	3.21
Total value of emissions (cents) per ounce of product			0.39		0.32

Table 4.3 Products Included in Juice Case Study

	Container		Lid		Label	
Item	Weight (oz)	Material	Weight (oz)	Material	Weight (oz)	Material
1/2 gallon	24.1	Glass	0.25	Steel	**	
1/2 gallon	3.15	PET	0.20	PP	0.25	Paper
1/2 gallon	2.45	Paperboard	NA		NA	
1/2 gallon	1.60	HDPE	0.10	HDPE	**	
46 ounces	5.65	Steel	NA		0.20	Paper
16 ounces	9.95	Glass	0.15	Steel	0.05	PS
16 ounces	0.90	Paperboard	NA		NA	
16 ounces	0.80	HDPE	0.10	HDPE	**	
11.5 ounces	0.60	Aluminum	NA		NA	
10 ounces	5.35	Glass	0.15	Steel	0.05	PS
8.45 ounce aseptic	0.45	*	NA		NA	
6 ounce	1.10	Steel	NA		0.05	Paper

Notes:

NA = not applicable

* Aseptic package consists of 0.09 oz LDPE, 0.28 oz paperboard, and 0.03 oz aluminum, with one 0.05 ounce polypropylene straw.

** Labels weighed with container.

Table 4.4 Juice Packaging - Half Gallon (64 oz.) and 46 Ounce: Valuation of Emissions from Production and Disposal

		Half Gallon Glass Bottle Steel Lid				Half Gallon Bottle (PET) Plastic Lid Paper Label	
Materials	cents/oz	oz/unit	cents/mat/unit	Recycled oz/unit	Recycled cents/mat/unit	oz/unit	cents/mat/unit
HDPE	1.68						
LDPE	1.81						
PET	3.46					3.15	10.91
PP	1.88					0.20	0.38
PS	1.94						
PVC	16.53						
Bleached Kraft Paperboard	1.38					0.25	0.35
Unbleached Coated Folded Boxboard	1.19						
Linerboard	1.23						
Corrugated Medium	0.64						
Unbleached Kraft Paperboard	1.22						
Folding boxboard from Wastepaper	0.77						
Linerboard from Wastepaper	0.80						
Corrugated Medium from Wastepaper	0.95						
Glass	0.49	24.10	11.84				
Recycled Glass	0.40			24.10	9.54		
Aluminum	6.13						
Recycled Aluminum	1.07						
Steel	1.15	0.25	0.29	0.25	0.29		
Recycled Steel	1.12						
Total		24.10	12.13	24.10	9.83	3.60	11.63
Total value of emissions (cents) per ounce of product			0.19		0.15		0.18

Table 4.4 Juice Packaging - Half Gallon (64 oz.) and 46 Ounce: Valuation of Emissions from Production and Disposal (cont.)

		Half Gallon Paperboard Carton		Half Gallon Plastic Bottle-HDPE Plastic Lid - HDPE		46 ounces Steel Can Paper Label			
Materials	cents/oz	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit	Recycled oz/unit	Recycled cents/mat/unit
HDPE	1.68			1.70	2.85				
LDPE	1.81	0.37	0.67						
PET	3.46								
PP	1.88								
PS	1.94								
PVC	16.53								
Bleached Kraft Paperboard	1.38	2.08	2.88			0.20	0.28	0.20	0.28
Unbleached Coated Folded Boxboard	1.19								
Linerboard	1.23								
Corrugated Medium	0.64								
Unbleached Kraft Paperboard	1.22								
Folding boxboard from Wastepaper	0.77								
Linerboard from Wastepaper	0.80								
Corrugated Medium from Wastepaper	0.95								
Glass	0.49								
Recycled Glass	0.40								
Aluminum	6.13								
Recycled Aluminum	1.07								
Steel	1.15					5.65	6.47		
Recycled Steel	1.12							5.65	6.32
Total		2.45	3.55	1.70	2.85	5.85	6.75	5.85	6.60
Total value of emissions per ounce of product			0.06		0.04		0.15		0.14

Table 4.5 Juice Packaging - One Pint (16 oz.) : Valuation of Emissions from Production and Disposal

		Glass Bottle Polystyrene Label Steel Lid				Paperboard Carton		Bottle HDPE Lid HDPE	
Materials	cents/oz	oz/unit	cents/mat/unit	Recycled oz/unit	Recycled cents/mat/unit	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit
HDPE	1.68							0.90	1.51
LDPE	1.81					0.14	0.24		
PET	3.46								
PP	1.88								
PS	1.94	0.05	0.10	0.06	0.10				
PVC	16.53								
Bleached Kraft Paperboard	1.38					0.77	1.06		
Unbleached Coated Folded Boxboard	1.19								
Linerboard	1.23								
Corrugated Medium	0.64								
Unbleached Kraft Paperboard	1.22								
Folding boxboard from Wastepaper	0.77								
Linerboard from Wastepaper	0.80								
Corrugated Medium from Wastepaper	0.95								
Glass	0.49	9.95	4.89						
Recycled Glass	0.40			9.95	3.94				
Aluminum	6.13								
Recycled Aluminum	1.07								
Steel	1.15	0.15	0.17	0.15	0.17				
Recycled Steel	1.12								
Total		10.15	5.16	10.15	4.21	0.90	1.30	0.90	1.51
Total value of emissions (cents) per ounce of product			0.32		0.26		0.08		0.09

Table 4.6 Juice Packaging - Single Serving: Valuation of Emissions from Production and Disposal

		11.5 ounces Aluminum Can				10 ounces Glass Bottle Polystyrene Label Steel Lid			
Materials	cents/oz	oz/unit	cents/mat/unit	Recycled oz/unit	Recycled cents/mat/unit	oz/unit	cents/mat/unit	Recycled oz/unit	Recycled cents/mat/unit
HDPE	1.68								
LDPE	1.81								
PET	3.46								
PP	1.88								
PS	1.94					0.05	0.10	0.05	0.10
PVC	16.53								
Bleached Kraft Paperboard	1.38								
Unbleached Coated Folded Boxboard	1.19								
Linerboard	1.23								
Corrugated Medium	0.64								
Unbleached Kraft Paperboard	1.22								
Folding boxboard from Wastepaper	0.77								
Linerboard from Wastepaper	0.80								
Corrugated Medium from Wastepaper	0.95								
Glass	0.49					5.35	2.63		
Recycled Glass	0.40							5.35	2.12
Aluminum	6.13	0.60	3.68						
Recycled Aluminum	1.07			0.60	0.64				
Steel	1.15					0.15	0.17	0.15	0.17
Recycled Steel	1.12								
Total		0.60	3.68	0.60	0.64	5.40	2.90	5.40	2.39
Total value of emissions (cents) per ounce of product			0.32		0.06		0.29		0.24

Table 4.6 Juice Packaging - Single Serving: Valuation of Emissions from Production and Disposal (cont.)

		8.45 ounces Aseptic Package		6 ounces Steel Can Paper Label			
Materials	cents/oz	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit	Recycled oz/unit	Recycled cents/mat/unit
HDPE	1.68						
LDPE	1.81	0.09	0.17				
PET	3.46						
PP	1.88	0.05	0.09				
PS	1.94						
PVC	16.53						
Bleached Kraft Paperboard	1.38	0.28	0.39	0.05	0.07	0.05	0.07
Unbleached Coated Folded Boxboard	1.19						
Linerboard	1.23						
Corrugated Medium	0.64						
Unbleached Kraft Paperboard	1.22						
Folding boxboard from Wastepaper	0.77						
Linerboard from Wastepaper	0.80						
Corrugated Medium from Wastepaper	0.95						
Glass	0.49						
Recycled Glass	0.40						
Aluminum	6.13	0.03	0.17				
Recycled Aluminum	1.07						
Steel	1.15			1.10	1.26		
Recycled Steel	1.12					1.10	1.23
Total		0.45	0.82	1.15	1.33	1.15	1.30
Total value of emissions (cents) per ounce of product			0.10		0.22		0.22

Table 4.7 Products Included in Hamburger Case Study

Item	Container Weight (oz)	Container Material
1/4 Pound in Clamshell	0.20	PS
1/4 Pound in Paper	0.03	LDPE
	0.16	Unblchd. Kraft
1/4 Pound in Clamshell	0.08	LDPE
	0.52	Blchd. Kraft

Table 4.8 Fastfood Packaging - Quarter Pound Hamburger Containers: Valuation of Emissions from Production and Disposal

		Clamshell: Bleached Coated Folded Boxboard		Clamshell: Polystyrene		Paper Wrapper	
Materials	cents/oz	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit
HDPE	1.68						
LDPE	1.81	0.08	0.14			0.03	0.06
PET	3.46						
PP	1.88						
PS	1.94			0.20	0.40		
PVC	16.53						
Bleached Kraft Paperboard	1.38	0.52	0.71				
Unbleached Coated Folded Boxboard	1.19						
Linerboard	1.23						
Corrugated Medium	0.64						
Unbleached Kraft Paperboard	1.22					0.16	0.19
Folding boxboard from Wastepaper	0.77						
Linerboard from Wastepaper	0.80						
Corrugated Medium from Wastepaper	0.95						
Glass	0.49						
Recycled Glass	0.40						
Aluminum	6.13						
Recycled Aluminum	1.07						
Steel	1.15						
Recycled Steel	1.12						
Total value of emissions (cents) per product			0.86		0.40		0.25

Table 4.9 Products Included in Microwave Dinners Case Study

Item	Meal box		Meal Tray		Meal Cover	
	Weight (oz)	Material	Weight (oz)	Material	Weight (oz)	Material
9 ounce meal	0.85	Blchd. PB	NA		NA	
	0.10	PET				
8.5 ounce meal	0.75	Blchd. PB	NA		0.30	LDPE
9 ounce meal	0.80	Unblchd. PB	0.85	PET	too light to determine	
9 ounce meal	1.15	Unblchd. PB	0.80	PB	0.10	Aluminum
8.65 ounce meal	1.40	Blchd. PB	1.35	PET	0.10	LDPE

Notes:
Blchd. PB = bleached kraft paperboard
Unblchd. PB = unbleached kraft paperboard
NA = not applicable

Table 4.10 Microwave Dinner Packaging: Valuation of Emissions from Production and Disposal

		Plasticoated Bleached Paperboard Box (Box is meal tray)		Plasticoated Bleached Paperboard Box Plastic Food Pouches	
Materials	cents/oz	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit
HDPE	1.68				
LDPE	1.81			0.40	0.73
PET	3.46	0.10	0.35		
PP	1.88				
PP	1.94				
PVC	16.53				
Bleached Kraft Paperboard	1.38	0.85	1.18	0.65	0.90
Unbleached Coated Folded Boxboard	1.19				
Linerboard	1.23				
Corrugated Medium	0.64				
Unbleached Kraft Paperboard	1.22				
Folding boxboard from Wastepaper	0.77				
Linerboard from Wastepaper	0.80				
Corrugated Medium from Wastepaper	0.95				
Glass	0.49				
Recycled Glass	0.40				
Aluminum	6.13				
Recycled Aluminum	1.07				
Steel	1.15				
Recycled Steel	1.12				
Total		0.95	1.52	1.05	1.62
Total value of emissions (cents) per ounce of product			0.17		0.19

Table 4.10 Microwave Dinner Packaging: Valuation of Emissions from Production and Disposal (cont.)

		Unbleached Plasticoated Paperboard Box Alum Foil Meal Cover Plasticoated Bleached Paperboard Meal Tray		Unbleached Plasticoated Paperboard Box PET meal tray Film Plastic Meal Cover		Plasticoated Bleached Paperboard Box HDPE meal tray Plastic Film Meal Wrappings	
Materials	cents/oz	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit	oz/unit	cents/mat/unit
HDPE	1.68						
LDPE	1.81	0.11	0.19			0.29	0.52
PET	3.46			0.85	2.94	1.35	4.68
PP	1.88						
PP	1.94						
PVC	16.53						
Bleached Kraft Paperboard	1.38	0.69	0.96			1.21	1.68
Unbleached Coated Folded Boxboard	1.19	1.15	1.37	0.80	0.95		
Linerboard	1.23						
Corrugated Medium	0.64						
Unbleached Kraft Paperboard	1.22						
Folding boxboard from Wastepaper	0.77						
Linerboard from Wastepaper	0.80						
Corrugated Medium from Wastepaper	0.95						
Glass	0.49						
Recycled Glass	0.40						
Aluminum	6.13	0.10	0.61				
Recycled Aluminum	1.07						
Steel	1.15						
Recycled Steel	1.12						
Total		2.05	3.14	1.65	3.90	2.85	6.87
Total value of emissions (cents) per ounce of product			0.35		0.43		0.79

Table 4.11 Products Included in Hardware Case Study

Item	Package	Material	Weight (oz)
100 Screws	Box	Unbleached kraft paperboard	0.25
16 Screws	Peelback front	Unbleached kraft paperboard	0.10
	Pack	PVC	0.10
100 Screws	Blister pack	PVC	0.45
Bulk Packaging Options			
	3"x4" ziplock	LDPE	0.05
	4"x8" paper bag	Unbleached kraft paperboard	0.25

Table 4.12 Hardware Packaging - Screws: Valuation of Emissions from Production and Disposal

		100 Machine Screws weight 3.75 oz. Unbleached Coated Folded Boxboard				100 All-Purpose Screws weight 5.70 oz. Snap Open Plastic Container	
Materials	cents/oz	oz/unit	cents/mat/unit	Recycled oz/unit	Recycled cents/mat/unit	oz/unit	cents/mat/unit
HDPE	1.68						
LDPE	1.81						
PET	3.46						
PP	1.88						
PS	1.94						
PVC	16.53					0.45	7.44
Bleached Kraft Paperboard	1.38						
Unbleached Coated Folded Boxboard	1.19	0.25	0.30				
Linerboard	1.23						
Corrugated Medium	0.64						
Unbleached Kraft Paperboard	1.22						
Folding boxboard from Wastepaper	0.77			0.25	0.19		
Linerboard from Wastepaper	0.80						
Corrugated Medium from Wastepaper	0.95						
Glass	0.49						
Recycled Glass	0.40						
Aluminum	6.13						
Recycled Aluminum	1.07						
Steel	1.15						
Recycled Steel	1.12						
Total value of emissions (cents) per product			0.30		0.19		7.44

Table 4.12 Hardware Packaging - Screws: Valuation of Emissions from Production and Disposal (cont.)

Materials	cents/oz	16 All-Purpose Screws (weight 0.95 oz.) Plasticoated Unbleached Paperboard Plastic Peelback Front				3" by 4" Plastic Zip-Lock Bag		Virgin 4" by 8" Paperbag	
		oz/unit	cents/mat/uni	Recycled oz/unit	Recycled cents/mat/unit	oz/unit	cents/mat/unit	oz/unit	cents/mat/un
HDPE	1.68								
LDPE	1.81					0.05	0.09		
PET	3.46								
PP	1.88								
PS	1.94								
PVC	16.53	0.10	1.65	0.10	1.65				
Bleached Kraft Paperboard	1.38								
Unbleached Coated Folded Boxboard	1.19	0.10	0.12						
Linerboard	1.23								
Corrugated Medium	0.64								
Unbleached Kraft Paperboard	1.22							0.25	0.30
Folding boxboard from Wastepaper	0.77			0.10	0.08				
Linerboard from Wastepaper	0.80								
Corrugated Medium from Wastepaper	0.95								
Glass	0.49								
Recycled Glass	0.40								
Aluminum	6.13								
Recycled Aluminum	1.07								
Steel	1.15								
Recycled Steel	1.12								
Total value of emissions (cents) per product			1.77		1.73		0.09		0.30

Table 4.13 Packaging Case Studies - Summary Cost Comparisons

Product	Size	Material	Environmental cost cents/unit	Unit
SOFT DRINK	2 liter	PET	0.12	fluid
	1 liter	PET	0.17	ounce
	1 pint	Virgin glass	0.31	
	1 pint	Recycled glass	0.26	
	1 pint	PET	0.25	
	12 ounce	virgin aluminum	0.31	
	12 ounce	recycled aluminum	0.05	
	12 ounce	PET	0.28	
	10 ounce	virgin glass	0.39	
	10 ounce	recycled glass	0.32	
JUICE	1/2 gallon	virgin glass	0.19	fluid
	1/2 gallon	recycled glass	0.15	ounce
	1/2 gallon	PET	0.18	
	1/2 gallon	paperboard carton	0.06	
	1/2 gallon	HDPE	0.04	
	46 ounces	virgin steel	0.15	
	46 ounces	recycled steel	0.14	
	1 pint	virgin glass	0.32	
	1 pint	recycled glass	0.26	
	1 pint	paperboard carton	0.08	
	1 pint	HDPE	0.09	
	11.5 ounce	virgin aluminum	0.32	
	11.5 ounce	recycled aluminum	0.06	
	10 ounce	virgin glass	0.29	
	10 ounce	recycled glass	0.24	
	8.5 ounce	aseptic packaging	0.10	
	6 ounce	virgin steel	0.22	
	6 ounce	recycled steel	0.22	
FAST FOOD BURGERS	clamshell	boxboard	0.86	quarter-pound hamburger
	clamshell	polystyrene	0.40	
	wrapper	paper	0.25	
MICROWAVE DINNERS	no-tray dinner	paperboard	0.17	ounce of food
	no-tray dinner	paperboard, pouches	0.19	
	light tray	boxboard, paperboard	0.35	
	light tray	boxboard, PET	0.43	
	heavy tray	paperboard, HDPE	0.79	
HARDWARE	box	virgin boxboard	0.30	100 screws/nails
	box	recycled boxboard	0.19	
	plastic container	PVC	7.44	
	blisterpack	PVC, paperboard	1.77	16 screws
	blisterpack	PVC, rec. paperboard	1.73	
	plastic bag	LDPE	0.09	
	paper bag	paperboard	0.30	