OCCASION

This publication has been made available to the public on the occasion of the 50th anniversary of the United Nations Industrial Development Organisation.

DISCLAIMER

FAIR USE POLICY

CONTACT

Please contact publications@unido.org for further information concerning UNIDO publications.

For more information about UNIDO, please visit us at www.unido.org

UNITED NATIONS INDUSTRIAL DEVELOPMENT ORGANIZATION
Vienna International Centre, P.O. Box 300, 1400 Vienna, Austria
Tel: (+43-1) 26026-0 · www.unido.org · unido@unido.org

20858

TELLUS INSTITUTE
for Resource and Environmental Strategies

93-157A

EVALUATION OF THE ENVIRONMENTAL IMPACT OF PACKAGING PRODUCTION FOR MEXICO

a final report to

United Nations International Development Organization
Project number SF/MEX/94/001
<u>Lifecycle Analysis and Legislation for Packaging Materials in Mexico</u>
Contract number 94/030

Frank Ackerman
Brian Zuckerman

Tellus Institute
November 1994

11 Arlington Street, Boston, MA 02116-3411 • Tel: 617-266-5400 • Fax: 617-266-8303

Table of Contents

Introduction

This report provides an evaluation, or impact assessment, of the emissions from production of packaging used in Mexico. The emissions data on which the assessment is based can be found in the Franklin Associates report, *Life Cycle Inventory of Packaging Materials in Mexico* (LCI).

There are four chapters in this report. Chapter 1, on impact assessment methods, reviews existing methods and explains the reasons for the adoption of the Tellus Institute *Packaging Study* methodology. Chapter 2 discusses the Tellus methodology in more detail. Chapter 3 applies the Tellus methodology, leading to calculation of impacts per tonne of packaging material and impacts per package. Chapter 4 begins by briefly recapitulating chapters 2 and 3, and then discusses the significance of our results.

I. Impact Assessment Methods

A. Why is impact assessment needed?

Lifecycle analysis attempts to measure the full range of environmental impacts caused by production, use and disposal of products. Efforts to standardize lifecycle methods have been made by the Society for Environmental Toxicology and Analytical Chemistry (SETAC). In SETAC's typology, there are three principal stages of analysis:[1]

- lifecycle *inventory*, consisting of documentation of the levels of emissions, material and energy requirements, and wastes (in this study, the Franklin Associates LCI);
- lifecycle *impact assessment*, consisting of evaluation of the inventory results; and
- *improvement analysis*, making recommendations for change based on results of the first two stages.[2]

Why should the second stage, evaluation of lifecycle impacts, be included? The purpose of a lifecycle study is to advance the state of understanding of the products or processes in question, and to improve public and private sector decision-making (the third stage, or improvement analysis, in the SETAC typology). Useful information for decision-making cannot be obtained from a lifecycle inventory alone: the results of a typical lifecycle inventory contain hundreds of times as much information as decision-makers are likely to absorb. Therefore, in the absence of an assessment, the inventory results will either be ignored,

---

[1] In recent SETAC discussion, definition of goals, boundaries, and assumptions of the study is sometimes described as a separate, preliminary stage.

[2] US EPA, *Life-Cycle Impact Assessment: A Conceptual Framework, Key Issues, and Summary of Existing Methods*, June 1994, page 1-1.

or will be compressed and digested -- in effect, evaluated -- by someone who summarizes them for review by decision-makers. Lifecycle impact assessment makes this valuation explicit rather than implicit.

When formal assessment is lacking, the implicit evaluation of inventory results is often based on the simplest possible rule: add up the quantities of all pollutants emitted. This does not avoid the evaluation process. Rather, it applies a strict, and obviously erroneous, standard of evaluation: namely, that a gram of every pollutant is exactly as important as a gram of every other pollutant. No scientific researcher would defend the notion that all the pollutants found in lifecycle inventories are equally harmful. Yet publication of inventory results without evaluation may give exactly this impression to nontechnical readers.

The preferable alternative is to explicitly engage in the comparison and evaluation of different emissions. When the evaluation criteria are explicit and visible, they can be subjected to review, debate, and revision. Once evaluation criteria have been accepted, detailed lifecycle inventory results can be applied systematically in the development of public policy.

B. Review of existing impact assessment methods

There are at least four approaches to impact assessment that provide quantitative evaluation and comparison of dissimilar emissions. They are:

- Qualitative scoring systems
- EPS Enviro-Accounting Method
- Swiss Eco-Factor Method
- Tellus US Packaging Study Approach

Qualitative scoring systems[3]

Impact assessment methodologies can be very complex, requiring analysis of relative impacts of many pollutants. Simplification appears desirable, particularly if it allows the analyst to avoid evaluation of intricate details in favor of broad categorical judgments. One popular approach to simplification is to use a qualitative scoring system, for example assigning observed pollutant emission levels to low, medium, or high risk categories. Many versions of such scoring systems can be found in the literature.

However, assignment of low, medium and high risks only appears to simplify the task, and may introduce inappropriate discontinuities in evaluation. The reason is that qualitative scoring systems necessarily attach great importance to the boundaries between categories. Suppose that emissions of 0-20 g of a pollutant are called "low risk", 20-40 g "medium", and more than 40 g "high". Then observed emissions of 19 and 21 g are evaluated quite

[3] See *Life-Cycle Impact Assessment*, pages 4-1 to 4-6 for examples.

differently, while 21 and 39 g are evaluated identically.

This approach, with its discontinuities at the category boundaries, could only be justified if the boundaries correspond to scientifically important threshold levels, which is rarely the case. In qualitative scoring systems, the complexity of evaluation is still present, in hidden form, in the definition of the impact categories: frequently it is impossible to define them in a rigorous, internally consistent manner.

EPS Enviro-Accounting Method[3]

The EPS Enviro-Accounting method, created by the Swedish Environmental Research Institute, attempts to account for environmental damage to five "safeguard" areas - human health, biodiversity, production, natural resources, and aesthetic values. Moreover, it takes a broad view of "human health": it attempts to characterize the health effects of global warming and ozone depletion, in addition to toxicity effects of chemicals. Finally, it takes a global perspective. The entire population of Earth is considered when discussing possible environmental effects. The time duration of effects is also considered, so that the damage resulting from the loss of a species is counted for millions of years, whereas the persistence of damage from ozone depletion effects accrues for one hundred years, and the damage from toxic chemicals for only a single year.

The ambitious scope of this effort has placed severe constraints on the method in practice. First of all, very few pollutants have been analyzed to date. This may be due to the difficulty of analyzing less-traditional subjects such as biodiversity, global warming and ozone depletion; further upgrades of the system may include a wider range of toxic emissions. The second constraint is the valuation method used. The Enviro-Accounting Method depends upon willingness-to-pay studies, which at times diverge substantially from the amounts society actually pays to remedy environmental damage. Finally, review of the extensive, detailed techniques for assignment of impacts to pollutants indicates that the method includes many rigorous calculations, but also many poorly justified assumptions.

The EPS Enviro-Accounting Method is a bold attempt to include all possible environmental effects of human activities in one simple spreadsheet-based system. If developed further, it may ultimately prove to be of great value. At its present level of development, however, it is unsuited for analyzing the effects of manufacturing processes.

Eco-Factor Method[4]

The Eco-factor method, developed in Switzerland, uses regulatory thresholds for

[3]Swedish Environmental Research Institute, *The EPS Enviro-Accounting Method*, December 1992.

[4]Jan A. Assies, "Introduction Paper to SETAC-Europe Workshop on Environmental Life Cycle Analysis of Products, December 2-3, 1991, Leiden, Netherlands, pages 3, 22.

maximum allowed concentrations as a basis for comparison of pollutants. The ranking score for each pollutant is the volume of air or water required in order to dilute an emission of 1 gram down to the allowed concentration. For example, if pollutant X had a maximum allowed concentration of 10 grams/m^3, and pollutant Y had a maximum allowed concentration of 25 grams/m^3, then one can conclude that pollutant X is 2.5 times as hazardous as pollutant Y. The ranking system based on this approach can then be used to evaluate the hazards of pollutant emissions.

This approach can be applied to many more pollutants than the EPS approach, and yields a greater quantity of useful information than qualitative scoring methods. The problem with the approach, however, is that it depends completely upon detailed regulatory information, which may change from country to country -- and may be far from complete in any one country. Dozens of pollutants are potentially involved in lifecycle assessments, and it is not certain that any country has a complete set of maximum allowed concentrations for all of them.

Tellus Packaging Study Method[6]

The Tellus *Packaging Study*, a lifecycle assessment of packaging production in the United States, includes a ranking system which relies on a combination of pollution control costs and a health or damage effects ranking to assess the impacts of emissions. A monetary value is assigned to each pollutant emission; these values are summed to estimate the total impact of each production process. Different methods are used to evaluate different categories of pollutants, but all rely on pollution control costs, directly or indirectly.

In the Tellus methodology, pollution control costs are used as a measure of the value society places on reducing pollution. If current regulations force industry to spend \$200 per kg on reduction of pollutant X emissions, then avoiding one kg of X is "worth" at least \$200 to society. If regulations only require spending \$100 per kg of pollutant Z reduction, then avoiding Z is worth only half as much as avoiding X.

In some cases, particularly for the so-called "criteria pollutants", control cost information was available for individual pollutants. For other emissions, a hybrid approach was used, combining control costs and health or damage effects. In addition to the criteria pollutants, three other categories were examined: greenhouse gases, carcinogens, and (noncarcinogenic) toxics.

Within each category, emissions were weighted by estimates of relative health effects or damages. For greenhouse gases, there are well-established estimates of global warming potential, or "carbon equivalents." For carcinogens, there are published cancer potency factors. For noncarcinogenic toxic emissions, the "oral reference dose" is the maximum daily exposure that will not cause harm. The oral reference dose is based upon toxicology, as opposed to the

[6] Council of State Governments and Tellus Institute, *CSG/Tellus Packaging Study*, May 1992.

regulatory thresholds used by the Eco-factor method, which might be based on toxicity, but might also reflect political decisions. The inverse of the oral reference dose (1/reference dose) was used as a measure of health effects.

Emissions were weighted by these factors and summed, yielding category totals such as total carcinogenic emissions, weighted by cancer potency. We developed an "exchange rate" between carcinogenic and non-carcinogenic toxicity, which allowed us to combine the rankings for the carcinogenic and other toxic compounds. We then applied control cost estimates to the greenhouse gases, the criteria air pollutants, and the toxic chemicals. For example, control costs for lead emissions were used as a standard for all toxic emissions. This technique allows calculation of total impacts, across all emissions categories, in monetary terms. Impacts were calculated per ton of each packaging material, and can easily be converted to per-package effects.

C. Selection of methodology

The Tellus approach is not free from potential problems. It depends on regulatory data for control costs; this information may not be available for all countries, although US control costs can be used in the absence of local data. Also, it includes only some of the possible range of health and environmental impacts.

Yet despite these problems the Tellus system is applicable to a wide range of pollutants, contains a consistent valuation scheme, and yields easily understandable results. It is simpler to use than the Eco-factor method, which requires regulatory thresholds for all pollutants studied. Tellus' system requires regulatory data - control costs - only for criteria air pollutants, carbon dioxide, and any one toxic/carcinogenic compound. Although no available system of impact assessment is free from problems, the Tellus approach is superior to any others encountered in the literature. Therefore, we have applied it to the evaluation of the LCI data for packaging in Mexico.

II. Discussion of Lifecycle Inventory and Tellus' Impact Assessment Methodology

A. Definition of goals, boundaries and assumptions

The LCI measured the impacts caused by production of packages for use in Mexico. These impacts are divided into three types:

- Energy use
- Generation of solid waste
- Emissions to air and water

Energy use includes the energy consumed during manufacturing processes and transportation. It also includes the energy of material resources. In the manufacture of wooden crates, for example, the study measures not only the energy required to cut trees, transport the logs to a factory, and manufacture crates from the wood, but also the energy content of the wood crates.

Generation of solid waste includes waste products of manufacturing processes, waste products of fuel extraction and combustion, and postconsumer solid wastes.

Emissions to air and water include emissions caused by manufacturing processes, and emissions caused by fuel combustion both during production and transportation.

The primary boundary set on tracking impacts was the "one-step-back" rule. Impacts were counted for processes one step back from the main production chain, but not for processes two or more steps back. For example, when calculating the impacts from manufacturing steel cans, the impacts associated with iron mining were included, while impacts associated with producing mining equipment were not.

B. What is included in the study?

Discussion of a lifecycle inventory begins with two questions.

- What materials and end-products are included in the inventory?

- Which impacts are included in the inventory?

The first of these two questions is obvious, and simple to answer. The second is often more difficult. While energy use and solid waste generation are relatively easily tracked and measured, it is much more difficult to measure air and water emissions from transportation and facilities. Hundreds and thousands of chemicals may be present in raw materials, added as part of the manufacturing process, or created as the result of chemical reactions during manufacturing processes or transport. Some of these chemicals are present in large amounts, others only in parts per million or less; some are harmful, while others are completely benign.

All lifecycle inventories must limit the number of chemical emissions they include. Frequency of occurrence, hazard, and ease of detection and inventory are some of the factors affecting which emissions are included and which are not.

List of packages and materials

The LCI analyzed 21 packages used in Mexico. The packages selected span the commonly used packaging **materials**; these are not necessarily the 21 most common **packages**. All are food or beverage related, with the exception of the shampoo bottle and the cement sack. The packages, and the primary material used to manufacture them, are shown in Table 1.

List of emissions

Table 2 shows the emissions that were included in the LCI. They are divided into two types - air emissions and water emissions. Several chemicals, including ammonia, lead, mercury, and sulfides, appear on both lists. Notice that the lists include entries like "Hydrocarbons," "Metal Ion," and "Other Organics." In these cases, multiple compounds, which may have widely divergent impacts, were grouped under a single heading. Also notice that the list of air emissions is considerably longer than the list of water emissions. These issues will affect the impact assessment.

C. Discussion of impact assessment methodology

Three sets of assumptions underlie any impact assessment.

- Which hazards are going to be considered?
- How is toxicity of different materials ranked **within** each hazard category?
- How is toxicity compared **across** hazard categories?

The Tellus Institute *Packaging Study* methodology explicitly addresses all three of these questions.

1. Hazards under consideration

The Tellus methodology is concerned with four hazard types - the US EPA's criteria air pollutants[7], greenhouse gases, chemicals that are carcinogenic (cancer-causing) and chemicals that have other toxic effects. Each of the four hazard types is ranked

[7]Particulates, nitrogen oxides (NOx), sulfur oxides (SOx), volatile organic chemicals (VOCs), and carbon monoxide (CO).

Table 1 - List of Packages and Materials[a]

Package Name	Primary Material
Three-piece can for chilies	Steel
Beer can	Aluminum
Non-returnable soft drink bottle, 355 ml	Glass
Returnable soft drink bottle, 500 ml (25 trips)	Glass
Returnable soft drink bottle, 1.5 L (6 trips)	Polyethylene terephthalate (PETG)
Edible oil bottle	Polyethylene terephthalate (PET)
Shampoo bottle	High-density polyethylene (HDPE)
Water bottle	Polyvinyl chloride (PVC)
Bread bag	Low-density polyethylene (LDPE)
Sugar bag	Polypropylene (PP)
Pancake syrup container	Polypropylene
Yogurt container	Polystyrene (PS)
Crate for grapes	Expanded polystyrene (EPS)
Corn flour sack	Bleached semi-kraft paper
Cement sack	Unbleached semi-kraft paper
Folding carton cereal box	Clay coated paperboard
Box for egg trays	Corrugated paperboard
Laminated snack pack	Metalized polypropylene (MOPP)
Gable-top milk carton	LDPE-coated paperboard
Aseptic brick for milk	Aseptic package
Fruit crate	Wood

differently.

Most of the pollutants in the "air emissions" and "water emissions" categories are

[a]From LCI, Appendix Table B-1. Parentheses indicate abbreviations used in the text of this report.

Table 2 - List of pollutants included in LCI

Pollutant Name
Criteria Pollutants
CO
CO_2
NOx
Particulates
SOx
Air Emissions
Aldehydes
Ammonia
Antimony
Beryllium
Cadmium
Chlorine
Chromium
Copper
Cyclohexane
Disodium dioxide
Ethylene glycol
Fluorine
Hydrocarbons
Hydrochloric acid
Hydrogen fluoride
Isopropyl acetate
Kerosene
Lead
Manganese
Mercury
Metals
Methane
Nickel
Other organics
Potassium dioxide
Selenium
Styrene
Vanadium pentoxide
Vinyl chloride
Zinc

Pollutant Name
Water Emissions
Acid
Aluminum
Ammonia
Arsenic
BOD
Chlorine ion
Chromium
COD
Cyanide
Dissolved solids
Fluorides
Fluorine ion
Hydrocarbons
Iron
Lead
Mercury
Metal Ion
Nickel
Nitrogen
Oil
Other chemicals
Phenol
Phosphates
Sulfides
Sulfuric acid
Suspended solids
Zinc

either carcinogens, noncarcinogenic toxics, or both. We discuss below which pollutants listed in Table 2 do not pose direct human health hazards.

2. Rankings within categories

Criteria air pollutants have no hazard ranking at all. Their valuation system includes an assessment of their relative hazards, as will be discussed below.

Greenhouse gases, for the purpose of this study, consist only of carbon dioxide and methane. Carbon monoxide's value as a criteria air pollutant includes an assessment of its greenhouse potential, and neither the *Packaging Study* nor the LCI included nitrous oxide or CFCs. We assigned methane ten times the greenhouse gas potential of carbon dioxide.[9] The valuation system applied to the criteria air pollutants is also applied to the greenhouse gases, as will be discussed below.

Direct human health effects - carcinogenic and noncarcinogenic After a review of the diverse possible approaches to hazard ranking suggested by the literature, the *Packaging Study* 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. Pollutants were classified as carcinogens or toxic noncarcinogens based on the US Environmental Protection Agency's classification system.

Carcinogens were ranked based upon each pollutant's cancer potency factor, measured as milligrams pollutant/kilogram bodyweight/day. This factor is indicative of the cancer risk associated with a pollutant. Isophorone has the lowest potency factor of the carcinogenic pollutants associated with packaging production and disposal; for convenience in reporting, 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."

Noncarcinogens were ranked based upon each pollutant's oral reference dose. The reference dose (measured as milligrams pollutant/kilogram bodyweight/day) is an estimate of the maximum daily level of exposure that 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 indicate lower toxicity. As xylene has the smallest value based upon this scale, it was used as the basis of comparison - noncarcinogenic toxicity is measured in "xylene equivalents."

[9]Note that this was the value in use at the time; methane is currently assigned 22 times the greenhouse potential of carbon dioxide. Since methane is the source of a minuscule percentage of impacts from packaging production, the difference is unimportant for the purposes of the impact assessment.

3. Comparing impacts across categories

To compare impacts in different categories, the Packaging Study assigned monetary values (in $/lb) for each pollutant. The values for each criteria air pollutant and greenhouse gas is based on its control costs, as will be discussed below. The more difficult task was the valuation of carcinogenic and noncarcinogenic toxics. Because only one control cost estimate, for lead emissions, was available for the carcinogens and other toxics, it was necessary to combine their hazards into a single, comprehensive hazard ranking system. The approach which the *Packaging Study* used was to establish an "exchange rate" between the carcinogenic and noncarcinogenic rankings. It used Occupational Safety and Health Administration (OSHA) permissible exposure levels for isophorone and xylene to derive this exchange rate (3 units of xylene = 1 unit of isophorone), from which we derived the overall ranking for each pollutant. The most toxic compound included, 2,3,7,8-TCDD (dioxin), is 115 million times as hazardous per unit as xylene, the baseline; the next-highest ranking compounds are arsenic and thallium, more than 20,000 times as hazardous per unit as xylene. All other compounds were less than 10,000 times as hazardous, and the bulk were between 50 and 750 times as hazardous as xylene per unit.

4. Comparing list of LCI pollutants with hazard rankings

The *Packaging Study* list of pollutants is far larger than the LCI's list; the LCI contains many fewer organic compounds, and a slightly smaller list of heavy metals. There are some pollutants included in the LCI which were not included in the *Packaging Study* hazard rankings. Reasons for the difference include

- Some pollutants listed in the LCI encompass many pollutants in the *Packaging Study* hazard ranking. This includes LCI categories "Other Organics," "Metals," "Hydrocarbons," etc.

- Some pollutants listed in the LCI were included in the *Packaging Study*, but were not ranked because they did not pose direct threats to human health. These include biological and chemical oxygen demand (BOD and COD), suspended solids, aluminum, iron, and phosphates.

- A few pollutants listed in the LCI were not mentioned in the *Packaging Study* at all. These are isopropyl acetate, vanadium pentoxide, sodium oxide, cobalt, potassium oxide, and kerosene.

In all three cases, we did not assign the pollutants hazard rankings. In the first case, categories such as "other organics" are too diverse to assign values to them. In the second case, there are no human health effects to value, although emissions may be significant for other reasons. In the third case, these pollutants might pose a threat to human health which we have not quantified correctly. Examining the emissions levels of these materials in the LCI, however, we concluded that emissions of these compounds were too small to affect the valuation system.

We are confident that the minor errors introduced by omitting them will have only a minimal effect on the final results.

D. The Tellus valuation system

1. Introduction[10]

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 the following chapters of this report, we apply the method developed in the *Packaging Study* to these environmental costs. The actual valuations have been updated. The update is discussed in greater detail in Appendix A of this report.

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. The *Packaging Study* 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.

2. Control costs for criteria air pollutants and greenhouse gases

For the criteria air pollutants and greenhouse gases, we are using control costs developed for the state of California. California, like Mexico, has one huge metropolitan area in which air pollution has reached crisis proportions, several other large cities, and vast amounts of low-density rural areas. California's response to pollution may prove appropriate for Mexico as well. Control costs for all CAPs and greenhouse gases are taken from a 1991 California Public Utilities Commission decision in 1991,[11] with one exception. Carbon monoxide emissions were not valued as part of this decision; we adopted the value set by the California South Coast Air Quality Management District in 1989.[12] All values have been

[10]The discussion in this section is drawn heavily from the *Packaging Study Executive Summary*, page 38.

[11]CA PUC Decision 91-06-022, June 5, 1991.

[12]South Coast Air Quality Management District, *Air Quality Management Plan, 1989 Revision*, March 1989.

updated to 1993 US dollars, and then converted to Mexican pesos using the conversion factor of 3.407 pesos/dollar.[13]

3. Control costs for lead

Two US energy researchers, Chernick and Caverhill, computed the costs of control for several heavy metals in 1991. They estimated that under the Clean Air Act Amendments, the marginal cost of control of lead emissions from secondary lead smelters was $500/lb lead.[14] This corresponds to $528/lb lead in 1993 US dollars. This control cost was applied to the hazard ranking system developed above. The cost per pound of each toxic is equal to

Cost per pound pollutant = $528 * (Ranking of pollutant/ Ranking of lead)

In the Tellus ranking system, lead has a hazard ranking of 1,429 per unit, while cadmium has a ranking of 4,346. The cost per pound of cadmium emitted, therefore, is

$528 * (4346/1429) = $1606.

The same calculation is made for each of the other carcinogenic and toxic noncarcinogenic pollutants in the Tellus hazard ranking. Table 3 shows pollutant prices for the pollutants included in the LCI and in the Tellus hazard ranking.

[13]Exchange rate on October 6, 1994.

[14]Paul Chernick and Emily Caverhill, *Joint Testimony on Behalf of Boston Gas Company, MA DPU Docket D PU 91-131*, October 4, 1991, page 61.

Table 3 - List of pollutants included in LCI and their impacts (N$/kg pollutant emitted)

Pollutant Name	Pollutant Impacts
Criteria Pollutants	
CO	3.16
CO2	0.03
NOx	34.03
Particulates	9.77
SOx	16.75
Air Emissions	
Aldehydes	15.85
Ammonia	6
Antimony	13,883
Beryllium	5,147
Cadmium	12,068
Chlorine	44
Chromium	6
Copper	150
Cyclohexane	N/A
Disodium dioxide	N/A
Ethylene glycol	N/A
Fluorine	139
Hydrocarbons	N/A
Hydrochloric acid	0
Hydrogen fluoride	93
Isopropyl acetate	N/A
Kerosene	N/A
Lead	3,967
Manganese	28
Mercury	18,511
Metal Ion	N/A
Methane	0.30
Nickel	1,036
Other organics	N/A
Potassium dioxide	N/A
Selenium	1,851
Styrene	46
Vanadium pentoxide	N/A
Vinyl chloride	4,912
Zinc	28

Pollutant Name	Pollutant Impacts
Water Emissions	
Acid	N/A
Aluminum	0
Ammonia	6
Arsenic	56,173
BOD	N/A
Chlorine ion	44
Chromium	6
COD	N/A
Cyanide	278
Dissolved solids	N/A
Fluorides	93
Fluorine ion	93
Hydrocarbons	N/A
Iron	N/A
Lead	3,967
Mercury	18,511
Metal Ion	N/A
Nickel	1,036
Nitrogen	N/A
Oil	N/A
Other chemicals	N/A
Phenol	9
Phosphates	N/A
Sulfides	86
Sulfuric acid	0
Suspended solids	N/A
Zinc	28

Report uses 10/6/1994 exchange rate - 3.407 pesos/dollar.

III. Application of Hazard Ranking and Valuation to LCI

Introduction

There are two stages in valuing the emissions from packaging. The first step is to value the emissions from the production of packaging **materials**, like steel and aluminum. In most cases, this step alone is inadequate for application to packaging policy. It is more important to evaluate the **per-package** emissions. The relevant equation is:

Per-package impact (N$/package) = N$/tonne of material * tonnes of material/package

For example, although the per-tonne emissions from aluminum production will be shown to be many times higher than the emissions from glass production, there is a much smaller difference when comparing aluminum beer cans to glass soda bottles. The following table shows the reason.

Material	Impact/tonne of material	Weight per package
Aluminum	High	Low
Glass	Low	High

Aluminum has a high impact per tonne, because production is energy-intensive and pollution-intensive when compared with glass production. But it has a low weight per package, since aluminum cans weigh 1,566 kg per 100,000 cans while glass nonreturnable soda bottles weigh 17,830 kg per 100,000 bottles; glass returnable soda bottles weigh 51,230 kg per 100,000 bottles.[15] As a result, the production impacts from aluminum beer cans are somewhat higher than the impacts from returnable glass soda bottles, but less than half the impact of nonreturnable soda bottles.

A. Impacts per tonne of packaging materials

1. Calculating pollutant emissions per tonne of materials

The LCI did not calculate impacts per tonne of packaging materials; it concentrated on impacts per final package. Tellus used the information in the appendices of the LCI to estimate per-material impacts.[16] Tables in Appendices C-P, where impacts per tonne of

[15] Franklin Associates, *Life Cycle Inventory of Packaging Materials in Mexico* (LCI), page Appendix B-2.

[16] The LCI appendix tables present information in two formats - emissions per tonne and emissions per 100,000 packages. We used the tables in emissions per tonne to calculate impacts per tonne of materials.

materials are found, present three forms of information that we used in our calculations.

- Emissions during the manufacturing process (called "environmental emissions" in the LCI) per tonne of material.

- Amount of fuel consumed per tonne of material

- Raw materials required per tonne of material

In order to calculate total impacts, we needed to use the three forms in different ways. The environmental emissions of each pollutant could simply be included in the total impacts with no additional calculations. For fuel-related emissions, we multiplied pollutants per unit of fuel by fuel use per tonne of material. Adding the results for all fuels used gave total fuel-related emissions. For each pollutant, summing emissions from fuel consumption and environmental emissions gave the total emissions of each pollutant per tonne of material.

Some packaging materials are "built" from simpler materials, which also were included in the LCI because of the one-step-back rule. For example, PVC is made from vinyl chloride monomer and styrene-butadiene rubber. In these cases, we calculated emissions per tonne of "building block" as explained above, and multiplied by tonnes of "building block" per tonne of packaging material. We then added "building block" emissions to the fuel and environmental emissions.

2. Application of valuation method

Once we calculated the emissions of each pollutant per tonne of packaging material, we multiplied the impact per unit of pollutant shown in Table 3 by the calculated emissions, yielding each pollutant's impacts per tonne of packaging material. Summing the results for all pollutants yields the total impacts per tonne of packaging material.

Table 4 shows that there is a wide range of emissions impacts per tonne of packaging material. US production of LDPE has the lowest impacts, N$ 294/tonne, followed by polypropylene at N$ 363. At the other extreme, polyvinyl chloride resin has by far the greatest impacts, N$ 10,382 per tonne. Most other materials fall between N$ 600 and N$ 1,200 per tonne. Several of our results merit further discussion. Three materials - bleached kraft paper, unbleached kraft paper, and LDPE - are made both in Mexico and in the United States. In all cases the impacts of the materials made in the United States were lower. Another surprising result was the high impacts of Swedish foodboard, which were the highest of any of the paper types. These results will be explained in Chapter 4. Note that polystyrene is not included in Table 4 because there is no intermediate "polystyrene resin" between the production of raw materials and the production of polystyrene packaging.

Table 4 - Impacts per tonne of material

Material [1]	Mexico LCI Pesos/tonne
LDPE (US)	$294
PP (US)	$363
Wood (Mexico)	$447
Linerboard (US)	$599
Corrugating Medium	$600
Virgin Glass (Mexico)	$687
Unbleached Kraft (US)	$702
Virgin Steel (US)	$795
HDPE (Mexico)	$835
PET (US)	$870
Unbleached Kraft (Mexico)	$923
Bleached Kraft (US)	$960
LDPE (Mexico)	$1,107
LDPE-coated paperboard (US)	$1,124
PETG (US)	$1,206
Bleached Kraft (Mexico)	$1,232
MOPP film (Mexico)	$1,509
Foodboard (Sweden)	$2,073
Aluminum sheet (US)	$3,331
PVC (Mexico)	$10,382

Notes:
[1] List of materials is taken from LCI. Parentheses indicate where the material is produced

B. Impacts per package

1. Calculating impacts per package

Franklin Associates calculated per-package emissions directly, using the information shown in the appendices of the LCI. The tables in Chapters 2-14 of the LCI show emissions per 100.000 packages for each of the 21 package types. Tellus applied its valuation method to each of the packages, translating emissions per package into impacts per package. Table 5 shows the resulting valuations.

The second column of Table 5 shows impacts per package. The packages are ordered from lowest-impact per package to highest. The laminated snack package has the lowest impacts per package, less than one centavo (N$ 0.01) per package, while the water bottle and the crate for grapes have impacts of greater than one peso per container. Fifteen of the 21 packages have impacts of less than N$ 0.10 (about 3 US cents) per package.

What does that mean for packaging, and packaging policy? One definite finding is that, to a first approximation, heavy packages have higher impacts than light ones. The third column of the table shows package weights in grams. The six packages with impacts above N$ 0.10 are six of the seven heaviest packages; the returnable glass soda bottle, which is assumed to make twenty-five trips before final disposal, is a low-impact package, despite the fact that it is the third-heaviest container. The returnable PETG soda bottle is the only other package which both weighs more than 100 grams and has per-package impacts less than N$ 0.10. On the other hand, the aluminum beer can and aseptic brick have high impacts relative to their package weights; they have impacts of N$ 0.05-0.07, despite the fact that the aluminum can is the fourth-lightest and the aseptic brick is the sixth-lightest.

But this comparison discriminates in favor of small packages while favoring light ones. Another method of comparing packages is to ask what the impacts would be per liter or per kilogram of contents. Column five shows the weight of contents and column seven the volume of contents for each of the packages.[17] The accompanying columns six and eight show impacts per kilogram of contents (for products sold by weight) and impacts per liter of contents (for products sold by volume). Of the seven packages sold by weight, five have impacts of less than N$ 0.10 per kilogram of contents; the exceptions are the MOPP snack pouch and the chili can. Although the snack pouch has the lowest impacts per package, each package contains only 25 grams of food; it requires 40 pouches to sell one kilogram of snacks. The cement sack, which has high impacts per package (N$ 0.34), contains fifty kg of cement per sack; the impacts per kg, therefore, are less than N$ 0.01 per kilogram. The chili can is the highest-impact package per kilogram of contents, while the PP sugar bag is the lowest.

[17] Note that LCI Appendix B did not give contents weight or volume for the box for egg trays, so there is no impact per kilogram or liter of contents for this package.

Table 5 - Impacts per package

Package Type [1]	Per-package Valuation	Package Weight (g)	Valuation/ kg package	Contents Weight (kg)	Valuation/ kg contents	Contents Volume (L)	Valuation/ L contents
Laminated snack pack	$0.00	1.73	$1.32	0.025	$0.13		
Bread bag	$0.01	8.74	$1.27	0.65	$0.02		
Yogurt container	$0.02	12.12	$1.63			0.24	$0.08
Corn flour sack	$0.02	15.70	$1.57	1	$0.02		
Pancake syrup container	$0.03	40.10	$0.63			0.354	$0.07
Folding carton cereal box	$0.03	80.00	$0.37	0.35	$0.08		
Returnable 500 ml glass soda bottle	$0.03	512.30	$0.06			0.5	$0.06
Gable-top milk carton	$0.04	31.85	$1.14			1	$0.04
Three-piece can for chilies	$0.04	44.14	$0.93	0.198	$0.21		
Returnable 1.5 L plastic soda bottle	$0.05	108.73	$0.43			1.5	$0.03
Beer can	$0.05	15.66	$3.28			0.34	$0.15
Edible oil bottle	$0.07	41.04	$1.66			1	$0.07
Aseptic brick for milk	$0.07	28.50	$2.58			1	$0.07
Sugar bag	$0.08	95.00	$0.88	50	$0.00		
Shampoo bottle	$0.09	59.00	$1.50			0.4	$0.22
Nonreturnable 355 ml soft drink bottle	$0.11	167.30	$0.68			0.355	$0.32
Fruit crate	$0.21	885.00	$0.23			35.2	$0.01
Cement sack	$0.34	285.00	$1.18	50	$0.01		
Box for egg trays	$0.55	1222.00	$0.45				
Water bottle	$1.05	113.50	$9.22			3.78	$0.28
Crate for grapes	$1.13	327.00	$3.45			21.23	$0.05

Notes:

[1] Package weights, contents weights, and contents volumes are taken from LCI, Appendix B.

Ten of the 14 packages sold by volume have impacts less than N$ 0.10 per liter of contents. The nonreturnable glass soda bottle is highest-impact per liter of contents, while the fruit crate is the lowest. The crate for grapes, the highest-impact container when viewed on a per-package basis, is among the lowest on a per-liter basis, as one crate holds 21.23 liters of grapes.

To summarize the data shown in Table 5:

Method of comparison	Lowest-impact[18]	Highest-impact
Impacts per package	Snack pouch (1)	Grape crate (21)
Impacts per kg of contents	Sugar bag (14)	Chili can (9)
Impacts per liter of contents	Fruit crate (17)	Nonreturnable glass soda bottle (15)

2. Case study of beer and soda containers

Beverage containers, especially beer and soda containers, are some of the most-regulated container types; the companion policy report discusses different countries' approaches to managing these easily-littered packages. There are four beer and soda containers in our 21 package study - the returnable PETG soda bottle, the returnable glass soda bottle, the nonreturnable glass soda bottle, and the nonreturnable aluminum beer can. The PETG bottle is the only multiple-serving container. On a per-package basis, the returnable glass has the lowest impacts, while the returnable PETG and the aluminum can have the same, slightly higher, impacts, and the nonreturnable glass bottle has the highest - nearly four time the returnable glass. But when viewed on a per-liter basis, the returnable PETG soda bottle has half the impacts per liter of soda of the returnable glass. The aluminum beer can and glass nonreturnable have many times higher impacts per liter.

So which is "better?" The returnable glass bottle has the lowest impacts on both a per-package and a per-liter basis of the single-serving containers. But based on the evidence shown in Table 5, it is impossible to state that it has "lower" impacts than the returnable PETG soda bottle, or vice versa.

[18]Parentheses indicate packages' ranking based on impacts per package.

IV. Analysis of Results and Conclusions

A. Summary of previous chapters

In the first three chapters of this report, we have calculated impacts per package for the twenty-one package types studied by the Franklin Associates *Life Cycle Inventory of Packaging Materials in Mexico* (the LCI). Our impact assessment used the methodology developed in the Tellus Institute *Packaging Study*, which used toxicology data to rank the harmfulness of pollutants and pollution abatement control costs to value the harm per unit of pollutant.

In Chapter Three, we calculated impacts for both packaging **materials** (aluminum sheet, plastic resin, etc.) and the packages studied by the LCI. Table 4 shows impacts per tonne of packaging. Low-density polyethylene resin, polypropylene resin (produced in the United States), and wood (produced in Mexico), have the lowest impacts per tonne of material, while PVC resin (produced in Mexico) and aluminum sheet (produced in the United States), has the highest.

Table 5 shows impacts per package. The metalized polypropylene film snack pack has the lowest impacts per package, while the expanded polystyrene grape crate has the highest. Another basis for comparison is impacts per kilogram or per liter of contents; using this standard removes the bias of package size on the impact calculations. Of packages measured by weight, the polypropylene sugar bag has the lowest impacts and the steel chili can the highest. Of packages measured by volume, the wooden fruit crate has the lowest impacts and the nonreturnable glass soda bottle the highest.

In this chapter, we will analyze the results and discuss their implications.

B. Impacts per tonne of material

Many of the estimates of impacts per tonne of material, shown in Table 4, are easy to explain. For example, per-ton impacts for wood and corrugating medium are comparatively low. Wood packaging is completely produced in Mexico, and the manufacturing process uses limited quantities of energy and few toxic chemicals. Corrugating medium is produced completely from waste paper products, a process that is much less energy-intensive than virgin production, and also does not use as many toxics.

It is also easy to understand why aluminum and PVC have such high impacts per tonne. Virgin aluminum production requires vast amounts of energy per tonne; emissions from fuel combustion account for the bulk of total impacts. PVC is manufactured by polymerization of vinyl chloride. Vinyl chloride is a known carcinogen, and large quantities of it are emitted during polymerization.

One of the more surprising results is the high estimate for Swedish foodboard, used in aseptic bricks. Swedish foodboard has the third-highest impacts on a per-tonne basis Table

Appendix O-1 of the LCI shows environmental emissions of 19.9 kilograms of hydrogen sulfide per tonne of foodboard; these emissions account for the bulk of the costs associated with foodboard production. Sulfide emissions in Swedish paperboard manufacture are considerably higher than sulfide emissions in either Mexico or the United States. The production of virgin bleached kraft paper in the United States, for example, emits only 0.85 kilograms of sulfides per tonne of paper produced.

C. Comparison with US production impacts

How do the valuations of the LCI data compare with similar results for US packaging production? Table 6 compares the LCI-based valuations (from Table 4) with the updated values from Tellus Institute's *Packaging Study*. The *Packaging Study* column uses the lifecycle inventory and hazard ranking originally developed for the study, and the revised valuation method. Our rationale from switching from the valuations presented in the *Packaging Study* to the current valuation procedure is discussed in Appendix A. The table presents two side-by-side comparisons.

The first two columns report the overall valuations from each study. Column 2 shows the impacts (in N$) per tonne of packaging materials calculated from the LCI. Column 3 shows the impacts (in N$) per tonne from the *Packaging Study*. In almost all cases the LCI implied higher impacts per tonne of material than did the *Packaging Study*. For polypropylene made in the United States, LDPE made in the United States, and PET made in the United States, however, the LCI inventory implied lower impacts per tonne than the lifecycle inventory in the *Packaging Study*.

Some of the differences between these two columns simply reflect the fact that the two studies included different sets of pollutants. Therefore, the last two columns of Table 6 report the valuation when restricted to the set of pollutants that are common to both studies. The Mexico column subtracts the impacts of carbon dioxide, which was not included in the *Packaging Study*, while the US column does not include the impacts of many organic compounds, such as benzene, naphthalene, and carbon tetrachloride, which were not included in the LCI. In this comparison, using the common set of pollutants, the LCI-based valuation is 1.4 to 2.8 times the *Packaging Study* value, for all but the three plastics -- LDPE, polypropylene, and PET -- for which the LCI values are lower than or roughly equal to the *Packaging Study* results.[19]

Two materials, glass and PVC, require further discussion. The LCI-based valuation of glass is surprisingly high. In the US, the *Packaging Study* found that glass manufacturing's impacts were the lowest of any packaging material per ton. However, Table 4 shows that

[19] The results for these three plastics probably reflect differences between the Franklin Associates and Tellus Institute models of the U.S. plastic industry. Use of the Tellus values would make LDPE, and to a lesser extent PET, look less attractive, but would cause little change in the qualitative picture of packaging impacts.

Tabie 6 - Comparison of LCI impacts per tonne of material with Packaging Study impacts per tonne of material

Material [1]	Overall results Mexico LCI Pesos/tonne	Overall results Tellus Institute Packaging Study (US Pesos/tonne [2]	Common set of pollutants Mexico LCI Pesos/tonne	Common set of pollutants Tellus Institute Packaging Study (US) Pesos/tonne
LDPE (US)	$294	$595	$250	$444
PP (US)	$363	$559	$311	$305
Wood (Mexico)	$447	Not studied	$422	Not studied
Linerboard (US)	$599	$358	$506	$358
Corrugating Medium	$600	$185	$522	$185
Virgin Glass (Mexico)	$687	$263	$654	$263
Unbleached Kraft (US)	$702	$363	$602	$363
Virgin Steel (US)	$795	$303	$702	$261
HDPE (Mexico)	$835	$484	$762	$341
PET (US)	$870	$1,245	$782	$999
Unbleached Kraft (Mexico)	$923	$363	$833	$363
Bleached Kraft (US)	$960	$456	$829	$454
LDPE (Mexico)	$1,107	$595	$1,017	$444
LDPE-coated paperboard (US)	$1,124	Not studied	$982	Not studied
PETG (US)	$1,206	Not studied	$1,085	Not studied
Bleached Kraft (Mexico)	$1,232	$456	$1,101	$454
MOPP film (Mexico)	$1,509	Not studied	$1,393	Not studied
Foodboard (Sweden)	$2,073	Not studied	$2,016	Not studied
Aluminum sheet (US) [3]	$3,331	$1,731	$3,064	$1,731
PVC (Mexico)	$10,382	$6,453	$10,226	$4,407

Notes:

[1] List of materials is taken from LCI. Parentheses indicate where the material is produced

[2] From Tellus Institute, US Packaging Study. The Packaging Study did not include transportation.

[3] Packaging Study valuation is based on assumption of 55% post-consumer content

in Mexico, glass has higher impacts per tonne than several other more toxics-intensive materials.

Figure 1 analyzes the Mexican vs. US valuations of glass productions impacts. Particulate emissions cause the divergence between the two studies; the LCI shows the particulate emissions to be more than 100 times the Packaging Study level, while for all other emissions that are included in both studies, the valuations are nearly identical. If the *Packaging Study*'s levels of particulate emissions were substituted for the LCI's, glass would have the second-lowest impacts per tonne of material, just above polypropylene.

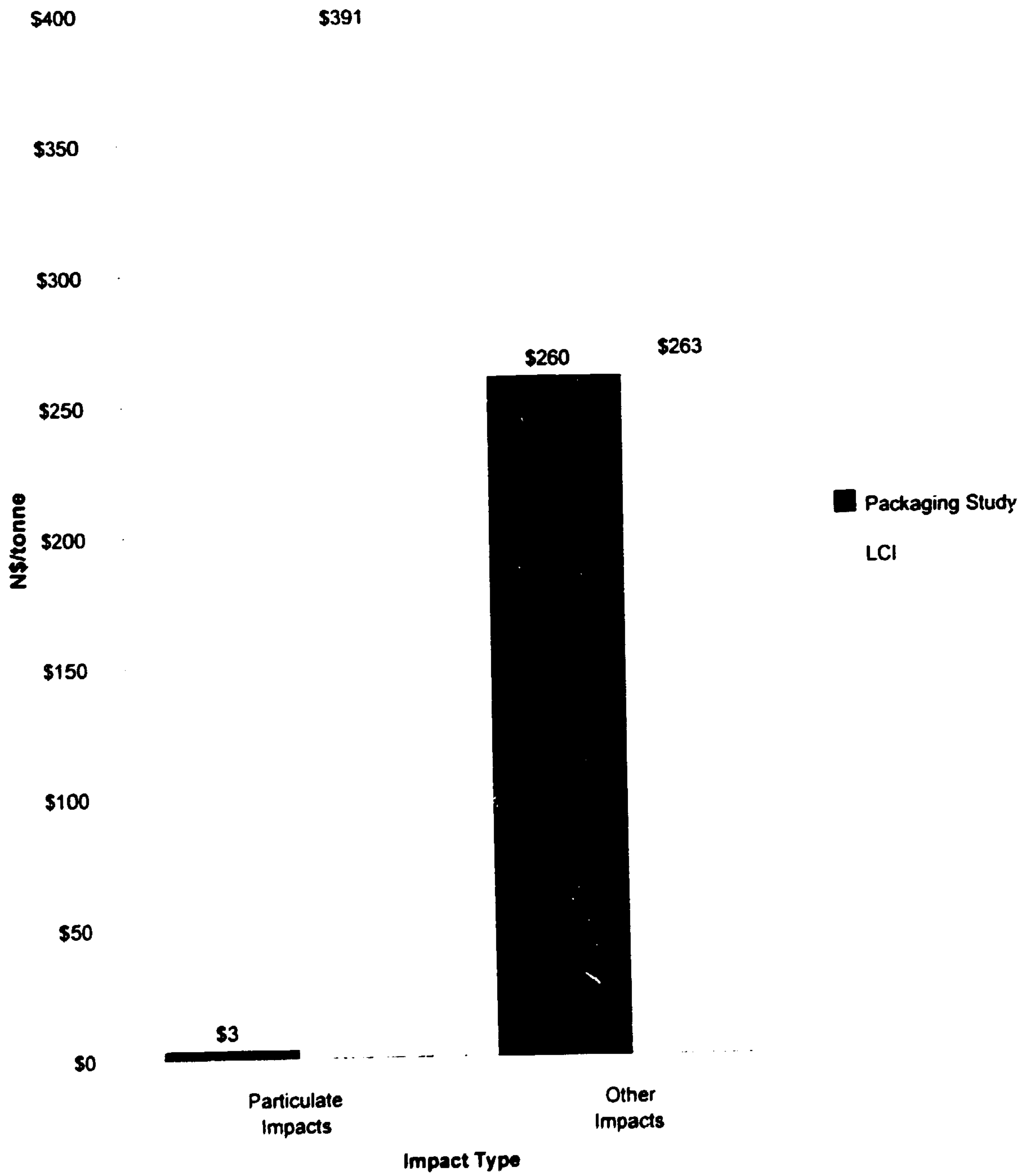
Appendix P of the LCI shows that almost all of the reported particulate emissions from glassmaking in Mexico occur during the final container fabrication stage, where raw materials are melted and blown into glass bottles. This difference between Mexican and US particulate emissions is so large that it may reflect a data error in the LCI. If there were a data error, and Mexican glass industry particulate emissions are actually equal to US emissi .is, then the valuation of the nonreturnable glass bottle would drop by about one-half, making it roughly equal to the aluminum beer can. (The valuation of the returnable glass bottle would be much less affected, since the emissions from cleaning, refilling, and transporting bottles account for most of the impact of returnables.)

On the other hand, if the data are correct, and glass industry particulate emissions are more than 100 times as great in Mexico as in the US, then this is an obvious area for improvement. There are many well-known technologies for capturing particulates. Controlling glass industry particulate emissions could prove to be one of the most cost-effective opportunities for pollution reduction.

PVC has the highest per-tonne impacts in both the LCI and the Packaging Study; impacts in Mexico are almost twice as high as in the US overall, and more than twice as high on the common set of pollutants. The high valuation is largely, though not entirely, due to the emissions of vinyl chloride (VC) monomer. Figure 2 shows the valuation of PVC emissions from three studies - the Packaging Study, the LCI, and a lifecycle assessment commissioned by the Vinyl Institute, the trade association of US manufacturers of PVC products. The LCI emissions of VC are more than twice the Tellus estimate of US emissions, and more than seven times the Vinyl Institute's estimate. The valuation of all other PVC production emissions in the LCI is close to the Tellus figure, although almost three times the Vinyl Institute value.

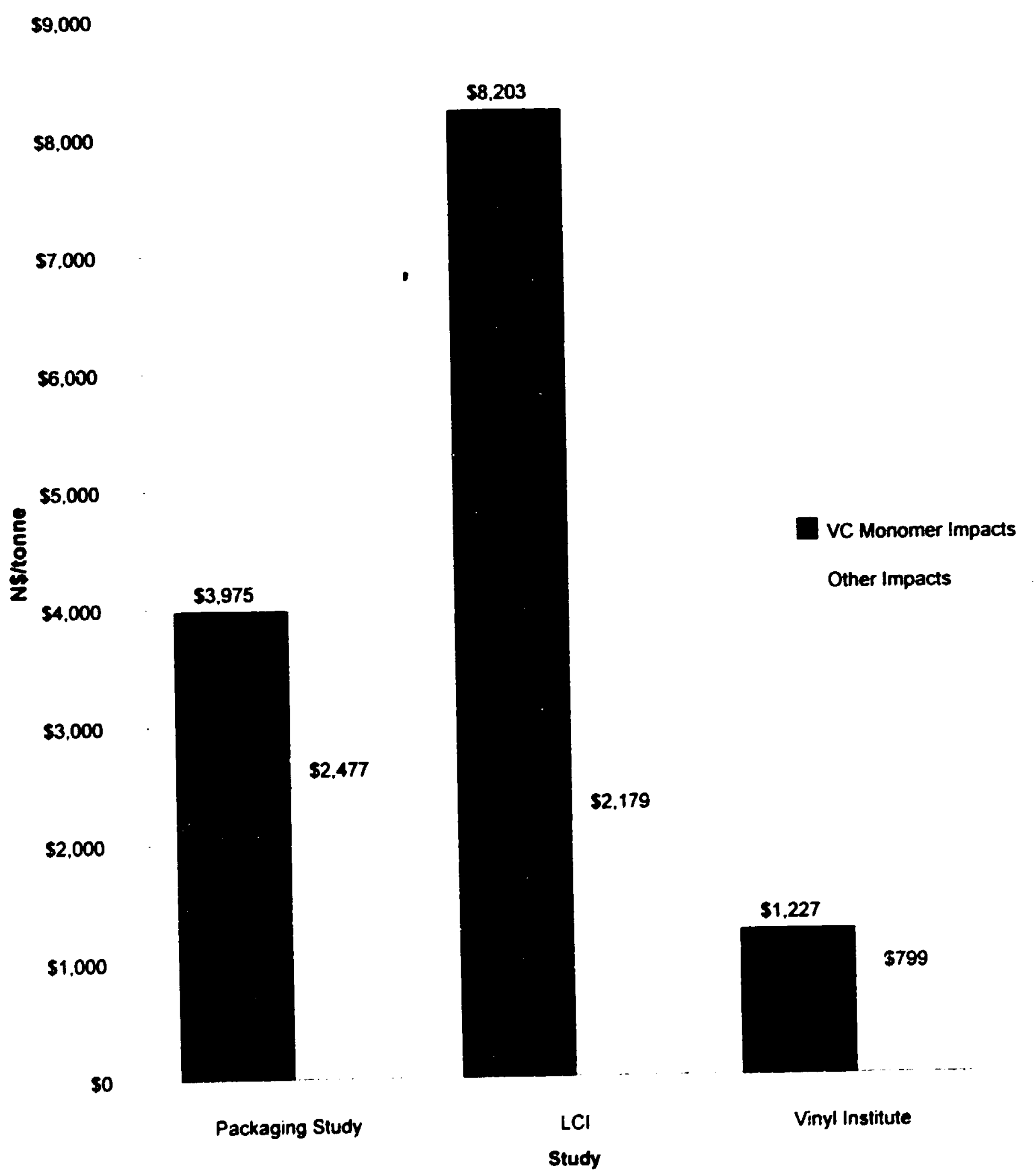
In the case of glass, reducing particulate emissions to US levels would largely eliminate the environmental problem that appears in the LCI. However, in the case of PVC, reduction of vinyl chloride emissions to US levels would still leave a substantial environmental burden. On a per-tonne basis, PVC production is responsible for the largest emissions, by far, of carcinogens in the Mexican packaging industry -- and this would still be true if it reached US emission levels.

Figure 1 - Comparison of impacts per tonne of virgin glass produced



Comparison includes only pollutants common to the two studies.

Figure 2 - Comparison of impacts per tonne of PVC produced



D. Impacts per package

It is difficult to compare the packages studied by the LCI with those included in the *Packaging Study*, as there is limited overlap between the two lists of packages. Soda and beer containers provide the best area of comparison. The *Packaging Study* computed costs per package for several soda containers, including the equivalents of the 355 ml glass nonreturnable and the 340 ml aluminum beer can. Using the new valuations and the amounts of material per package from the *Packaging Study*, we compute that the US nonreturnable aluminum beer can has impacts of N$ 0.03 per package, the glass nonreturnable has impacts of N$ 0.06, and a **nonreturnable** 1.5 liter PET bottle has impacts of N$ 0.06.[20] The impacts per package for the aluminum and glass bottles for the US market are, as expected, approximately half the impacts reported in the LCI.

It is more instructive to compare packages within the LCI, especially those made from Mexican vs. imported materials. Some packages produced from Mexican raw materials are among the lowest-impact on a per-weight or per-volume basis, including the LDPE bread bag, the flour sack, the cement sack, and the wood fruit crate. Other packages made from Mexican raw materials have much higher impacts, notably the nonreturnable glass bottle and the PVC water bottle. Packages produced from US raw materials are more likely to fall in the middle of the range. The aseptic package also falls in the middle of the range; its light weight offsets the high valuation of emissions from Swedish foodboard production.

Knowing the per-package impacts is not always sufficient to develop policy options. Knowing where and how the emissions are generated is also important. If emissions from manufacturing facilities are the bulk of the total impacts (as is the case of PVC production) then site-specific policies may be appropriate. But if the bulk of emissions are associated with Mexican electricity production, transportation, or processes taking place outside of Mexico, then policies directed at Mexican packaging industries may be very indirect methods of influencing pollution.

Table 7 shows a breakdown of the emissions per package into four categories - impacts from electricity used by Mexican manufacturers, impacts from transportation in Mexico, on-site impacts in Mexico, and impacts outside Mexico.[21] The "on-site" column represents on-site impacts, including emissions from manufacturing processes and emissions from on-site fuel combustion.

For five of the packages - the syrup container, the gable-top milk container, the chili can, the beer can, and the aseptic brick - more than three-quarters of the impacts occur outside of Mexico.

---

[20]The *Packaging Study* did not include a 1.5 liter bottle size; this value is the midpoint of the impacts of the 1 liter and 2 liter bottles.

[21]Calculated from the LCI and tables supplied by Kent Hart, Franklin Associates.

Table 7 - Detailed breakdown of impacts per package

11/16/94 11:55 AM

Package Type [1]	Per-package Valuation	% Mexican Electrical	% Mexican Transportation	% Mexican On-site [2]	% Outside Mexico
Laminated snack pack	$0.00	44%	11%	21%	24%
Bread bag	$0.01	52%	5%	43%	0%
Yogurt container	$0.02	53%	5%	30%	11%
Corn flour sack	$0.02	35%	3%	14%	49%
Pancake syrup container	$0.03	0%	8%	0%	92%
Folding carton cereal box	$0.03	4%	1%	57%	38%
Returnable 500 ml glass soda bottle	$0.03	25%	4%	54%	17%
Gable-top milk carton	$0.04	6%	7%	0%	86%
Three-piece can for chilies	$0.04	4%	14%	5%	78%
Returnable 1.5 L plastic soda bottle	$0.05	39%	3%	11%	47%
Beer can	$0.05	11%	5%	6%	79%
Edible oil bottle	$0.07	47%	5%	2%	46%
Aseptic brick for milk	$0.07	4%	2%	0%	93%
Sugar bag	$0.08	49%	9%	0%	42%
Shampoo bottle	$0.09	77%	2%	15%	6%
Nonreturnable 355 ml soft drink bottle	$0.11	26%	6%	64%	4%
Fruit crate	$0.21	65%	20%	15%	0%
Cement sack	$0.34	54%	2%	23%	22%
Box for egg trays	$0.55	59%	3%	35%	2%
Water bottle	$1.05	8%	1%	86%	5%
Crate for grapes	$1.13	37%	6%	56%	0%

Notes:

[1] Package weights, contents weights, and contents volumes are taken from LCI, Appendix B.

[2] "On-site" includes process emissions and all emissions from non-electrical process fuel combustion

For nine of the packages - the laminated snack pack. the bread bag. the yogurt container. the edible oil bottle. the sugar bag. the shampoo bottle. the fruit crate. the cement sack. and the box for egg trays - the impacts from electrical generation are both the most important and are more than 40% of total impacts.

For five of the packages - the folding carton cereal box. the returnable glass soda bottle. the nonreturnable glass soda bottle. the water bottle and the crate for grapes - on-site emissions are most important.[22] Moreover. for the crate for grapes and the cereal box. it is on-site fuel emissions rather than process emissions that account for the bulk of the on-site impacts. Only for the nonreturnable glass and PVC bottles are process emissions - particulate emissions from glass manufacturing and vinyl chloride emissions from PVC manufacture - the most important contributors to total impacts.

For the flour sack and returnable plastic soda bottle. impacts outside Mexico are nearly 50% of the total. with Mexican electrical generation contributing nearly 40% of the total.

Impacts arising from the generation and use of electricity, therefore. contribute much more to the total impacts in Mexico than do impacts from on-site manufacturing emissions.

E. Conclusions

Our results have important policy implications. They suggest that Mexican packaging producers are directly responsible for only some of the impacts associated with packaging production. Measures to promote better emissions controls at electrical generating facilities and increased fuel efficiency in packaging production may have the greatest potential for impact reductions.

1. Possible effects of packaging taxes

At the outset of this project. some participants thought that a per-package tax based on the Tellus evaluation of production impacts might be an effective method for reducing total impacts from packaging production. It is now clear that such a tax would fall particularly heavily on two industries. glass (if the information is correct) and PVC; beyond these two cases. the tax might have the perverse effect of penalizing Mexican industry. in large part because it uses Mexican electricity. To avoid this perverse effect. we believe it is preferable to address specific emissions problems more directly. We therefore do not recommend the use of our valuations as the basis for a packaging tax.

[22]If particulate emissions from the glass mill were comparable to US glassmaking emissions. it is likely that emissions from electrical generation would also be the most important source of glassmaking impacts.

2. Priorities for emissions reductions.

In the absence of a packaging tax, other policy measures can be used to reduce emissions. There are four areas for improvement, of which the first three are of greatest importance for most packaging industries.

Reduce emissions from electrical generation. Valuing the emissions factors in Appendix A-12 of the LCI shows that emissions from the Mexican power grid are much higher than US levels in SOx and heavy metals, while lower in particulates and NOx. Total impacts per kilowatt-hour are 50% higher in Mexico than in the US. Reducing emissions per kWh to US levels would substantially lower the impacts per package of the nine packages whose primary source of impacts is the Mexican electrical grid.

Increase energy efficiency at manufacturing facilities. Another method of decreasing impacts from fuel combustion is to increase fuel efficiency at manufacturing facilities. This solution would impose direct costs on the packaging producers as they changed their processes and equipment, but it would eventually lead to direct benefits from lower energy use as well as the social benefit of lower impacts from packaging manufacturing.

Improve emissions controls for on-site fuel combustion and transportation. This method would impose direct costs on producers, but would yield only the social benefits. This method would presumably be employed in conjunction with the second.

'mprove process controls for selected industries. As discussed above, process emissions are the source of most impacts from glass (if the particulate information is correct) and PVC manufacture. For these two industries, additional process emissions controls would be necessary to significantly decrease manufacturing impacts.

Appendix A - Updating the Tellus Valuation Method

A. Adjustments to *Packaging Study* valuation methods

The literature regarding control costs for criteria air pollutants and heavy metals was sparse when the *Packaging Study* was first conceived and researched. Updated information has led us to change our strategy for valuing both the criteria air pollutants and toxics.

1. Adjustments to valuation of criteria air pollutants

The values employed by the *Packaging Study* were based on a 1990 decision by the California Energy Commission. Since then, the CEC has updated its values for certain areas within California[23] and the California Public Utilities Commission has published its own values, both for the South Coast Air Quality Management District and for the state as a whole.[24] The new CEC information is based upon estimates of the damages caused per unit of pollutant, while the PUC values represent control costs. We have chosen to use the PUC values rather than the CEC ones because they are control costs, rather than damage costs.

2. Adjustments to valuation of toxic pollutants

An earlier Tellus report, *Valuation of Environmental Externalities for Electric Utility Resource Planning in Wisconsin*, calculated six costs of control for lead based on costs of emissions control of heavy metals in five different facilities and the Massachusetts DPU's costs of control of criteria air pollutants. The *Packaging Study* used the arithmetic mean of that range.[25] Our revised valuation uses one value from that range - the cost of control of lead at secondary lead smelters, $528 in 1993 dollars. This is a superior value for three reasons.

1)It was the only control cost for lead emitted from industrial processes; all others calculated the cost of lead based upon other pollutants or other types of lead emissions.

2)Three of the six costs of control for lead were calculated on a basis which is technically inconsistent with the *Packaging Study*'s hazard rankings. Correction of this error would have raised the arithmetic mean of the six values to $4000 per pound of lead, as opposed to the $1600 per pound published in the *Packaging Study*.

3)The smelter control cost is virtually identical to the geometric mean of the six (corrected)

---

[23]California Energy Commission, Docket Number 90-ER-92S. Order Adopting Residual Emission Values for SCAQMD. April 26, 1993.

[24]CA PUC 91-06-022, June 5, 1991.

[25]*Packaging Study*, Report #4, page 1-11, and *Environmental Externalities*, page 67.

values: this is arguably a more appropriate average, for widely disparate values, than the arithmetic mean used in the *Packaging Study*.

3. Implications of the new valuations

Table Appendix A-1 shows the new valuations (in 1993 $/pound of pollutant emitted) and compares them with the valuations shown in the *Packaging Study*.

Table Appendix A-1 - Comparison of impacts per pound of pollutant emitted

Pollutant	New Valuation (1993 $/lb)	Packaging Study Valuation (1992 $ /lb)
Carbon monoxide	$0.48	$0.42
NOx	$4.53	$3.63
SOx	$2.23	$5.87
Particulates	$1.30	$5.85
Volatile organic chemicals	$2.11	$2.50
Lead	$528	$1,600

The values for SOx, particulates and lead are much lower in the new valuation, while VOCs are slightly lower and NOx is higher. The carbon monoxide value has only been adjusted for inflation.

Table Appendix A-2 shows the effects of changing the valuations on the impacts per ton of packaging materials. For most materials, the new valuation is slightly less than half the *Packaging Study* valuation; glass and recycled paper decrease less significantly, since few toxics are emitted during their production. Toxics-intensive materials such as steel, virgin papers and plastics benefit to a greater extent from the revision of our valuation methods.

Table Appendix A-2 - Comparison of new environmental cost/ton with Packaging Study environmental cost/ton

Material Type	New Valuation	Packaging Study
PLASTIC		
HDPE	$128	$292
LDPE	$158	$344
PET	$331	$854
PP	$148	$367
PS	$162	$385
PVC	$1,714	$5,053
PAPER		
Bleached Kraft Paperboard	$121	$330
Unbleached Coated Folding Boxboard	$94	$269
Linerboard	$95	$273
Corrugating Medium	$49	$83
Unbleached Kraft Paper	$96	$277
Folding Boxboard from wastepaper	$76	$135
Linerboard from wastepaper	$77	$135
Corrugating Medium from wastepaper	$109	$183
Virgin Glass	$70	$85
Recycled Glass	$48	$55
Virgin Aluminum	$926	$1,933
Recycled Aluminum	$76	$313
Virgin Steel	$80	$230
Recycled Steel	$78	$222