# ESTIMATION OF THE EFFECTS OF VARIOUS MUNICIPAL WASTE MANAGEMENT STRATEGIES ON GREENHOUSE GAS EMISSIONS

**Contract No. K2093-3-2130**

Summary Report

prepared for

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Solid Waste Management Division

and

Natural Resources Canada
Alternative Energy Division

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**MARCH, 1994**

**PROJECT E.O. 92446**

## ACKNOWLEDGMENTS

This project was funded by Canada's Green Plan, through Environment Canada's Greenhouse Gas Program and by Natural Resources Canada.

Parts I and II of the study were carried out by Proctor & Redfern Limited in association with Ortech International. The Summary Report was prepared by Proctor & Redfern Limited.

We would like to thank the following industries and associations who provided valuable information on manufacturing and waste management processes:

Alcan Aluminum
Consumers Glass
Canadian Portland Cement Association
Canadian Cement Council
Dow Chemical Canada
Environment and Plastics Institute of Canada
Paper & Paperboard Environmental Council
Rubber Manufacturers Association

Lynne Patenaude
Scientific Authority
Solid Waste Management Division
Office of Waste Management
Environment Canada

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## ABSTRACT

The quantity of municipal solid waste (MSW) generated in Canada was estimated using data from waste sampling studies. The impact of various diversion and utilization options on the quantity and composition of waste that will require landfill in the year 2000 was projected, based on five different waste management scenarios postulated for achieving the Canadian Council of Ministers of the Environment (CCME) goal of 50% diversion from disposal in the year 2000, and two scenarios for achieving a diversion rate of 70% in that year. In addition, a scenario based on the diversion rate in 2000 remaining at 1992 levels was evaluated in order to provide an indication of the maximum greenhouse gas (GHG) emissions from landfilled waste in the year 2000. Under each scenario, different roles were assigned to source reduction, reuse, recycling, composting, incineration and landfill.

Various models for estimating GHG emissions from landfills were reviewed to identify a model which would take into account the change in waste composition under the different waste management scenarios as well as variations in climatic conditions across the country. An appropriate model was selected and suitable values for model constants were assigned based on waste composition and climatic conditions in each province.

The study findings indicate that in the year 2000, 65% of total methane emissions from landfill will be emitted due to MSW landfilled prior to 1991 and that despite the achievement of waste reduction targets, methane emissions from landfill in the year 2000 will exceed those in 1990.

Recognizing that alternatives to landfill may also impact GHG generation, a number of waste reduction and utilization options were identified for each waste type, and the GHG emissions associated with each option determined through the preparation of a GHG budget. In this process, GHG credits were assigned to waste management practices with material displacement and energy recovery potential. GHG debits were assigned to practices which impose increased transportation requirements, require energy input and/or produce a residue requiring disposal. Net GHG emissions were computed by combining the credits and debits accruing to each process and expressed in terms of 20-year and 100-year global warming potentials (GWPs). A life cycle approach was used to ensure that the total GHG emissions associated with each management option were accounted for.

It was found that waste reduction initiatives such as source reduction, reuse and recycling have the potential to significantly reduce the net GHG emissions associated with the production and disposal of products. All waste utilization options evaluated were found to result in lower net GHG emissions than those associated with the landfilling of mixed MSW.

## 1.0 INTRODUCTION

This summary report provides an overview of work carried out on the study entitled "Estimation of the Effects of Various Municipal Waste Management Strategies on Greenhouse Gas Emissions." The study was carried out in two parts. Detailed Part I and II reports are available separately.

Increased concentrations in the atmosphere of gases such as carbon-dioxide, methane, CFCs and nitrous oxide have the potential to affect the earth's heat balance by re-absorbing infra-red radiation transmitted from earth toward space. It is believed that the trapping of heat by these gases will result in an increase in the temperature of the earth, a phenomenon known as global warming. Gases which produce this effect are referred to as greenhouse gases.

The magnitude of the impact of different gases on global warming is expressed in terms of their global warming potentials. The GWP is defined as the time integrated change in "radiative forcing" due to the instantaneous release of 1 kg of a trace gas expressed relative to the radiative forcing from the release of 1 kg of $CO_2$ (IPCC, 1990). GWPs are a relative measure of the warming effects of gases and while the ranking of gases is widely accepted, there are many uncertainties associated with the exact magnitudes of the GWPs. In this study, the revised GWPs of well-mixed gases developed in 1992 (IPCC, 1992) have been used.

The decomposition of municipal solid wastes in landfills is a major source of GHG emissions, contributing as much as 38% of total anthropogenic emissions of methane and approximately 1% of total anthropogenic emissions of carbon-dioxide (Jaques, 1992). The rate of emission of GHGs at landfills is controlled by the rate at which micro-organisms metabolize the waste. The type of micro-organisms that dominate depend on the availability of oxygen. Recently landfilled waste serves as a substrate for mainly aerobic decomposition. The available oxygen is, however, rapidly depleted by the decay processes and anaerobic decomposition becomes dominant for the remaining life of the landfill. The initial aerobic period is negligible compared to the total active period of gas generation (approximately 50 years). Under anaerobic conditions, emissions from a landfill on average comprise approximately equal quantities of carbon-dioxide and methane with relatively minor amounts of other gases. Under aerobic conditions, bio-oxidation of methane results in the emission of mainly carbon-dioxide.

Alternatives to landfill, including waste reduction options such as source reduction, reuse, and recycling (the 3Rs), and waste utilization options such as composting, anaerobic digestion, pyrolysis, fermentation and incineration, may also result in a release of GHGs (carbon-dioxide, methane and nitrous oxide). The releases are associated with increased transportation requirements

of materials and wastes, changes made in production processes, and bio-degradation and combustion waste management processes. Waste reduction practices reduce total GHG emissions by reducing the quantity of new materials and products that must be produced. Waste utilization processes, on the other hand, are generally used to recover energy from wastes, thus reducing the requirement to produce energy from fossil fuels. An exception to this is composting, which is carried out to stabilize the wastes and produce a soil conditioner.

The impact of recovering energy from waste on GHG production can be evaluated by comparing the GHG emissions from the utilization process with the sum of emissions from landfilling of the waste and emissions from the combustion of conventional fuels to produce the same quantity of energy as would be produced by the utilization process. Where the emissions associated with the utilization process are less than the combined emissions of landfilling of waste and energy production, the utilization process is the more favourable option.

The objectives of this study were:

1. to estimate the contribution of landfilled MSW to GHG emissions in 1990 and 2000 and examine the change in year 2000 emissions associated with the adoption of different waste management scenarios; and

2. to quantify and compare the net GHG emissions associated with the management of different waste materials by alternative waste reduction and utilization options, using a life cycle approach.

The two study objectives are addressed in Parts I and II of the study, respectively.

The study results provide an indication of the impact of waste management policy changes on year 2000 GHG emissions, and a comparison of the performance of different waste management systems in terms of contribution to global warming, an environmental issue which continues to receive considerable attention.

Section 2.0 of the report outlines the methodology employed in the two parts of the study. Section 3.0 summarizes the study results and Section 4 presents the study conclusions.

# 2.0 METHODOLOGY

## 2.1 Estimation of GHG Emissions from Landfilled Waste

### 2.1.1 Current (1988, 1990, 1992) MSW Generation and Composition

In order to project waste quantities requiring landfill to the year 2000 and to develop waste management scenarios to that year, it was necessary to estimate current waste generation, composition and diversion. Waste generation and composition can be estimated using data from either waste sampling studies or from material consumption studies. Waste sampling studies examine the types and quantities of materials discarded into the waste stream. Material consumption studies on the other hand, investigate material production and consumption patterns to obtain an indication of the quantities and types of materials discarded. In this study, data generated by waste sampling studies is used to estimate the quantity and composition of MSW generated. This approach was selected because it provides empirical data on the waste stream, whereas material consumption studies are based on a postulated relationship between material production, consumption and waste generation.

The majority of the waste sampling studies carried out in Canada have been for municipalities in Ontario. The Regional Municipality of Ottawa-Carleton has recently completed a waste composition study (Stanley et al. 1992) and one is currently underway for the Greater Vancouver Regional District. However, results from these studies were not available in time for use in this study. Estimates of the quantity of MSW generated and its composition were, therefore, based on data obtained from the following waste composition studies carried out in Ontario:

<u>Residential Waste</u>

1. Waste composition study for three Ontario municipalities (Borough of East York in Metro Toronto with 96,500 residents, Town of Fergus in Wellington County with 7,000 residents, and the City of North Bay, with 51,500 residents) (Gore & Storrie, 1991).

2. Waste composition study undertaken by the City of Guelph as part of its wet/dry pilot project. This study provides data from 340 waste samples taken over a 51 week period between November 1989 and November 1990 (City of Guelph, 1991).

3. A two-month residential waste composition study carried out by the Centre and South Hastings County measuring waste generated from 50 households in the City of Trenton in April and May 1991 (Centre and South Hastings County, 1991).

4. Various waste sorting studies carried out by Proctor & Redfern in the Cities of Peterborough, Mississauga and Oakville.

IC&I Waste and C&D Wastes

1. The Metropolitan Toronto Solid Waste Composition Study (Proctor & Redfern et al. 1991).

2. Proctor & Redfern file notes developed through discussions with various regional municipalities.

To generate provincial estimates of waste generation and composition, the data from these studies were adjusted to take into account provincial differences in waste generation patterns as influenced by the proportion of the province's population residing in urban and rural areas and the average provincial per capita income relative to Ontario. Data in most of the studies mentioned above pertained to the period 1990/1991 and were used to produce waste quantity and composition estimates for 1990.

The quantity and composition of MSW generated in 1988 (which is the year adopted by the CCME as the base against which waste reduction is to be measured) was also estimated. For this purpose, it was assumed that the per capita generation rate in 1988 was equal to that in 1990. This assumption was based on the fact that there were no major new source reduction and reuse activities initiated during this period and that recycling of MSW only reduces the quantity of waste disposed and not the quantity generated. Estimates of MSW generated and MSW requiring disposal in 1992 were computed from the 1990 estimates based on assumptions of waste diversion achieved by different provinces in that year (assumed to be 21% for Ontario and 15% for all other provinces).

### 2.1.2 Projected MSW Generation and Composition to 2000

Theoretical waste generation was projected for the period from 1988 to 2000 by applying the per capita waste generation rate in 1988 to forecasted population. The theoretical waste generation is the quantity of waste that would be generated in the absence of any new source reduction and reuse initiatives between 1988 and 2000. Actual waste generation (or waste that requires disposal) is calculated by subtracting waste diverted from disposal by waste reduction practices, from theoretical waste generation. For the CCME target to be achieved, per capita waste requiring disposal in 2000 must be 50% or less of that in 1988.

The quantity of MSW requiring landfill in 2000 was calculated by postulating, in consultation with the Scientific Authority for the study, eight different waste management scenarios to the year 2000. The roles assigned to source reduction, reuse, recycling, composting, incineration and landfill in 2000 under each scenario are presented in Table 2.1. Scenarios A through E are based on the CCME waste reduction target of 50% being achieved in 2000 and were chosen to maximize the role of each management practice. Scenarios F and G assume that per capita waste generation is reduced by 70% by 2000. Scenario H was based on the assumption that the proportion of MSW diverted from landfill by 3Rs programs and the quantity of MSW incinerated remain constant at 1992 levels up to 2000.

Based on current trends, Scenario A is considered to be the most likely combination of diversion practices that will be employed to achieve the 50% target. Scenario B is identical to Scenario A, but assumes that all waste remaining after 3Rs will be incinerated and only incinerator ash will be landfilled. Scenarios C, D, and E maximize the contribution of reuse, recycling, and composting, respectively. These scenarios assume, for the purposes of the calculations, that incineration capacity will be available to manage 50% of residual waste in 2000. Scenario F assumes that all waste remaining after 3Rs will be incinerated and only ash will be landfilled. Scenario G assumes that all residual waste will be landfilled. Scenario H, assumes that only 17% (which is the estimated diversion rate for 1992) of waste generated between 1992 and 2000 is diverted by 3Rs programs. This scenario also assumes that no additional incineration capacity is created over the period 1992-2000. Scenario H represents the "worst-case" scenario, and was evaluated to define the upper bound of GHG emissions from landfills in 2000.

For each of the above scenarios, the quantity and composition of MSW requiring landfill over the period 1988 to 2000 were calculated assuming a linear growth in waste reduction and utilization initiatives to the year 2000.

The chemical composition of waste requiring landfill under each scenario was determined, since the GHG generation potential of waste is a function of the carbon content of the waste and the proportion associated with bio-degradable wastes. The chemical composition of the MSW stream and the manner in which it changes under each of the waste management scenarios was determined from the material composition of MSW requiring landfill in each year over the period 1988 to 2000, together with the average compositions of the individual waste materials obtained from literature.

**TABLE 2.1: WASTE MANAGEMENT SCENARIOS POSTULATED FOR THE YEAR 2000**

| Waste Management Scenario | Source Reduction | Reuse | Recycling | Composting | Incineration | Landfilling |
|---|---|---|---|---|---|---|
| Scenario A | 15% | 5% | 15% | 15% | 5% | 45% |
| Scenario B | 15% | 5% | 15% | 15% | 40% | 10% |
| Scenario C | 5% | 20% | 10% | 15% | 25% | 25% |
| Scenario D | 5% | 5% | 30% | 10% | 25% | 25% |
| Scenario E | 5% | 5% | 10% | 30% | 25% | 25% |
| Scenario F | 20% | 10% | 20% | 20% | 23% | 7% |
| Scenario G | 20% | 10% | 20% | 20% | 30% | - |
| Scenario H | 5% | 2% | 10% | <1% | 5% | 78% |

### 2.1.3 Selection of Model

A review of the literature was conducted to locate the most recent methodologies for estimating GHG emissions from landfilled wastes. Direct contact was also made with the U.S. EPA which is the major source of new developments in modelling GHG emissions from landfills.

The U.S. EPA is currently pursuing two methodologies. The first is the continued use of the Scholl-Canyon model which is a mechanistic first order decay model. The second method is a simple regression model based on measured landfill emissions data. In its present form, the only independent variable in the regression model is the total quantity of waste-in-place. The U.S. EPA is developing a statistical model based on multi-linear regression techniques. Since one of the

prime objectives of the present study is to estimate emission trends in terms of the waste composition and regional climate variability, it is essential that these factors be parameters of the selected model. This would imply the use of a multi-parameter regression model or the use of a mechanistic model based on the likely kinetics of the bio-degradation process. One of the most important requirements of this study is that it should be able to properly account for the change in GHG emissions as the MSW composition is varied under the waste management scenarios developed. As the multi-linear regression model is still under development, the Scholl-Canyon model was selected for this study. More sophisticated variants of the first order decay model were rejected since it was felt that, given the present understanding of the quantitative relationships between the model parameters and landfill conditions, their use was not warranted.

The Scholl-Canyon model has the form:

$$R(t) = k\, L_0 \exp(-kt)$$

where:

- R is the gas generation rate after a time t following the placement of an incremental amount of waste in the landfill at t=0; the equation is usually solved with time increments of one year;
- $L_0$ is the total gas production potential associated with the incremental (usually annual) amount of waste over the life of the landfill, and is generally estimated from the carbon content of the waste and the fraction of carbon which is associated with biodegradable material; and
- k is the rate constant describing the depletion of biodegradable waste with time and is a function of temperature and moisture in the landfill as well as the availability of nutrients, pH and the presence of toxins such as certain heavy metals.

### 2.1.4 Selection of Model Parameters

Use of the Scholl-Canyon model requires that the quantity of waste-in-place, the value of the rate constant k, and the value of the methane generation potential $L_o$, be specified. Since the active life of waste in a landfill (that is the time during which it contributes to GHG emissions), is of the order of 50 years, the annual waste landfilled in the previous 50-year period is required for each year of model computation. Even after 50 years, the model predicts some contribution of the aged waste to methane emissions, particularly for regions such as the prairies where a low k-value is used. Since one of the objectives of the study is to analyse trends in emissions from landfills, a constant period of 50 years is used. For example, the year 2000 estimate considers waste deposited

over the period 1951 to 2000. The Levelton (1991) study provides estimates of wastes landfilled annually over the period 1941 to 1990. For model input, Levelton data have been used for 1941 to 1987 together with estimates made in this study for 1988 to 2000.

Theoretical and measured values of $L_0$ are highly variable, ranging from approximately 100 to 250$m^3$ of methane per tonne of waste. For this study, $L_0$ was estimated using the following formula which is based on the stoichiometric conversion to methane of the biodegradable carbon in the waste using the gross chemical composition of the waste:

$$L_0 = M_c . F_b . S$$

where S is a stoichiometric factor (44/12 for $CO_2$ and 16/12 for $CH_4$), $M_c$ is the carbon content (tonne C/tonne waste) estimated from the gross formula composition of the waste, and $F_b$ is the biodegradable carbon fraction.

Using this method, changes in $L_0$ which result from the changing waste composition due to various diversion practices are included in the emission estimates. Table 2.2 summarizes the computed values of $L_0$ over the period 1990 to 2000 for each of the eight waste management scenarios.

Estimation of the rate constant k is more difficult as no explicit functional relationships for selecting values of k based on climate variables or waste compositions are available (Peer et al., 1991). Levelton (1991) assigned k-values to landfills in different parts of the country on the basis of mean daily temperature and mean annual precipitation. In this study, the provincial k values shown in Table 2.3 were assigned based on the Levelton values, except in the case of Quebec for which a higher value was selected. As most of the landfilled waste is in Southern Quebec, a value similar to that of Ontario was considered appropriate.

### 2.1.5 Uncertainty of GHG Emission Estimates

The major sources of uncertainty in the GHG emission estimates were identified, and the sensitivity of the emissions and trends, reported in this phase of the study, was examined.

## TABLE 2.2: METHANE GENERATION POTENTIALS ($L_0$): 1990 TO 2000 BY WASTE MANAGEMENT SCENARIO

| Year | Scenario | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | A: High Landfill | B: High Incineration | C: High Reuse | D: High Recycling | E: High Composting | F:70% Diversion & High Incln. | G:70% Diversion & High Landfill | H: Worst Case |
| 1990 | 195 | 195 | 195 | 195 | 195 | 195 | 195 | 195 |
| 1991 | 194 | 181 | 187 | 187 | 184 | 179 | 190 | 195 |
| 1992 | 190 | 164 | 176 | 177 | 172 | 162 | 183 | 195 |
| 1993 | 186 | 148 | 166 | 170 | 158 | 146 | 176 | 195 |
| 1994 | 183 | 132 | 155 | 159 | 146 | 129 | 170 | 195 |
| 1995 | 178 | 115 | 145 | 151 | 132 | 112 | 163 | 195 |
| 1996 | 175 | 99 | 136 | 141 | 120 | 95 | 157 | 195 |
| 1997 | 170 | 83 | 126 | 133 | 109 | 77 | 151 | 195 |
| 1998 | 167 | 65 | 116 | 124 | 95 | 61 | 144 | 195 |
| 1999 | 163 | 50 | 105 | 114 | 82 | 44 | 137 | 195 |
| 2000 | 160 | 33 | 97 | 106 | 69 | 27 | 131 | 195 |

**Note:** Units are $m^3$ CH4/tonne

**TABLE 2.3: COMPARISON OF K-VALUES USED WITH LEVELTON (1991) VALUES**

| | k-value this study ($yr.^{-1}$) | k-value Levelton (1991) |
|---|---|---|
| Newfoundland | 0.011 | 0.010-0.011 |
| Prince Edward Island | 0.011 | 0.011 |
| Nova Scotia | 0.011 | 0.011 |
| New Brunswick | 0.011 | 0.010-0.011 |
| Quebec | 0.024 | 0.010-0.020 |
| Ontario | 0.024 | 0.010-0.025 |
| Manitoba | 0.006 | 0.006 |
| Saskatchewan | 0.006 | 0.003-0.006 |
| Alberta | 0.006 | 0.003-0.006 |
| British Columbia | 0.028 | 0.010-0.028 |
| Northwest Territories | 0.003 | 0.003 |
| Yukon | 0.003 | 0.003 |

## 2.2 GHG Emissions from Various Waste Management Practices

### 2.2.1 Selection of Waste Material/Management Practice Combinations to be Evaluated

The waste material/management practice combinations to be evaluated were based upon waste categories developed earlier in the study for the purpose of estimating waste generation and composition. Material sub-categories (e.g. sub-category "newspapers" within the main category "paper") were examined to ascertain whether there is a significant difference between sub-categories with respect to:

- GHG production potential of the material processing operations; and
- waste management practices applicable to each sub-category.

Where the quantity of a sub-category in the waste stream is small or where no significant difference was found in either one of the above two parameters, materials were aggregated for ease of evaluation. In some instances, segregation of sub-categories was found to be necessary for waste reduction practices, but not for waste utilization technologies. An example of this is the segregation of the material category "paper" into the sub-categories "newspapers", "fine paper", and "corrugated cardboard" for the evaluation of source reduction (lightweighting fine paper, and newspaper), reuse (reuse of corrugated cardboard containers) and recycling (recycling of all grades of paper) options. However, only the main category "paper" is carried through in the evaluation of waste utilization and disposal technologies (composting, fermentation, anaerobic digestion, pyrolysis, incineration and landfill) because all paper exhibits similar characteristics during these processes. Waste management practices to be evaluated for each waste material were selected on the basis of :

- feasibility in terms of material and chemical composition; and
- availability of emission data based on demonstrated full or pilot scale performance.

Final selection of options to be evaluated was based upon input provided by Environment Canada and Natural Resources Canada. The waste management options selected for each material type are shown in Table 2.4.

Source reduction of paper, glass and plastic materials shown in this table refers to the lightweighting/downsizing of products. The impact of reusing corrugated containers, glass and PET bottles on GHG emissions was evaluated. Reuse of wood residues refers to the mulching process and that of tires to retreading . Recycling of paper, glass, metals, plastics and tires (crumb

## TABLE 2.4: WASTE MATERIAL /WASTE MANAGEMENT OPTION COMBINATIONS SELECTED FOR EVALUATION

| Waste Material | Source Reduction | Reuse | Recycling | Backyard Composting | Composting | Fermentation | Anaerobic Digestion | Mulching | Pyrolysis | Incineration | Landfill |
|---|---|---|---|---|---|---|---|---|---|---|---|
| PAPER | | | | | X | X | X | | X | X | X |
| Fine Paper | X | | X | | | | | | | | |
| Newspaper | X | | X | | | | | | | | |
| Corrugated | | X | X | | | | | | | | |
| GLASS | X | X | X | | | | | | | X | X |
| FERROUS | | | X | | | | | | | X | X |
| NON-FERROUS | | | X | | | | | | | X | X |
| PLASTICS | | | | | | | | | X | X | X |
| HDPE | | | X | | | | | | | | |
| PET | X | X | X | | | | | | | | |
| FOOD & YARD WASTE | | | | X | X | | X | | X | X | X |
| WOOD RESIDUES | | | | | X | X | X | X | X | X | X |
| TIRES | | X | X | | | | | | X | X | X |
| MIXED MSW | | | | | X | | X | | X | X | X |

rubber production) was evaluated.

Composting (carried out to stabilize the waste and produce a soil conditioner) and anaerobic digestion (to produce biogas, a mixture of methane, carbon-dioxide and various other gases) of mixed paper, food and yard waste, wood residues and mixed MSW are examined. Backyard composting which differs from centralized composting in that it eliminates the need to transport the waste materials to a facility, does not require the input of process energy, and results in a process that is only partially aerobic due to the absence of mechanical aeration was evaluated. Fermentation of paper, and wood residues for the production of industrial alcohol were also examined.

Pyrolysis of paper, plastics, food & yard wastes, wood residues, tires and mixed MSW to produce a solid residue containing almost pure carbon, pyrolytic oils, and a low BTU gas consisting of hydrogen, methane, carbon-dioxide and carbon-monoxide, were evaluated.

Final disposal of all materials by incineration and landfill were evaluated, including several energy recovery options.

### 2.2.2 Preparation of GHG Budgets

In order to assess the net effect of different waste reduction practices (source reduction, reuse, recycling) on the production of GHG emissions, the impact of the management practice on the product's life cycle must be examined. For instance, the recycling of paper and aluminum reduces the quantity that must be produced from virgin materials thus avoiding the GHG emissions associated with the extraction of these materials. On the other hand, the recycling process itself may impose additional collection, separation and transportation requirements which result in the production of GHG emissions.

Waste utilization technologies only impact the product life cycle stages that occur after a material is discarded. They affect the total quantity of GHG emissions produced by the processes necessary to meet the combined requirements of management of a waste and the production of other products and/or energy which can potentially be recovered from the waste. For instance, the recovery of energy from waste eliminates the need to produce an equivalent amount of energy through conventional methods such as fossil fuel combustion. Similarly, carbon black produced by the pyrolysis of waste represents a raw material which can potentially be used in the production of rubber products.

GHG emissions associated with the manufacturing stage of a product life cycle are usually a result of the need to meet the process's energy requirements. Energy required for production may be provided through the combustion of fossil fuels such as coal and petroleum, purchase of electricity (which in turn may be generated by thermal, nuclear, or hydro-electric power stations), or through the utilization of excess heat produced in another part of the process. In a few instances, $CO_2$, $CH_4$ or $N_2O$ may be generated directly by the production process.

A methodology was developed which assigns a GHG credit to those elements of a waste management practice that have the potential to reduce the quantity of GHG emissions produced during one or more stages in the product life cycle, and a GHG debit to those elements that have the potential to increase the quantity of greenhouse gases produced. A GHG budget is then drawn up, and all debits and credits assigned to a given waste management practice are combined to give the net GHG emissions associated with the practice.

In order to compute the GHG budgets for waste reduction and utilization processes, it was necessary to make a number of assumptions relating to:

- distances from the point of material extraction to production facilities;
- distances from MSW collection points to recovery and utilization facilities and from the facilities to landfill;
- the type of energy or fuel used for processes that require energy input (electrical, coal, oil, natural gas, etc.); and
- the nature of the energy source replaced when energy is recovered.

As the scope of this study is national, and the GHG emissions calculated do not relate to existing waste management practices in any specific province but rather, attempt to quantify the impact of managing one tonne of a material by a given process anywhere in Canada, the assumption has been made that the management facilities are available within a reasonable distance from the point of generation. The specific distances assumed for each process are given in the individual GHG budgets contained in the Part II report.

For energy consumption, wherever possible, emission factors associated with the form of energy typically consumed in that process was used. For processes that use electricity, it was assumed that the electricity was produced from the Canadian "pool" of fuels for electricity production (including nuclear and hydro-electric power generation). Where the production of a material is

known to be carried out in only one province, as in the case of aluminum production in Quebec, the provincial "pool" of fuels for electricity production was used. For those processes for which information on fuel type used was not available, it was assumed that the Canadian "pool" of industrial fuels is used.

Similarly, for energy recovery processes, except where the nature of the fuel replaced was known, it was assumed that energy produced from the Canadian "pool" of fuels (for electricity production or direct firing) was replaced by the recovered energy.

Emission factors used in the study to estimate transportation and energy related emissions were based upon emission factors reported by Jaques (1992). Emission factors developed for different modes of transportation and sources of energy are summarized in Tables 2.5 and 2.6 below.

**TABLE 2.5: GHG EMISSION FACTORS FOR TRANSPORTATION**

| | $CO_2$ | $CH_4$ | $N_2O$ |
|---|---|---|---|
| | (tonne/tonne-km)[1] | (kg/tonne-km)[1] | (kg/tonne-km)[1] |
| **Trucks - Heavy-Duty Diesel** | | | |
| Fuel-Based Emission Factors[2] | 2.73 t/kL | 0.20 kg/kL | 0.40 kg/kL |
| Mass/Distance-Based Emission Factors[3] | $5.5 \times 10^{-5}$ | $4.0 \times 10^{-6}$ | $8.0 \times 10^{-6}$ |
| **Rail - Diesel** | | | |
| Mass/Distance-Based Emission Factors[4] | $3.6 \times 10^{-5}$ | $2.9 \times 10^{-6}$ | $5.2 \times 10^{-6}$ |
| **Water - Diesel** | | | |
| Mass/Distance-Based Emission Factors[5] | $1.3 \times 10^{-5}$ | $1.1 \times 10^{-6}$ | $2.0 \times 10^{-6}$ |

Notes:

1. Unless otherwise stated.
2. Jaques 1992.
3. Calculated from fuel-based emission factors using a fuel efficiency of 2.5 km/L (Jaques 1992) and a typical load of 20 tonnes.
4. Calculated assuming an energy consumption of $0.51 \times 10^{-6}$ TJ/tonne-km (ORF 1976).
5. Calculated assuming an energy consumption of $0.18 \times 10^{-6}$ TJ/tonne-km (SRI International 1983).

## TABLE 2.6: CANADIAN INDUSTRIAL POOL EMISSION FACTORS (1990)

(Jaques, 1992)

| Description | Demand | $CO_2$ | $CH_4$ | $N_2O$ | $CO_2$ | $CH_4$ | $N_2O$ |
|---|---|---|---|---|---|---|---|
| | TJ | kt | t | t | t/TJ | kg/TJ | kg/TJ |
| Industrial (no electric) | 1,388,873 | 75,350 | 3,000 | 2,000 | 54.25 | 2.16 | 1.44 |
| Industrial (with elec.) | 2,043,556 | 114,958 | 3,422 | 2,844 | 56.25 | 1.67 | 1.39 |
| Electric Canadian Pool | 1,551,647 | 93,873 | 1,000 | 2,000 | 60.50 | 0.64 | 1.29 |
| Electric Ontario Pool | 476,515 | 25,935 | 276 | 523 | 54.43 | 0.58 | 1.16 |
| Electric Quebec Pool | 531,627 | 1,430 | 15 | 30 | 2.69 | 0.03 | 0.06 |
| Electric BC Pool | 49,275 | 1,227 | 13 | 26 | 24.90 | 0.27 | 0.53 |

Notes: The emission factors in this table are based on purchased energy values (either primary of secondary).

GHG emissions associated with different production and utilization processes were estimated through a detailed analysis of the process. The computation basis for the budgets is one tonne of material in the MSW stream.

For the purposes of comparing processes that produce differing proportions of $CO_2$, $CH_4$, and $N_2O$, the emissions are converted into $CO_2$ equivalencies using the GWPs of the different gases as scaling factors. The 20 and 100-year GWPs shown below have been used.

| | $CO_2$ | $CH_4$ | $N_2O$ |
|---|---|---|---|
| 20-year | 1 | 34 | 250 |
| 100-year | 1 | 11 | 270 |

The major considerations in the preparation of GHG budgets for the different waste reduction and utilization practices are discussed below.

## Waste Reduction Practices

**Source Reduction:** In the preparation of a GHG budget for the source reduction of one tonne of waste material, the emissions associated with raw material extraction, production (direct process emissions and emissions associated with the provision of energy) and transportation of one tonne of material were assigned to the process as a material displacement credit.

**Reuse:** This is based on substituting one tonne of reusable products for an equivalent weight of one-way products fulfilling an equivalent function. The GHG budget for reuse contains a credit to account for the fact that each time a product is reused, the production of a new product is avoided (material displacement credit), and debits to account, where applicable, for the increased transportation (transportation debit) and product preparation (process energy debit) requirements of the reuse system. The net GHG emissions have been calculated "on a per time reused" basis.

**Recycling:** A GHG credit is assigned to recycling for its displacement of virgin materials. This credit takes into account losses of material in the system, and the point at which the recovered material enters the production cycle of the new product. Debits are assigned to take into account the increased material transportation and preparation requirements. The net GHG emission associated with the recycling of various materials have been calculated on a "per time recycled" basis.

## Waste Utilization/Disposal Technologies

**Composting:** Composting of wastes produces a soil conditioner that improves a soil's structure and optimizes soil tilth. It can act as a slow release fertilizer reducing the requirement of fertilizer application. However, due to the lack of data on the quantity of each type of nutrient replaced, a material displacement credit has not been assigned to composting. Debits have been assigned to take into account direct emissions of $CO_2$ from the composting process (process emissions), energy requirement of a mechanized composting operation (process energy debit) and transportation to a composting facility (transportation debit).

**Anaerobic Digestion:** Anaerobic digestion of biodegradable organic wastes results in the production of biogas which can be used as a medium to high energy fuel. The gas produced can be used for steam production or power generation, or upgraded to pipeline quality by removing the $CO_2$. In this study, anaerobic digestion with the production of electricity using a gas turbine has been evaluated. No material displacement credit has been assigned to this process. A process emissions debit was assigned to account for emissions from the combustion of the biogas. An energy credit was assigned to account for the displacement of energy produced by conventional

fuels. Debits were assigned to take account of disposal of process residue and transportation of waste materials to the anaerobic digestion facility.

**Fermentation:** The fermentation of wastes to industrial alcohol, which can be blended with gasoline to produce gasohol, was analyzed. A process emissions debit was assigned to account for emissions of GHGs from the fermentation process; an energy credit was assigned to account for gasoline replaced; and debits were assigned to account for the energy requirements of the fermentation process, transportation of wastes to the fermentation facility and disposal of residue. Fermentation produces large quantities of residues (consisting of unhydrolysed cellulose and non-fermentable sugars) relative to other utilization processes. The economic viability of fermentation is often dependent upon the utilization of the residue either as animal feed or as a fuel. In this analysis, it has been assumed that the residue will be used as a fuel. The residue disposal debit assigned to this process is therefore the net debit after subtracting the energy credit accruing to the residue.

**Pyrolysis:** Pyrolysis of organic wastes produces a low energy gas stream (consisting of hydrogen, methane, carbon monoxide, carbon dioxide and other gases), a tar/oil (pyrolytic oils) stream, an aqueous liquid stream and a solid residue containing char and inert materials. The proportion and composition of the products of pyrolysis are a function of operating parameters. The process can be operated to maximize gas or oil production. In this analysis, for materials other than tires, it has been assumed that the pyrolysis process is operated for maximum gas production and that the gas is combusted to raise steam. The pyrolysis of tires is usually operated to produce a liquid fuel that can be used to replace fuel oil. Pyrolysis can theoretically be used to recover carbon black. However, the recovery of carbon black of sufficiently good quality from pyrolysis is problematic. Reports of recovery of carbon black of marketable quality were only found for the pyrolysis of tires (Roy et al, 1990). A material displacement credit for carbon black was therefore assigned only to this feedstock. The material displacement credit was based upon the GHG emissions associated with the production of carbon black from refinery catalytic cracker residue. For other waste materials, a displacement credit for the reported recovery of ferrous metal was assigned. An energy credit was assigned to take account of the energy produced from the gas and oil streams. Debits were assigned to account for the process's energy and transportation requirements.

**Incineration:** The incineration of components of MSW with and without energy recovery were considered. Three alternative energy recovery options were investigated: energy recovery for the generation of electricity, energy recovery for the production of steam, and use of refuse-derived fuel (RDF) in cement kilns. No material displacement credit was assigned to incineration based on the assumption that any materials recovered during feed preparation would have been recovered

regardless of whether the material is subsequently incinerated. Energy displacement credits were assigned to the three alternative energy recovery options based on the following efficiencies of energy recovery: 20% for electricity generation, 70% for steam production and 85% for use of RDF in cement kilns. Debits were assigned to account for the emissions of GHGs from the combustion process, the requirement of process energy and the transportation of MSW to the facility and of ash to landfill. The incineration of inert materials, such as glass and metals, produces GHG emissions during transportation, and results in an increase in process energy required since the temperature of the inert material must be brought up to the incineration temperature.

**Landfill:** The GHG emissions associated with the landfilling of refractory materials (inert materials and organic materials which are not readily degradable, such as plastics and synthetic fabrics) are limited to transportation emissions. A debit was assigned to account for this. For bio-degradable materials, a further debit was assigned to account for the emissions of GHGs from the landfilled waste. Landfill gas can be collected and either flared or used to recover energy. Four different landfill options were considered: no gas recovery (as the majority of landfills in Canada do not currently have gas collection systems in place), gas collection with no energy recovery (flaring), gas collection with electric energy recovery, gas collection with non-electric energy recovery. A collection efficiency of 50% of landfill gas generated was assumed. As GHG emissions from landfills are produced over a period of over 50 years, $CO_2$ equivalencies were only computed on the basis of the 100-year GWP.

# 3.0 STUDY RESULTS

## 3.1 GHG Emissions from Landfills in 2000

Estimates of this study indicate that over 23 million tonnes of MSW were generated in Canada in 1992. Tables 3.1, and 3.2 present estimates of MSW generation in 1992 by material type for residential and IC&I wastes, respectively. Of total MSW generated in 1992, it is estimated that approximately four million tonnes were diverted from disposal by 3Rs programs and approximately one million tonnes were incinerated. The remaining MSW (approximately 18 million tonnes) was landfilled. It is projected that if 3Rs programs divert 50% of waste from disposal by the year 2000 as targeted, approximately 14 million tonnes of MSW will require disposal in that year. Assuming that waste incineration capacity remains at existing levels, approximately 12.5 million tonnes will be landfilled. Under Scenario B which is based upon increasing incineration capacity to the maximum extent possible for MSW remaining after achieving the 50% reduction target, the quantity of waste requiring disposal in landfills in the year 2000 would be reduced to 2.7 million tonnes. This provides an indication of the theoretical lower bound of the quantity of MSW requiring landfill in 2000. By contrast, Scenario H defines the upper bound by assuming that diversion rates and incineration capacity remain at 1992 levels. Under this scenario the quantity of waste requiring landfill in 2000 is estimated to be approximately 22 million tonnes.

The composition of landfilled waste, which is a major consideration in the evaluation of the GHG production potential, also varies significantly with the waste diversion and utilization process selected. As is to be expected, the biodegradable carbon content of waste in landfills was lowest for scenarios that postulated incineration of all residual waste remaining after 3Rs (scenarios B & F) since only ash was landfilled, followed by the scenario which focuses on composting to achieve the year 2000 waste reduction target and allows for a moderate increase in incineration capacity (Scenario E).

Methane emissions to the atmosphere, as well as the methane generation potential of in-place waste were estimated for 1990 and for each year to 2000 for all eight scenarios defined in the study. The total 1990 methane emission from landfills in Canada in 1990 was estimated to be 839 ktonne which is approximately 22% of total anthropogenic emissions of methane. This estimate is lower than the earlier estimate of 38% reported in the Environment Canada report on GHG emission estimates for 1990 (Jaques, 1992).

# TABLE 3.1: ESTIMATED RESIDENTIAL MSW GENERATION AND COMPOSITION IN CANADA IN 1992

| Material Category | Percentage of Residential Waste | | Kilograms Generated Per Capita Per Year | | Quantity of Residential Waste (tonnes) | |
|---|---|---|---|---|---|---|
| PAPER | 26.9% | | 100.8 | | 2,766,140 | |
| Newsprint | | 10.1% | | 37.9 | | 1,039,179 |
| Fine Paper/ledger | | 1.4% | | 5.3 | | 144,572 |
| Magazine | | 3.2% | | 12.1 | | 330,868 |
| Waxed/Plastic | | 1.7% | | 6.4 | | 174,684 |
| Boxboard | | 3.6% | | 13.5 | | 371,821 |
| Kraft | | 1.1% | | 4.1 | | 113,081 |
| OCC | | 2.4% | | 8.9 | | 243,688 |
| Tissues | | 3.0% | | 11.3 | | 310,693 |
| Other Paper | | 0.4% | | 1.4 | | 37,552 |
| GLASS | 5.8% | | 21.8 | | 599,541 | |
| Beverage Containers (NA) | | 0.7% | | 2.6 | | 71,394 |
| Food Containers | | 2.9% | | 10.8 | | 295,672 |
| Beverage Containers (Al) | | 1.7% | | 6.3 | | 172,341 |
| Other | | 0.6% | | 2.2 | | 60,134 |
| FERROUS | 3.8% | | 14.2 | | 390,589 | |
| Beverage Containers | | 0.5% | | 1.8 | | 49,383 |
| Food Containers | | 2.1% | | 7.8 | | 215,247 |
| Aerosol | | 0.2% | | 0.8 | | 20,900 |
| Other | | 1.0% | | 3.8 | | 105,059 |
| NON-FERROUS | 0.7% | | 2.6 | | 70,586 | |
| Beverage Containers (NA) | | 0.1% | | 0.5 | | 14,146 |
| Beverage Containers (Al) | | 0.1% | | 0.4 | | 10,276 |
| Other Packaging | | 0.1% | | 0.4 | | 10,276 |
| Other | | 0.3% | | 1.3 | | 35,889 |
| PLASTICS | 6.5% | | 24.3 | | 665,744 | |
| Rigid Containers - HDPE | | 0.8% | | 3.0 | | 82,472 |
| Other Rigid Containers | | 0.2% | | 0.7 | | 20,551 |
| PET Containers | | 0.2% | | 0.7 | | 19,275 |
| Polystyrene | | 0.5% | | 1.9 | | 51,378 |
| Bags - LDPE | | 1.2% | | 4.6 | | 127,243 |
| Other Film | | 0.7% | | 2.6 | | 71,878 |
| Other Packaging | | 0.2% | | 0.8 | | 21,616 |
| Other Plastics | | 2.6% | | 9.9 | | 271,332 |
| ORGANICS | 38.4% | | 143.9 | | 3,948,739 | |
| Food Waste | | 21.8% | | 81.5 | | 2,237,373 |
| Yard Waste | | 16.7% | | 62.4 | | 1,711,366 |
| WOOD WASTE | 1.5% | | 5.6 | | 153,449 | |
| RUBBLE | 1.4% | | 5.2 | | 144,012 | |
| DIAPERS | 3.3% | | 12.4 | | 340,187 | |
| TEXTILES | 3.5% | | 13.1 | | 360,705 | |
| HHW | 0.3% | | 1.1 | | 31,542 | |
| BULKY ITEMS | 5.0% | | 18.6 | | 509,380 | |
| MISCELLANEOUS | 2.9% | | 10.7 | | 294,896 | |
| TOTAL | 100.0% | | 374 | | 10,275,508 | |

| | |
|---|---|
| Population (1992) | 27,445,000 |

RES92B.XLS

# TABLE 3.2: ESTIMATED IC&I MSW GENERATION AND COMPOSITION IN CANADA IN 1992

| Material Category | Percentage of IC&I Waste | Kilograms Generated Per Capita Per Year | Quantity of IC&I Waste (tonnes) |
|---|---|---|---|
| PAPER | 39.8% | 188.52 | 5,174,020 |
| Corrugated Containers | 13.3% | 62.8 | 1,724,673 |
| Newspapers | 4.4% | 20.9 | 574,891 |
| Fine Paper | 3.5% | 16.8 | 459,913 |
| Mixed Paper | 8.8% | 41.9 | 1,149,782 |
| Magazines | 1.8% | 8.4 | 229,956 |
| Boxboard | 3.5% | 16.8 | 459,913 |
| Telephone Books | 0.9% | 4.2 | 114,978 |
| Books - general | 1.8% | 8.4 | 229,956 |
| Other | 1.8% | 8.4 | 229,956 |
| GLASS | 2.7% | 12.57 | 344,935 |
| Clear | 1.6% | 7.5 | 206,961 |
| Coloured | 0.5% | 2.5 | 68,987 |
| Other | 0.5% | 2.5 | 68,987 |
| FERROUS | 4.4% | 20.95 | 574,891 |
| Beverage & Food Cans | 1.3% | 6.3 | 172,467 |
| Drums | 0.1% | 0.4 | 11,498 |
| Scrap | 0.4% | 2.1 | 57,489 |
| Other | 2.6% | 12.1 | 333,437 |
| NON-FERROUS | 0.6% | 2.93 | 80,485 |
| Beverage & Food Cans | 0.4% | 2.1 | 57,489 |
| Other | 0.2% | 0.8 | 22,996 |
| PLASTICS | 8.8% | 41.89 | 1,149,782 |
| Rigid Plastic Containers | 1.8% | 8.4 | 229,956 |
| Plastic Films | 3.5% | 16.8 | 459,913 |
| Other Plastics | 3.5% | 16.8 | 459,913 |
| ORGANICS | 10.6% | 50.3 | 1,379,739 |
| WOOD | 6.2% | 29.3 | 804,848 |
| CONSTRUCTION & DEMOLITION | 16.0% | 75.8 | 2,079,722 |
| Wood Waste | 5.9% | 28.1 | 771,039 |
| Rubble, Aggregate | 3.1% | 14.7 | 403,177 |
| Drywall | 1.3% | 6.1 | 168,213 |
| Paper & Paperboard | 1.2% | 5.7 | 155,600 |
| Metals | 1.2% | 5.6 | 153,605 |
| Other C&D Waste | 3.3% | 15.6 | 428,087 |
| MISCELLANEOUS | 10.9% | 51.5 | 1,414,232 |
| TOTAL | 100.0% | 474 | 13,002,653 |

| | |
|---|---|
| Population (1992) | 27,445,000 |

ICI92B.XLS

Table 3.3 shows the quantity of MSW requiring landfill under each scenario evaluated and the corresponding increase(+)/decrease(-) in GHG emissions in 2000 relative to 1990.

**TABLE 3.3: PROJECTIONS OF THE QUANTITY OF WASTE REQUIRING LANDFILL AND EFFECT ON GHG EMISSIONS IN 2000 FOR SCENARIOS A TO H**

| | MSW Landfilled | Increase (+)/Decrease in GHG Emissions (2000:1990) |
|---|---|---|
| **50% Diversion** | | |
| **Scenario A** High Landfill | 12,593 | 16% |
| **Scenario B** High Incineration | 2,738 | -3% |
| **Scenario C** High Reuse | 7,567 | 4% |
| **Scenario D** High Recycling | 7,084 | 5% |
| **Scenario E** High Composting | 6,879 | 2% |
| **70% Diversion** | | |
| **Scenario F** High Incineration | 1,658 | -4% |
| **Scenario G** High Landfill | 8,291 | 8% |
| **Scenario H** Worst Case | 22,163 | 44% |

In the year 2000, approximately 640 ktonnes of methane (representing almost 65% of total methane emissions) are emitted due to MSW landfilled prior to 1991. The maximum reduction potential in 2000 of waste management policy changes is therefore about 35%. Under Scenarios B and F, small reductions in methane emissions relative to 1990 levels are obtained. Scenario E, which maximizes composting and allows for a moderate increase in incineration, produces only a marginal increase in emissions. In the worst case, where the proportion of waste diverted from landfill by waste reduction and incineration remains at 1992 levels results in a 44% increase in emissions relative to 1990 (Scenario H). The results of this part of the study also demonstrate that even if the 50% waste diversion target is achieved, if the majority of the remaining waste continues to go to landfill, a significant increase in methane emissions (16% under Scenario A) will occur.

## 3.2 Comparison of GHG Emissions from Various Waste Management Practices

Table 3.4 presents the GHG budgets developed for waste reduction options. GHG budgets for waste utilization options are presented in Tables 3.5 and 3.6 for bio-degradable and refractory materials, respectively. Individual GHG emission credits and debits assigned are shown in these tables in terms of kilograms of $CO_2$ $CH_4$ and $N_2O$ for each material/management option combination. Table 3.7 shows the increase/decrease in GHG emissions, expressed as kilograms of $CO_2$ based on 100-year GWPs, associated with the management of one tonne of waste material by a given waste management process. In Tables 3.4 to 3.7, positive figures denote the production of GHG emissions, while negative figures represent a saving in GHG emissions.

Source reduction of all materials results in savings in GHG emissions as a result of "avoided production". For example, it is estimated that for every tonne of glass reduced at source through the lightweighting of containers, a savings in GHG emissions equivalent to 1,057 kg of $CO_2$ is obtained.

# TABLE 3.4 : GHG BUDGETS FOR WASTE REDUCTION OPTIONS

| Material/Management Option | Material Displacement Credit | | | Process Energy Debit | | | GHG (Energy) Credit | | | Transportation Debit | | | Residue Disposal Debit | | | Process Emissions Debit | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O |
| **Paper** | | | | | | | | | | | | | | | | | | |
| Source Reduction | | | | | | | | | | | | | | | | | | |
| - Newsprint | -1,540 | -0.02 | -0.07 | | | | | | | | | | | | | | | |
| - Fine Paper | -3,690 | -0.03 | -0.20 | | | | | | | | | | | | | | | |
| Reuse | | | | | | | | | | | | | | | | | | |
| - Corrugated Boxes | -1,480 | -0.13 | -0.08 | | | | | | | 30 | 0.00 | 0.00 | 0.2 | 0.00 | 0.00 | | | |
| Recycle | | | | | | | | | | | | | | | | | | |
| - Newsprint | -1,130 | -0.02 | -0.05 | 330 | 0.01 | 0.01 | | | | 10 | 0.00 | 0.00 | | | | | | |
| - Fine Paper | -3,280 | -0.03 | -0.19 | 330 | 0.01 | 0.01 | | | | 10 | 0.00 | 0.00 | 0.2 | 0.00 | 0.00 | | | |
| - Corrugated Boxes | -3,130 | -0.03 | -0.02 | 330 | 0.01 | 0.01 | | | | 10 | 0.00 | 0.00 | | | | | | |
| **Glass** | | | | | | | | | | | | | | | | | | |
| Source Reduction | -1,040 | -0.03 | -0.06 | | | | | | | | | | | | | | | |
| Reuse | -590 | -0.02 | -0.01 | 20 | 0.00 | 0.00 | | | | 30 | 0.00 | 0.00 | 0.2 | 0.00 | 0.00 | | | |
| Recycle | -160 | 0.00 | -0.01 | 4 | 0.00 | 0.00 | | | | 20 | 0.00 | 0.00 | 0.2 | 0.00 | 0.00 | | | |
| **Ferrous** | | | | | | | | | | | | | | | | | | |
| Recycle (steel cans) | -2,700 | -0.02 | -0.03 | 330 | 0.01 | 0.00 | | | | 10 | 0.00 | 0.00 | | | | | | |
| **Non-Ferrous** | | | | | | | | | | | | | | | | | | |
| Recycle (aluminum cans) | -5,800 | -0.11 | -0.20 | 200 | 0.01 | 0.00 | | | | 140 | 0.01 | 0.02 | | | | | | |
| **Plastics** | | | | | | | | | | | | | | | | | | |
| Source Reduction | | | | | | | | | | | | | | | | | | |
| - PET | -2,856 | -0.15 | -0.15 | 0 | 0.00 | 0.00 | | | | 0 | 0.00 | 0.00 | | | | | | |
| Reuse | | | | | | | | | | | | | | | | | | |
| - PET | -1,142 | -0.06 | -0.06 | 120 | 0.00 | 0.00 | | | | 61 | 0.03 | 0.06 | 0.2 | 0.00 | 0.00 | | | |
| Recycling | | | | | | | | | | | | | | | | | | |
| - PET | -1,885 | -0.10 | -0.10 | 304 | 0.01 | 0.01 | | | | 69 | 0.00 | 0.01 | | | | | | |
| - HDPE | -1,419 | -0.06 | -0.06 | 304 | 0.01 | 0.01 | | | | 69 | 0.00 | 0.01 | | | | | | |
| **Tires** | | | | | | | | | | | | | | | | | | |
| Reuse (retreading) | -2,988 | -0.09 | -0.07 | 0.000 | 0.00 | 0.01 | | | | 6.786 | 0.01 | 0.34 | | | | | | |
| Recycling | | | | | | | | | | | | | | | | | | |
| - coarse Shred | -270 | 0.00 | 0.00 | 3 | 0.00 | 0.00 | | | | 17 | 0.04 | 0.84 | 3 | 0.00 | 0.00 | | | |
| - 5x5 cm chip | -270 | 0.00 | 0.00 | 17 | 0.00 | 0.00 | | | | 17 | 0.04 | 0.84 | 3 | 0.00 | 0.00 | | | |
| - 2.5 x 2.5 cm chip | -270 | 0.00 | 0.00 | 99 | 0.00 | 0.00 | | | | 17 | 0.04 | 0.84 | 3 | 0.00 | 0.00 | | | |
| - crumb (0.65 to 1.25 cm) | -270 | 0.00 | 0.00 | 281 | 0.01 | 0.01 | | | | 17 | 0.04 | 0.84 | 3 | 0.00 | 0.00 | | | |
| - cryogenic granulate | -270 | 0.00 | 0.00 | 662 | 0.03 | 0.02 | | | | 17 | 0.04 | 0.84 | 3 | 0.00 | 0.00 | | | |

Notes: debits represent the generation of GHG emissions and are shown as positive numbers.
credits represent savings in GHG emissions and are shown as negative numbers.

## TABLE 3.4 CONT'D : GHG BUDGETS FOR WASTE REDUCTION OPTIONS

| Material/Management Option | Net Emissions | | | CO2 Equivalencies (20 years) | CO2 Equivalencies (100 years) |
|---|---|---|---|---|---|
| | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CO2 |
| **Paper** | | | | | |
| Source Reduction | | | | | |
| - Newsprint | -1,540 | -0.02 | -0.07 | -1,557 | -1,558 |
| - Fine Paper | -3,690 | -0.03 | -0.20 | -3,741 | -3,744 |
| Reuse | | | | | |
| - Corrugated Boxes | -1,450 | -0.13 | -0.08 | -1,474 | -1,473 |
| Recycle | | | | | |
| - Newsprint | -787 | -0.01 | -0.05 | -799 | -800 |
| - Fine Paper | -2,937 | -0.02 | -0.18 | -2,983 | -2,987 |
| - Corrugated Boxes | -2,787 | -0.02 | -0.17 | -2,831 | -2,834 |
| **Glass** | | | | | |
| Source Reduction | -1,040 | -0.03 | -0.06 | -1,056 | -1,057 |
| Reuse | -540 | -0.02 | -0.01 | -543 | -542 |
| Recycle | -136 | 0.00 | 0.00 | -137 | -137 |
| **Ferrous** | | | | | |
| Recycle (steel cans) | -2,360 | -0.01 | -0.02 | -2,366 | -2,367 |
| **Non-Ferrous** | | | | | |
| Recycle (aluminum cans) | -5,460 | -0.10 | -0.18 | -5,508 | -5,509 |
| **Plastics** | | | | | |
| Source Reduction | | | | | |
| - PET | -2,856 | -0.15 | -0.15 | -2,899 | -2,898 |
| Reuse | | | | | |
| - PET | -961 | -0.02 | 0.00 | -962 | -961 |
| Recycling | | | | | |
| - PET | -1,512 | -0.09 | -0.08 | -1,536 | -1,536 |
| - HDPE | -1,046 | -0.05 | -0.05 | -1,059 | -1,059 |
| **Tires** | | | | | |
| Reuse (retreading) | -2,981 | -0.07 | 0.27 | -2,916 | -2,909 |
| Recycling | | | | | |
| - Coarse Shred | -246 | 0.04 | 0.84 | -35 | -19 |
| - 5x5 cm chip | -233 | 0.04 | 0.84 | -22 | -6 |
| - 2.5 x 2.5 cm chip | -150 | 0.04 | 0.84 | 62 | 77 |
| - crumb (0.65 to 1.25 cm) | 31 | 0.05 | 0.85 | 245 | 261 |
| - cyrogenic granulate | 412 | 0.06 | 0.86 | 628 | 644 |

Notes: debits represent the generation of GHG emissions and are shown as positive numbers.
credits represent savings in GHG emissions and are shown as negative numbers.

# TABLE 3.5: GHG BUDGETS FOR WASTE UTILIZATION OPTIONS FOR BIO-DEGRADABLE MATERIALS

| Material/Management Option | Material Displacement Credit | | | Process Energy Debit | | | GHG (Energy) Credit | | | Transportation Debit | | | Residue Disposal Debit | | | Process Emissions Debit | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O |
| **Paper** | | | | | | | | | | | | | | | | | | |
| Centralized Composting | 0 | 0.00 | 0.00 | 7.2 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 5 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 320 | 0.00 | 0.00 |
| Anaerobic Digestion | 0 | 0.00 | 0.00 | 30 | 0.00 | 0.00 | -51 | 0.00 | 0.00 | 5 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 285 | 0.00 | 0.00 |
| Fermentation | 0 | 0.00 | 0.00 | 588 | 0.02 | 0.02 | -841 | -0.79 | -0.34 | 3 | 0.00 | 0.00 | 632 | 0.00 | 0.00 | 827 | 1.00 | 0.34 |
| Pyrolysis | 0 | 0.00 | 0.00 | 40 | 0.00 | 0.00 | -454 | -0.02 | -0.01 | 4 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 1,462 | 0.00 | 0.00 |
| Incineration | | | | | | | | | | | | | | | | | | |
| - Electricity Generation | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -203 | 0.00 | 0.00 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 1,499 | 0.00 | 0.00 |
| - Steam Production | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -636 | -0.03 | -0.02 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 1,499 | 0.00 | 0.00 |
| - RDF in a Cement Kiln | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -1,058 | -0.02 | -0.01 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 1,499 | 0.00 | 0.00 |
| - No Energy Recovery | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 1,499 | 0.00 | 0.00 |
| Landfill | | | | | | | | | | | | | | | | | | |
| - Electricity Generation | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | -460 | -0.01 | -0.01 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 1,118 | 136.00 | 0.00 |
| - Non-electric energy | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | -419 | -0.02 | -0.01 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 1,118 | 136.00 | 0.00 |
| - No energy recovery | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 3 | 0.00 | 0.00 | | | | | | |
| - No Gas Recovery | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 745 | 272.00 | 0.00 |
| **Wood Residues** | | | | | | | | | | | | | | | | | | |
| Anaerobic Digestion | 0 | 0.00 | 0.00 | 30 | 0.00 | 0.00 | -113 | 0.00 | 0.00 | 5 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 227 | 0.00 | 0.00 |
| Mulching | 0 | 0.00 | 0.00 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 31 | 11.00 | 0.00 |
| Centralized Composting | 0 | 0.00 | 0.00 | 7 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 5 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 250 | 0.00 | 0.00 |
| Fermentation | 0 | 0.00 | 0.00 | 346 | 0.01 | 0.01 | -495 | -0.47 | -0.20 | 3 | 0.00 | 0.00 | 906 | 0.00 | 0.00 | 469 | 0.00 | 0.20 |
| Pyrolysis | 0 | 0.00 | 0.00 | 40 | 0.00 | 0.00 | -505 | -0.02 | -0.01 | 4 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 1,453 | 0.00 | 0.00 |
| Incineration | | | | | | | | | | | | | | | | | | |
| - Electricity Generation | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -225 | 0.00 | 0.00 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 1,452 | 0.00 | 0.00 |
| - Steam Production | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -706 | -0.03 | -0.02 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 1,452 | 0.00 | 0.00 |
| - RDF in a Cement Kiln | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -1,058 | -0.02 | -0.01 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 1,499 | 0.00 | 0.00 |
| - No Energy Recovery | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 1,499 | 0.00 | 0.00 |
| Landfill | | | | | | | | | | | | | | | | | | |
| - Electricity Generation | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | -460 | -0.01 | -0.01 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 1,118 | 136 | 0.00 |
| - Non-electric energy | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | -419 | -0.02 | -0.01 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 1,118 | 136 | 0.00 |
| - No energy recovery | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 3 | 0.00 | 0.00 | | | | | | |
| - No Gas Recovery | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 745 | 272 | 0.00 |
| **Food & Yard Waste** | | | | | | | | | | | | | | | | | | |
| - Centralized Composting | 0 | 0.00 | 0.00 | 7 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 5 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 350 | 0.00 | 0.00 |
| - Backyard Composting | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 360 | 7.30 | 0.00 |
| - Anaerobic Digestion | 0 | 0.00 | 0.00 | 30 | 0.00 | 0.00 | -31 | 0.00 | 0.00 | 5 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 318 | 0.00 | 0.00 |
| - Pyrolysis | 0 | 0.00 | 0.00 | 40 | 0.00 | 0.00 | -151 | -0.01 | 0.00 | 4 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 603 | 0.00 | 0.00 |
| - Pyrolysis | | | | | | | | | | | | | | | | | | |
| Incineration | | | | | | | | | | | | | | | | | | |
| - Electricity Generation | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -68 | 0.00 | 0.00 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 620 | 0.00 | 0.00 |
| - Steam Production | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -212 | -0.01 | -0.01 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 620 | 0.00 | 0.00 |
| - RDF in a Cement Kiln | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -353 | -0.01 | -0.01 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 620 | 0.00 | 0.00 |
| - No Energy Recovery | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 620 | 0.00 | 0.00 |
| Landfill | | | | | | | | | | | | | | | | | | |
| - Electricity Generation | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | -178 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 433 | 53 | 0.00 |
| - Non-electric energy | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | -162 | -0.01 | 0.00 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 433 | 53 | 0.00 |
| - No energy recovery | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 433 | 53 | 0.00 |
| - No Gas Recovery | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 288 | 105 | 0.00 |

Notes: debits represent the generation of GHG emissions and are shown as positive numbers.
credits represent savings in GHG emissions and are shown as negative numbers.

## TABLE 3.5: GHG BUDGETS FOR WASTE UTILIZATION OPTIONS FOR BIO-DEGRADABLE MATERIALS (CONT'D)

| Material/Management Option | Net Emissions | | | CO2 Equivalencies (20 years) | CO2 Equivalencies (100 years) |
|---|---|---|---|---|---|
| | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CO2 |
| **Paper** | | | | | |
| Centralized Composting | 332 | 0.00 | 0.00 | 332 | 332 |
| Anaerobic Digestion | 269 | 0.00 | 0.00 | 270 | 270 |
| Fermentation | 1,209 | 0.23 | 0.02 | 1,214 | 1,213 |
| Pyrolysis | 1,052 | -0.02 | -0.01 | 1,049 | 1,049 |
| Incineration | | | | | |
| - Electricity Generation | 1,312 | 0.00 | 0.00 | 1,311 | 1,311 |
| - Steam Production | 878 | -0.02 | -0.02 | 874 | 874 |
| - RDF in a Cement Kiln | 456 | -0.02 | -0.01 | 452 | 452 |
| - No Energy Recovery | 1,515 | 0.00 | 0.00 | 1,515 | 1,515 |
| Landfill | | | | | |
| - Electricity Generation | 661 | 136.00 | 0.00 | - | 2,155 |
| - Non-electric energy | 702 | 135.98 | -0.01 | - | 2,196 |
| - No energy recovery | | | | - | |
| - No Gas Recovery | 748 | 272.00 | 0.00 | - | 3,740 |
| **Wood Residues** | | | | | |
| Anaerobic Digestion | 148 | 0.00 | 0.00 | 149 | 149 |
| Mulching | 38 | 11.00 | 0.00 | 422 | 158 |
| Centralized Composting | 262 | 0.00 | 0.00 | 262 | 262 |
| Fermentation | 1,229 | 0.01 | 0.01 | 1,232 | 1,232 |
| Pyrolysis | 992 | -0.02 | -0.01 | 988 | 989 |
| Incineration | | | | | |
| - Electricity Generation | 1,242 | 0.00 | 0.00 | 1,241 | 1,241 |
| - Steam Production | 761 | -0.03 | 0.00 | 756 | 756 |
| - RDF in a Cement Kiln | 456 | -0.02 | -0.01 | 452 | 452 |
| - No Energy Recovery | 1,515 | 0.00 | 0.00 | 1,515 | 1,515 |
| Landfill | | | | | |
| - Electricity Generation | 661 | 136 | 0.00 | - | 2,155 |
| - Non-electric energy | 702 | 136 | -0.01 | - | 2,196 |
| - No energy recovery | | | | - | |
| - No Gas Recovery | 748 | 272 | 0.00 | - | 3,740 |
| **Food & Yard Waste** | | | | | |
| - Centralized Composting | 362 | 0.00 | 0.00 | 362 | 362 |
| - Backyard Composting | 360 | 7.30 | 0.00 | 616 | 440 |
| - Anaerobic Digestion | 322 | 0.00 | 0.00 | 322 | 322 |
| - Pyrolysis | 496 | 0.00 | 0.00 | 496 | 496 |
| Incineration | | | | | |
| - Electricity Generation | 568 | 0.00 | 0.00 | 569 | 569 |
| - Steam Production | 424 | -0.01 | 0.00 | 422 | 422 |
| - RDF in a Cement Kiln | 283 | -0.01 | -0.01 | 282 | 282 |
| - No Energy Recovery | 636 | 0.00 | 0.00 | 636 | 636 |
| Landfill | | | | | |
| - Electricity Generation | 258 | 53 | 0.00 | - | 840 |
| - Non-electric energy | 274 | 53 | 0.00 | - | 856 |
| - No energy recovery | 436 | 53 | 0.00 | - | 1,019 |
| - No Gas Recovery | 291 | 105 | 0.00 | - | 1,446 |

Notes: debits represent the generation of GHG emissions and are shown as positive numbers.
credits represent savings in GHG emissions and are shown as negative numbers.

## TABLE 3.6: GHG BUDGETS FOR WASTE UTILIZATION OPTIONS FOR REFRACTORY MATERIALS

| Material/Management Option | Material Displacement Credit | | | Process Energy Debit | | | GHG (Energy) Credit | | | Transportation Debit | | | Residue Disposal Debit | | | Process Emissions Debit | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CH4 | kg N2O |
| **Inerts (Glass, Metals)** | | | | | | | | | | | | | | | | | | |
| Incineration | 0 | 0.00 | 0.00 | 33 - 52 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 0 | 0.00 | 0.00 |
| Landfill | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 3 | 0.00 | 0.00 | 0.0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 |
| **Plastics** | | | | | | | | | | | | | | | | | | |
| Pyrolysis | 0 | 0.00 | 0.00 | 40 | 0.00 | 0.00 | -884 | -0.04 | -0.02 | 4 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 2,082 | 0.00 | 0.00 |
| Incineration | | | | | | | | | | | | | | | | | | |
| - Electricity Generation | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -394 | 0.00 | -0.01 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 2,156 | 0.00 | 0.00 |
| - Steam Production | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -1,238 | -0.05 | -0.03 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 2,156 | 0.00 | 0.00 |
| - RDF in a Cement Kiln | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -2,059 | -0.04 | -0.03 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 2,156 | 0.00 | 0.00 |
| - No Energy Recovery | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 2,156 | 0.00 | 0.00 |
| Landfill | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 |
| **Tires** | | | | | | | | | | | | | | | | | | |
| Pyrolysis | -2,052 | 0.00 | 0.00 | 40.000 | 0.00 | 0.00 | -1750 | -0.04 | -0.02 | 4 | 0.00 | 0.00 | 0.000 | 0.00 | 0.00 | 1786 | 0.00 | 0.00 |
| Incineration | | | | | | | | | | | | | | | | | | |
| - Electricity Generation | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -599 | -0.01 | -0.01 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 2,515 | 0.00 | 0.00 |
| - Steam Production | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -1,253 | -0.05 | -0.03 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 2,515 | 0.00 | 0.00 |
| - RDF in a Cement Kiln | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | -1,962 | -0.04 | -0.03 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 2,515 | 0.00 | 0.00 |
| - No Energy Recovery | 0 | 0.00 | 0.00 | 13 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 2 | 0.00 | 0.00 | 1 | 0.00 | 0.00 | 2,515 | 0.00 | 0.00 |
| Landfill | 0 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0.000 | 0.00 | 0.00 | 3 | 0.00 | 0.00 | 0 | 0.00 | 0.00 | 0.000 | 0.00 | 0.00 |

Notes: debits represent the generation of GHG emissions and are shown a
credits represent savings in GHG emissions and are shown as negative numbers.

## TABLE 3.6: GHG BUDGETS FOR WASTE UTILIZATION OPTIONS FOR REFRACTORY MATERIALS (CONT'D)

| Material/Management Option | Net Emissions | | | CO2 Equivalencies (20 years) | CO2 Equivalencies (100 years) |
|---|---|---|---|---|---|
| | kg CO2 | kg CH4 | kg N2O | kg CO2 | kg CO2 |
| **Inerts (Glass, Metals)** | | | | | |
| Incineration | 33 - 52 | 0.00 | 0.00 | 33 - 52 | 33 - 52 |
| Landfill | 3 | 0.00 | 0.00 | 3 | 3 |
| **Plastics** | | | | | |
| Pyrolysis | 1,242 | -0.03 | -0.02 | 1,235 | 1,236 |
| Incineration | | | | | |
| - Electricity Generation | 1,778 | 0.00 | -0.01 | 1,776 | 1,776 |
| - Steam Production | 934 | -0.05 | -0.03 | 924 | 925 |
| - RDF in a Cement Kiln | 113 | -0.04 | -0.03 | 104 | 105 |
| - No Energy Recovery | 2,172 | 0.00 | 0.00 | 2,172 | 2,172 |
| Landfill | 3 | 0.00 | 0.00 | 3 | 3 |
| **Tires** | | | | | |
| Pyrolysis | -1,972 | -0.04 | -0.02 | -1,979 | -1,979 |
| Incineration | | | | | |
| - Electricity Generation | 1,932 | -0.01 | -0.01 | 1,929 | 1,928 |
| - Steam Production | 1,277 | -0.05 | -0.03 | 1,267 | 1,268 |
| - RDF in a Cement Kiln | 568 | -0.04 | -0.03 | 560 | 560 |
| - No Energy Recovery | 2,531 | 0.00 | 0.00 | 2,531 | 2,531 |
| Landfill | 3 | 0.00 | 0.00 | 3 | 3 |

Notes: debits represent the generation of GHG emissions and are shown as positive numbers.
credits represent savings in GHG emissions and are shown as negative numbers.

## TABLE 3.7: GHG IMPACT OF WASTE MATERIAL/WASTE MANAGEMENT OPTION COMBINATIONS EVALUATED

## IN TERMS OF CO2 EQUIVALENCIES (100 YEAR GWPs) (KG OF C02/TONNE OF WASTE MATERIAL)

| Waste Management Option / Waste Material | Source Reduction | Reuse | Recycle | Backyard Composting | Mulching | Compost | Fermentation | Anaerobic Digestion | Pyrolysis | Incineration | | | | Landfill | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | No Energy Recovery | Electricity Production | Steam Production | RDF in a Cement Kiln | No Gas Collection | Gas Collection | | |
| | | | | | | | | | | | | | | | No Energy Recovery | Non-Electric Energy Recovery | Electric Energy Recovery |
| PAPER | | | | | | 332 | 1,213 | 270 | 1,049 | 1,515 | 1,311 | 874 | 452 | 3,740 | 2,618 | 2,196 | 2,155 |
| Fine Paper | -3,744 | | -2,987 | | | | | | | | | | | | | | |
| Newspaper | -1,558 | | -800 | | | | | | | | | | | | | | |
| Corrugated | | -1,473 | -2,834 | | | | | | | | | | | | | | |
| GLASS | -1,057 | -542 | -137 | | | | | | | 53 | 53 | 53 | 53 | 3 | | | |
| STEEL CANS | | | -2,367 | | | | | | | 36 | 36 | 36 | 36 | 3 | | | |
| ALUMINUM CANS | | | -5,509 | | | | | | | 55 | 55 | 55 | 55 | 3 | | | |
| PLASTICS | | | | | | | | | 1,236 | 2,172 | 1,775 | 924 | 144 | 3 | | | |
| HDPE | | | -1,059 | | | | | | | | | | | | | | |
| PET | -2,899 | -961 | -1,536 | | | | | | | | | | | | | | |
| FOOD & YARD WASTE | | | | 440 | | 362 | | 323 | 496 | 636 | 568 | 423 | 282 | 1,446 | 1,019 | 856 | 840 |
| WOOD RESIDUES | | | | | 158 | 262 | 1,232 | 149 | 989 | 1,468 | 1,241 | 756 | 287 | 3,575 | 2,503 | 2,100 | 2,061 |
| TIRES**** | | -2,909 | -19 to 644** | | | | | | -1,979 | 2,531 | 1,928 | 1,268 | 560 | 3 | | | |
| MIXED MSW | | | | | | 210 | | 80 | 411 | 847 | 719 | 445 | 181 | 1,914 | 1,347 | 1,130 | 1,109 |

Notes: debits represent the generation of GHG emissions and are shown as positive numbers.
credits represent savings in GHG emissions and are shown as negative numbers.
*** for crumb size ranging from coarse shred to cryogenic granulate
**** reuse of tires refers to retreading; recycling to crumb rubber production

The substitution of one tonne of reusable products for an equivalent weight of one-way products results in savings in GHG emissions which are a function of the weight of the reusable product relative to that of the one way product, and the number of times the product is reused (the trippage rate). In calculating the net GHG emissions from reuse, the weight of reusable containers relative to that of one-way containers was taken into account. To make the results independent of the trippage rate, the net GHG emissions were calculated on a per time reused basis. For example, the substitution of one tonne of reusable glass bottles for that quantity of one-way bottles delivering the same amount of product, is estimated to result in the saving of 542 kg of $CO_2$, each time it goes through the reuse cycle.

In general, recycling also results in net savings in GHG emissions. Possible exceptions to this are cases where the recycling process requires a large energy input relative to that required for the manufacture of the displaced material. For example, in this study, the production of cryogenic granulate from tires is seen to result in a net increase in GHG emissions because reprocessing energy is high and no material displacement credit was assigned due to the large number of potential applications and the lack of information on the energy requirements of these processes. The total savings in GHG emissions resulting from recycling are a function of the number of times a material is recycled. There are limitations on the number of times a material is recycled which are determined by the material losses that occur each time one tonne of material collected at the curb is used to produce a new product. Product specifications may also impose limits on the maximum recycled content of products. The net GHG emissions/savings arising from recycling have been calculated on a per time recycled basis.

Aerobic composting of biodegradable materials results in the net production of GHG emissions. The magnitude of these emissions is relatively small, as the majority of GHG emissions from composting are $CO_2$. Backyard composting of food and yard wastes also results in the production of GHG emissions from the metabolization of the wastes by microorganisms. Due to the absence of mechanical mixing, the process is likely to be partially anaerobic, resulting in some methane production and net GHG emissions which are higher than those associated with aerobic composting.

Anaerobic digestion with combustion of the biogas produced for the recovery of energy, results in the smallest net production in GHG emissions relative to all other waste utilization processes. Mulching of wood residues results in a net production in GHG emissions associated with the biological degradation of the waste.

Fermentation of wood and paper and combustion of the resulting alcohol, results in relatively high emissions of GHGs. The estimate of the quantity of GHG emissions produced by fermentation is strongly influenced by the assumption made with respect to the utilization/disposal of the residual cellulosic cake which is estimated to account for between 40% and 60% of the waste material input. In this study, it has been assumed that this will be combusted for energy recovery.

Pyrolysis of MSW components results in a net production of GHG emissions in all cases except that of tires. These results are critically dependent upon the assumption made with respect to the recovery of materials from the process. The pyrolysis of tires shows a net saving in GHG emissions due to the assumption that carbon black of sufficiently good quality can be recovered using this feedstock. The effect of removing the material displacement credit assigned for the recovery of carbon black from tire pyrolysis would be a small positive net GHG emission (equivalent to 87 kg of $CO_2$ per tonne pyrolysed) for this process.

Incineration of mixed MSW and its components with no energy recovery, shows the largest net production of GHG emissions compared to other utilization processes. The net GHG emissions for biodegradable materials are, however, lower than those associated with landfilling of these waste materials. The reason for this is that $CO_2$ is the primary GHG generated by incineration while emissions from landfill contain approximately 50% methane, which has a 100-year GWP of 11 relative to $CO_2$. The recovery of energy reduces the magnitude of the net GHG emissions from incineration. The highest net GHG emission for the recovery options is associated with the production of electricity because of the relatively low efficiency of energy recovery associated with this process. The utilization of MSW components as refuse-derived fuel (RDF) in a cement kiln results in low net emissions of GHGs due to the high efficiency of energy recovery and the fact that the fuels replaced are fossil fuels (approximately 64% of which is coal) with relatively high GHG emission factors. The only waste utilization option with a lower net GHG emission than RDF use in cement kilns is anaerobic digestion.

The magnitude of the net GHG emissions from landfills is reduced by the collection and combustion of the GHG gas produced (through the conversion of methane to the more benign carbon-dioxide). If energy is recovered from the combustion process, the GHG emissions associated with landfilling are further reduced. However, based on a collection efficiency of 50%, the net GHG emissions associated with all landfill options are higher than those for any other waste management method. The impact of increasing the collection efficiency to 75% of gas generated, is to reduce the GHG emissions associated with the landfilling of mixed MSW with energy recovery by about 30%. In this case, the landfilling of mixed MSW with energy recovery

at 75% gas collection efficiency has a net GHG emission potential marginally lower than that of incineration with no energy recovery.

## 3.3 Interpretation of Results

### 3.3.1 Uncertainties in Landfill Gas Emission Estimates, 1990 and 2000

The major sources of uncertainty in the landfill gas emission estimates developed in this study are:

- completeness of the model formulation used in the estimations;
- uncertainty in the values of the parameters $L_o$ and k input to the model;
- uncertainty in the estimates of annual waste quantities landfilled; and
- uncertainty in the relative percentages of $CO_2$ and $CH_4$ in the landfill gas.

The Scholl Canyon model is a gross simplification of the complex physical and biological processes which are precursors to the $CH_4$ emissions process. This model has, however, been empirically fitted to individual landfill gas emissions data using the $L_o$ and k parameters as adjustable constants. The resulting range of values for these constants, so determined, is large. Unfortunately, there are no objective methods for predicting $L_o$ from waste composition and there are presently no established methods of relating k to landfill conditions.

It should be noted that since the model constants are determined by empirically fitting $L_o$ and k to landfill gas emissions data, the values chosen are not independent since a single emission rate can be modelled using a high value of $L_o$ and a low value of k, or equally, a low value of $L_o$ and a high value of k. The temporal trend in the emissions predicted by these two sets of values will, however, be different.

In this study, the $CH_4$ generation potential has been estimated independently of k. The $L_o$ values obtained for this study, using the balanced stoichiometric method, depend on the scenario and range from 69 to 195 $m^3$/tonne, with the value of 195$m^3$/tonne representing currently generated waste materials. Sensitivity tests detailed in the Part I report indicate that the percentage change in emissions over the period 1990 to 2000 is relatively insensitive to the choice of k, and the ranking of the effectiveness of the several scenarios in reducing $CH_4$ emissions from landfills is unchanged.

Other uncertainties are associated with the quantity of waste landfilled and the estimated degradable carbon in the waste which is related to the estimation of $L_o$. There is a direct linear

relationship between $CH_4$ emissions and the percentage uncertainty in the quantity landfilled and fraction of degradable carbon.

### 3.3.2 Impact of Major Assumptions on the GHG Emission Estimates for Alternative Management Practices

As discussed in Section 2.2.2, the GHG budgets for waste reduction and utilization processes, were based on a number of assumptions relating to:

- distances from the point of material extraction to production facilities;
- distances from MSW collection points to recovery and utilization facilities and from the facilities to landfill;
- the type of energy or fuel used for processes that require energy input (electrical, coal, oil, natural gas, etc.); and
- the nature of the energy source replaced when energy is recovered.

Since the GHG emissions from transportation were found to be typically less than 1% of GHG emissions associated with energy consumption and recovery, the impact of transportation assumptions on the net GHG emissions calculated for the different processes was found to be negligible.

The impact of the assumptions made in the preparation of the GHG budgets, with respect to the type of energy consumed and type of energy replaced, are significant. These assumptions were necessary in the light of the generic nature of the analyses undertaken in this study. For example, the net GHG emissions from an energy recovery process will be reduced significantly if it is assumed that the fuel substituted is coal rather than the Canadian "pool" of energy sources which includes electricity generated at hydro-electric power stations. A change in the assumptions made in the preparation of the GHG budgets may impact the ranking of waste management options presented in this study. To overcome this shortcoming, the assumptions made in the preparation of the individual GHG budgets have been clearly laid out in the detailed Part II report to enable the GHG emissions to be recalculated on the basis of site-specific assumptions, if required.

The comparison of GHG emissions from landfilled waste, which persist over many years, with other sources of emissions which are emitted to the atmosphere instantaneously (as in combustion) or over a relatively short period of time (as in composting), is problematic. For instantaneous emissions, GWPs as $CO_2$ equivalents over a time period of 20, 100 or 500 years, can be used as a basis for comparison. In the case of emissions due to a specific amount of

waste landfilled, however, the effective GWP of this waste would be a summation (or integration) of the GWP of individual annual landfill emissions over the active life of the landfill. Presently, only the 20, 100 and 500 year GWPs for instantaneous emissions have been published. These represent the integrated global warming effect of a gas emitted at a point in time over periods of 20, 100 and 500 years. To obtain an index for comparing landfills with other emission sources in this study, it has been assumed that the full GHG emission potential of a material will be realized in the year of placement. This has the effect of overestimating the short term GHG impact of landfilling. Comparison of GHG emissions from landfilling with emissions from other sources has therefore only been done using 100 year GWPs.

The net GHG emissions associated with the management of one tonne of waste material by different management methods is also subject to uncertainties associated with the use of GWPs to calculate $CO_2$ equivalencies. The net emissions from various waste reduction and utilization processes are therefore presented in terms of both the individual greenhouse gases and their 20-year and 100-year GWPs to enable recalculation of the results, should at some future date the GWPs be revised.

## 4.0 STUDY CONCLUSIONS

The results of this study indicate that despite the achievement of waste reduction targets, large quantities of MSW will require disposal in the year 2000. The composition of MSW that requires disposal, and its potential for releasing GHGs to the atmosphere, is significantly affected by the nature of waste reduction and waste utilization initiatives emphasized. The magnitude of the increase in GHG emissions from landfills relative to generation in 1990, can be reduced by increasing the diversion of bio-degradable materials through composting and utilizing residual waste for energy recovery. However, the influence of MSW landfilled prior to 1991 on GHG emissions in 2000 limits the short term emission reduction potential of waste management policy changes.

Waste reduction initiatives such as source reduction, reuse and recycling have the potential to significantly reduce the net GHG emissions associated with the production, and disposal of products. Exceptions to this are processes which require large energy inputs relative to the energy for virgin production.

The energy recovery options for mixed MSW can be ranked from the practice with the lowest net GHG emissions to the highest, as follows:

- anaerobic digestion with energy recovery;
- use as RDF in cement kilns;
- incineration with steam production;
- pyrolysis;
- fermentation;
- incineration with electricity generation.

In addition to reducing GHG emissions from landfills, the recovery of energy from residual waste will result in a net reduction in total GHG emissions associated with the management of MSW, since all energy recovery options considered in this study result in net GHG emissions which are lower than those associated with landfilling of MSW.

This suggests that to reduce the impact of MSW management on global warming the quantity of waste landfilled in the future should be minimized. Recognizing that waste-in-place in landfills is still associated with significant methane emissions, a short term solution would be to encourage landfill gas recovery.

The results of the study, can be used to optimize the waste management system for specific materials by allowing comparison between different combinations of waste management practices. For example, the impact of the following two management systems for the diversion of glass can be compared: lightweighting, recycle, landfill vs. reuse, recycle, landfill. Finally, the methodology developed in this study can be used to evaluate the impact of waste management options on GHG production for proposed waste management facilities taking into account site and application-specific factors for transportation, material and energy replaced and energy consumed.

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