Final Report

Assessing the Environmental Effects of Disposal Alternatives for Household Batteries

Institute for Risk Research
University of Waterloo
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INSTITUTE FOR RISK RESEARCH

ASSESSING THE ENVIRONMENTAL EFFECTS OF DISPOSAL ALTERNATIVES FOR HOUSEHOLD BATTERIES

Final Report

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Prepared for:

Canadian Battery Manufacturer's Association
(CBMA)

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EXECUTIVE SUMMARY

The CBMA funded the Institute for Risk Research (IRR) to undertake an independent investigation into the impacts of used household battery disposal and management practices on the environment, including an evaluation of the potential risks associated with the identified options. This report discusses the issues relating to used dry-cell battery disposal practices, their potential impacts on the environment, the potential risks to humans, and offers recommendations for what is considered as acceptable disposal practices for used household batteries. The steps followed in the investigation are shown in Figure ES-1.

The batteries of concern in this study include the dry-cells classified by the AAA, AA, C, D and 9 volt sizes/formats, and which belong to the alkaline, zinc-carbon/zinc-chloride and nickel-cadmium family of batteries. Current estimates for the Canadian household battery market for the selected dry-cells are given in Table ES-1. Alkaline batteries represent the largest share of household battery market; these are used for several applications, including radios, toys, flashlights and portable appliances. Zinc-carbon batteries are generally less powerful, although there are some "high performance" cells in this set. Nickel-cadmium (Ni-Cad) rechargeable batteries which are a small portion of the total market, can be recharged up to 1000 times and are also becoming popular. Ni-Cads are found in such appliances as power tools and portable vacuum cleaners; most of these are imbedded in the appliances and are not user-replaceable. Alkaline batteries have, in the past, contributed to the amount of mercury in household waste streams; however, with the current practice adopted by the battery industry to reduce the amount of mercury, this situation has been significantly improved. On the other hand, nickel-cadmium batteries are presently a major contributor of cadmium to the waste streams.

Potential Health and Environmental Impacts from Disposal Practices:

The metals of potential concern present in the household batteries studied are cadmium, manganese, mercury, nickel and zinc. In whatever disposal or management practice that is adopted for the used household batteries, there is the potential for the release of metals which might affect human health directly or indirectly, or which might impact the environment. Currently, used household batteries are almost exclusively disposed of in domestic garbage which is eventually incinerated or landfilled. Lately, the idea of used battery collection, separation and possible recycling is becoming another focus of attention. Table ES-2 summarizes the results of risk characterization carried out for the various disposal alternatives; no quantitative evaluation was performed for the recycling options.

Figure ES-1
Steps in the Evaluation of the Impacts of Used Household Batteries on the Environment.

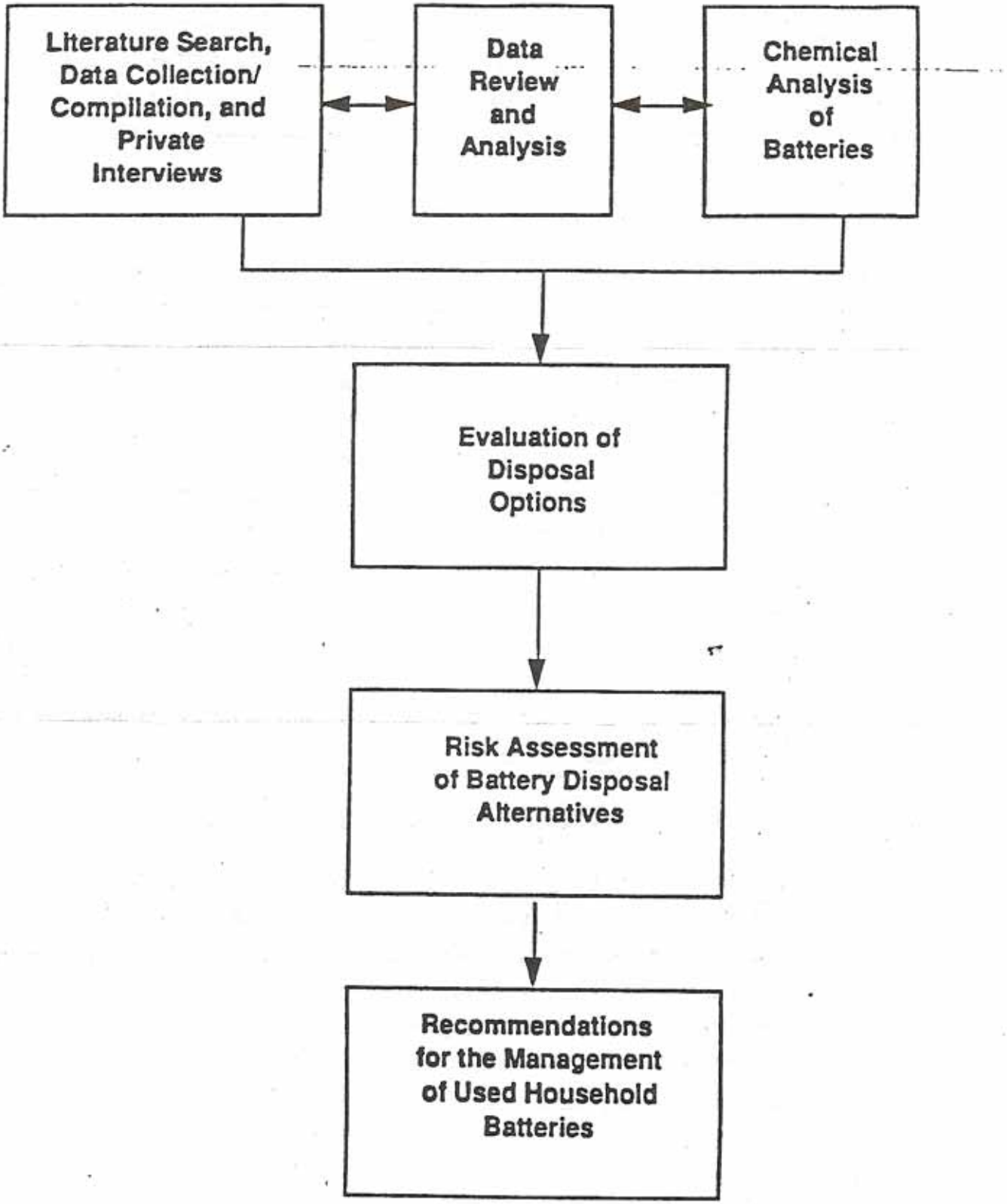

Table ES-1
Canadian Household Battery Market Estimates (1990/1991)

Battery Type	Battery Size							
	Total in Millions	AAA	AA	C	D	9 Volt	Other	% of Totals
Alkaline	100	9	60	9	8	9	5	65.4
Zinc Carbon	25	1	21	7	9	6	1	16.3
Zinc Chloride	20							13.1
Nickel-Cadmium	8	0.5	5	1	1	0.5	--	5.2
Totals	153	10.5	86	17	18	15.5	6	
% of Totals		6.9	56.2	11.1	11.8	10.1	3.9	

Table ES-2
Risk Comparison for Disposal Alternatives

Disposal Option	Quantitative Risk Measure*	
	Hazard Index	Carcinogenic Risk
Landfilling	0.4 (1.6)**	0.0 (0.0)
Incineration	3.5 (0.0)	5.3×10^{-3} (1.4×10^{-6})
Combined Landfilling and Incineration	1.9	2.7×10^{-3}
Recycling	Not Quantified	Not Quantified

* Shows value for the most sensitive potential receptor, i.e., population indicating highest risk measure.
** Numbers in parentheses show values for typical/actual case studies for selected disposal options; these are represented by Waterloo Landfill Site (Waterloo) and Tricil SWARU incinerator facility (Hamilton), both in Ontario.

Notes: Acceptable Hazard Index ≤ 1.
Acceptable Carcinogenic Risk Range is 10^{-4} to 10^{-7}.

Theoretically, incineration of the batteries of concern in this study will present the greatest risks; in practice, mixed with municipal solid waste (MSW), these batteries may safely be incinerated with MSW without any significant risks. Landfilling of the dry-cell batteries with MSW will generally

present no significant risks of concern. Although the recycling of the household batteries has no been quantified, the qualitative indicators are that it is not the best disposal option for the alkaline (manganese) and the zinc-carbon/zinc-chloride cells. Ni-Cad recycling programs may, however be a worthwhile effort.

Used household batteries will undergo degradation under landfill conditions, with the rate and degree of decay depending on the battery types, state of charge in battery and the physical conditions at the landfill site. Beyond the degradation process, it is important to determine if metals from the batteries will leach from a landfill into an underlying aquifer. Several variables, including landfill management practices will determine this. Indeed, under ideal landfill conditions, metals will not leach rapidly through landfills and soils into groundwater. On the other hand, metals do not decompose or degrade, and thus have the potential of leaching into aquifers over long periods of time.

Metals are of critical concern in an incineration process since they are not combustible. Thus, the protection of the environment in an incineration management option depends on the ability of the incinerator to capture and remove metals from air emissions. Although technologies exist for most metals removal, which makes such a process available, albeit expensive, it is not completely effective in abating mercury emissions due to the low vapor pressure of mercury. The presence of other metals, including cadmium in the incinerator fly ash arising, in part, from the incineration of household batteries, renders such ash potentially highly toxic and as a result, such ash may not be disposed of at municipal landfills.

Study Conclusions:

Several conclusions were made based on this investigation:

- The dry-cell batteries investigated (i.e. the alkaline, zinc-carbon/zinc chloride and Ni-Cads) do not generally represent a concentrated source of heavy metals in MSW.
- There is no clear evidence to suggest that the co-disposal of dry-cell batteries with MSW via incineration or landfilling presents environmental or health problems.
- Risks to the environment from battery disposal by landfilling and incineration are not likely to be significant. Thus, most household batteries may be safely disposed of in municipal landfills or municipal incinerators; Ni-Cads are better landfilled than incinerated unless recycled.
- At present "recycling" is more likely to present significant risks. There appear to be significant health-related problems associated with the separate collection, storage and disposal of most household batteries. However, recycling for Ni-Cads may be a more viable and desirable

measure to adopt. With the currently reduced levels of mercury in most primary cells (especially the alkaline and zinc-carbon/zinc-chloride batteries), recycling of alkaline and zinc-carbon/zinc-chloride cells is not necessary or needed.

Recommended Disposal Alternatives and Management Programs:

Mercury in household batteries has been drastically reduced in the past few years and should be reduced even further because of current research and development activities by the Canadian battery industry. Concern about alkaline household batteries being a major source of mercury in municipal solid waste is no longer true; in the past, these batteries had up to 1.5% mercury by weight, but today they have only 0.025% by weight and this is expected to go down further in the future. Most of the information filtering to the public as to the amount of mercury in alkaline batteries manufactured in Canada today is out of date, leading to over-reaction by environmentalists and the general public to a non-existent problem.

On the other hand, Ni-Cad batteries may be of concern and it may be desirable to recycle them. The collection and recycling of Ni-Cads is desirable, although effective technologies may not readily available at the present time. There is therefore the need to develop adequate programs and technologies, or better yet develop substitutes for Ni-Cads which do not contain cadmium. Indiscriminate policy decision aimed at all batteries, on the other hand, could be detrimental and would only result in ineffective and uneconomical programs at best, and be potentially hazardous and environmentally unsound at worst.

Based on the investigations carried out for the alkaline, zinc-carbon/zinc-chloride and Ni-Cad batteries, it is concluded that current disposal practices appear to be safe and adequate. Improvements may however be achieved by adopting the following recommendations:

- There is the need to educate the general public with respect to distinction between lead-acid automotive batteries and the various types of dry-cell household batteries.
- Further research needs to be conducted to determine the effect of household battery disposal on landfill leachate quality, and the potential impacts on groundwater resources.
- There should be a policy implemented, that requires all municipal incinerators to be equipped with wet gas scrubbers. In that case, mercury emitted during combustion of municipal waste can then be removed. Also, since cadmium is carcinogenic by the inhalation pathway, it is crucial that adequate scrubbers are used on municipal solid waste incinerators, that will capture as much of the fly ash as possible and minimize the amounts that could eventually reach potential human receptors. In the absence of that, Ni-Cads which may be a significant

contributor of cadmium to municipal solid waste may have to be removed from the wastes to be incinerated.

- There appear to be some potential health-related problems associated with the separate collection, storage and disposal of most household batteries. Thus, with the currently reduced levels of mercury in most primary cells (especially the alkaline and zinc-carbon/zinc-chloride batteries), recycling of alkaline and zinc-carbon/zinc-chloride cells is not presently necessary or needed. However, recycling of Ni-Cads may be a more viable and necessary measure to adopt.

Overall, none of the current disposal practices for the used dry-cell batteries investigated in this study present any real risks. However, it may be prudent and safety-effective to adopt the recommended management options described in Table ES-3.

Table ES-3
Recommended Management Methods for Used Dry-Cell Batteries

Battery Type	Preferred Management Option	Alternative Management Option	Comments
Alkaline (manganese)	Landfilling	Incineration	Neither landfilling or incineration of even concentrated forms appear to present any significant risks
Zinc-carbon/zinc chloride	Landfilling	Incineration	Neither landfilling or incineration of even concentrated forms appear to present any significant risks.
Ni-Cads	Recycling	Landfilling	Separate collection and recycling of Ni-Cads preferred due to potential risks from Cadmium.

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1.0 INTRODUCTION

All batteries are electrochemical energy storage units, in which electrons are passed between positive and negative electrodes through an electrolyte. A battery delivers electrical energy produced in an oxidation-reduction (redox) reaction to an external circuit or load. The energy content is dissipated after the battery is used for a period of time. The two major types of consumer batteries used in Canada are dry-cell (household) batteries and lead-acid (automotive) batteries. The focus of attention here will be on dry-cell batteries only. Various types of dry-cell batteries are in common use, each representing a different system of chemical materials. Used dry-cell batteries are generally disposed of in mixed municipal solid wastes from domestic and business garbage. Eventually, such wastes are managed via incineration or landfilling. Recycling is becoming a popular option for the management of wastes. In any one of the applicable waste management options, metals from the household batteries may pose different health and environmental concerns through different means.

The batteries of concern in this study include the dry-cells classified by the AAA, AA, C, D and 9 volt sizes/formats, and which belong to the alkaline, zinc-carbon/zinc-chloride and nickel-cadmium family of batteries. Alkaline batteries represent a larger share of household battery market; these are used for several applications, including radios, toys, flashlights and portable appliances. Zinc-carbon batteries are generally less powerful, although there are some "high performance" cells in this set. Nickel-cadmium (Ni-Cad) rechargeable batteries which are a small portion of the total market can be recharged up to 1000 times and are also becoming popular. Ni-Cads may be found in such appliances as power tools and portable vacuum cleaners; most of these are imbedded in appliances and are not user-replaceable.

The CBMA funded the Institute for Risk Research (IRR) to undertake an independent investigation into the impacts of used household battery disposal and management practices on the environment, including an evaluation of the potential risks associated with possible disposal methods. The IRR, by contractual arrangement, has the final say over the content and publications of the study findings.This report discusses the issues relating to used dry-cell battery disposal practices, their potential impacts on the environment, the potential risks to humans, and offers recommendations for what is considered as acceptable disposal practices for used household batteries.

.1 Background to the Study

ne general public is becoming increasingly concerned with the issues relating to chemical ntamination of the environment. This public concern about the effects of potentially toxic emicals entering the environment has resulted in an increasing demand for more stringent ntrols over chemical waste disposal practices. One area of concern is the disposal of hazardous bstances in landfills; an important area of current public attention is with regards to the perceived gative environmental impacts of landfilling dry-cell batteries. However, environmental impact udies reviewed by the Canadian Battery Manufacturer's Association (CBMA) on the disposal of y-cell batteries in municipal waste management systems do not indicate that such disposal actices pose significant threats to the environment. In fact, Canadian battery manufacturers have ken steps towards becoming environmentally responsible producers through the implementation significant reductions (up to about 98% reduction over the past decade alone) in the mercury ntent of household batteries, mercury being one of the most toxic metals used in household tteries. The CBMA believes that the public perception of risks to the environment and human alth due to disposal of household batteries in municipal landfills is affected by:

- outdated general public information regarding the composition of household batteries;
- a lack of understanding of the distinction between the composition of household batteries and lead-acid batteries; and
- a lack of information for comparing the risks of the various options available for disposal of household batteries.

hile the public tends to support the option of recycling these products, the separate collection dry-cell batteries may contribute to various hazardous situations including fire, explosion d human exposures to chemicals causing skin irritation. In addition, separate collection of /-cell batteries for the purpose of recycling is not seen to be a viable option currently due to difficulty of recycling including, the large energy requirements, the poor quality of overable materials and the lack of markets for recovered materials. The recycling process y also result in the production of new and different kinds of waste streams of potential ncern.

e CBMA, convinced that a comprehensive review conducted by an independent research group uld help the public to better understand the potential risks of the various household battery posal options, funded the Institute for Risk Research (IRR) to conduct this study.

1.2 Study Objectives and Scope

The overall purpose of this study is to investigate the environmental impacts of household batteries. The study considers only AAA, AA, C, D, and 9 volt formats and which belong to the alkaline (manganese), zinc-carbon/zinc chloride and nickel-cadmium family of batteries. The main objectives of the study conducted by the IRR are:

- To review and synthesize available literature on the issue of disposal of household batteries and the environment.
- To evaluate feasible disposal alternatives for household batteries, including landfilling, incineration and recycling.
- To investigate the impacts of present and future formulations of dry-cell batteries on the environment in relation to the various disposal options.

Recommendations are also made on preferred disposal methods that minimizes overall environmental and health hazards.

1.3 Method of Approach

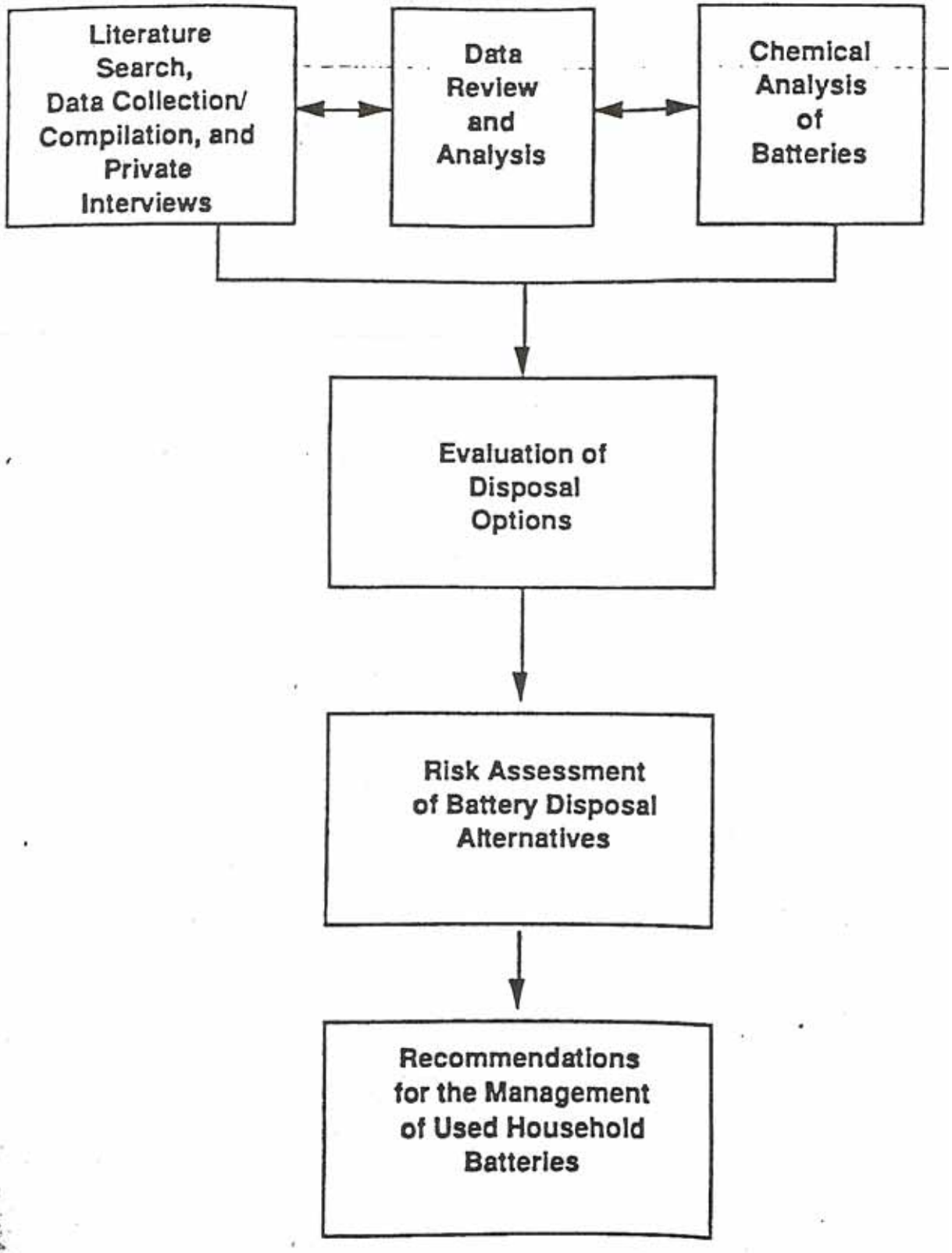
The study process is shown in Figure 1. Several types of information were reviewed and analyzed, including the following:

- Information on the composition of household batteries including chemical analyses performed on samples of batteries.
- Average domestic population consumption of household batteries.
- Contribution of the various metals of concern to municipal solid waste (MSW) streams from household batteries.
- Research studies pertaining to health and environmental impacts of the management of used household batteries.

Data and relevant information were obtained through extensive library and database searches and personal communication with specific experts. Additional general information on proprietary organic substitutes for mercury was further provided by the CBMA, to enable a complete assessment of potential impacts due to the reduction in mercury.

The data review and analysis comprised of a review of the literature pertaining to environmental impacts of the various household battery disposal options. The literature review focuses on North American studies; however where data gaps exist, world-wide studies are used noting any limitations in applying the results to the Canadian situation.

Figure 1
Steps in the Evaluation of the Impacts of Used Household Batteries on the Environment.



The IRR has based the assessment on the known chemical constituents of the household batteries under review. Pertinent properties of the chemical composition of household batteries, including the toxicity, fate and transport, persistence and attenuation effects have been determined. Information obtained on pertinent properties of the proprietary organic surfactants used as substitute for Hg in the household batteries are extrapolated in order to complete the environmental impact and risk assessment. Environmental impacts and potential risks posed by household batteries are evaluated and compared for the various disposal options.

In general, policy decisions on battery disposal practices depends on answers to several questions such as:

- to what degree do batteries contribute to the amount of heavy metals in the environment?
- what is the risk associated with that contribution?
- what level of risk will necessitate incurring the costs associated with changes in battery design and/or disposal?

Policies on used household battery management are still in the evolutionary stage, and answers to these and related questions will help shape policy regulations. Risk assessment constitutes one of the most important steps in the study, that will help answer these questions.

Risk assessment is a process that seeks to estimate the likelihood of occurrence of adverse effects due to specified exposures to chemical, physical, and/or biological agents in humans and ecological receptors within an ecosystem. Risk assessment may be performed in response to either short-term (acute) exposures to toxic chemicals, long-term (chronic) exposures, or to combinations of these. It may generally require some level of efforts in mathematical modeling, especially with respect to exposure estimations. Several uncertainties may surround the risk assessment process, especially because of possible incomplete exposure assessments, limited and questionable monitoring information, limitations on dose-response assessments, and/or the unavailability of complete toxicological profiles on some chemicals of potential concern identified for the risk assessment.

The purpose of a health and environmental risk assessment is to characterize the magnitude and extent of human or environmental exposure to selected pollutants, and then to quantify or qualify the potential adverse effects due to such exposures. The assessment can be used both to provide a baseline estimate of existing risks attributable to an environmental pollutant and to determine the potential reduction in exposure and risk for various corrective actions. Potential risks are estimated by considering the probability of occurrence, the potential effects, and the exposure scenarios. Important factors to consider in assessing risks include the following:

- Degree to which human health, safety, and welfare may be affected by exposure to contaminants
- Effects of contamination on the environment
- Current and future beneficial uses of impacted resources
- Application of appropriate, applicable or relevant regulatory standards and other requirements or guidelines.

Risk is defined a priori by standards set by regulatory agencies based on several extensive studies. For chemical exposures, the acceptable level of risk and/or applicable standards are based on several factors, including:

- Epidemiologic studies (involving accidental human exposures);
- Laboratory animal studies and subsequent inter-species extrapolations;
- Identification of sensitive human receptors and intra-species extrapolations;
- Application of safety factors to ensure adequate protection of potential receptors.

If the risk assessment results indicates a level of risk which falls within the acceptable standards, then no problems are anticipated from exposure of potential receptors. On the other hand, a level of risks greater than the acceptable limit may or may not pose actual risks. A more detailed evaluation will generally be needed to determine whether or not such risks are of concern under the case-specific exposure scenarios. It must be recognized that risk control measures have their own risks, and that reduction of one risk may lead to other risks. Thus, even if the initial risk assessment indicates an unacceptable risk, a comprehensive evaluation should help determine the cost-effectiveness and trade offs in any risk control programs to be implemented.

2.0 DRY-CELL BATTERY USAGE IN CANADA

A battery is a device for storing energy and then releasing it in the form of electric current as and when needed. Household battery types commonly found on the Canadian market include the following systems:

- Zinc-Carbon/Zinc-Chloride
- Alkaline (Manganese)
- Mercury Oxide
- Silver Oxide
- Zinc-Air
- Lithium
- Nickel-Cadmium Secondary Cells (Ni-Cads)

Only the alkaline manganese, zinc-carbon/zinc-chloride, and nickel-cadmium systems are considered in this study. However, it is important to note the relative advantages of having different battery systems on the market.

Why are there Different Battery Systems?

The various energy systems offered by the different battery systems are available to provide appropriate performance levels in terms of the usage of the cells in popular consumer appliances. There are different battery systems because in some applications, cell weights are the most important consideration whereas in others, cell volume is more critical.

Cell weight can be an important consideration in certain specialized applications. Different systems can be compared in watt-hours per unit weight of cell, referred to as gravimetric energy density (or energy density in terms of weight). For instance, the gravimetric energy density for selected dry-cell batteries in increasing order is as follows:

- Zinc-carbon
- Alkaline Manganese
- Zinc-Chloride
- Mercury
- Silver
- Zinc-Air
- Lithium

Where battery size is an important factor, watt-hours per unit volume (called Volumatric energy density) becomes a major consideration. The volumetric energy density, in increasing order for selected batteries is:

- Zinc-Carbon
- Zinc-Chloride
- Alkaline Manganese
- Mercury
- Zinc-Air
- Lithium
- Silver

From the standpoint of most consumer applications currently anticipated, volumetric efficiency is a most significant factor because the amount of space available for the cell in the equipment is usually fixed. For cells of the same system, the volumetric energy density may vary with the size of the cell. As cells are made smaller, the relative amount of active materials decreases in relation to the structural components needed to make the cell function. It is therefore logical to expect the volumetric energy density to diminish accordingly, as is the case for silver watch cells of varying sizes. However, this rule does not necessarily hold in all cases; in fact, the volumetric energy density of alkaline manganese cells increases with decreasing size because the smaller cells are made more compact and efficient in manufacture.

2.1 The Household Battery Types of Concern

Several sizes, types and brands of batteries are available on the Canadian market. The most common dry-cell batteries that are the focus of this study are the AAA, AA, C, D, and 9 volt varieties belonging to the alkaline, zinc-carbon/zinc-chloride and nickel-cadmium families. Batteries can be classified as primary or secondary cells according to their general electrochemical characteristics; secondary cells are rechargeable whereas the primary cells are not. Table 1 gives a classification scheme for the selected common battery types found on the Canadian market.

Table 1.
Characteristics of Common Household Batteries

Type of Battery	Size of Battery	Constituents of Battery: Anode	Cathode	Electrolyte	Mercury content (by weight)
Primary (Not Rechargeable)					
1. Alkaline*	D, C, AA, AAA, 9 Volt, Button	Zinc	Manganese dioxide	Alkaline solutions (Potassium hydroxide)	.1% or less and decreasing
2. Zinc-Carbon/ Zinc Chloride*	D, C, AA, AAA, 9 Volt	Zinc	Manganese dioxide	Ammonium chloride and/or Zinc chloride	.01% or less and decreasing
3. Mercury	D, C, AA, AAA, Button, some cylindrical	Zinc	Mercuric oxide	Alkaline solution (Potassium or Sodium hydroxide)	30% average
4. Lithium	C, AA, Coin and Button, 9 Volt	Lithium	Various metal oxides	Organic solvent or salt solution	Nil
5. Silver	Button	Zinc	Silver oxide	Alkaline solution	1.0%
6. Zinc-Air	Button	Zinc	Oxygen	Alkaline solution	1.0%
Secondary (Rechargeable)					
7. Nickel-Cadmium*	D, C, AA, AAA, 9 Volt	Cadmium	Nickel oxide	Alkaline solution	Nil

* Battery types investigated in this study.

Sources: Various Including :
Environment Canada, 1991: MPCA, 1991; CMU, 1989: Linden , 1984; Duracell, 1981.

Table 2
Performance Characteristics and Uses for Common Household Batteries

Type of Battery	Performance Characteristics: Shelf-Life	Power	Cost	Other	Examples of Household Uses
1. Alkaline	excellent extended	- high performance - high current demand - long term supply - 1.5 volts	more expensive than carbon	- better safety than carbon	Casettes, radios, electronic photoflash, cameras, toys, tape recorders & players, calculators, smoke alarms, shavers, TV sets, signal lights, electronic games
2. Carbon	moderate	- least amount/unit - light, intermittent uses - low power output - not suitable for sudden & sustained output - 1.5 volts	least expensive	- tendency to leak	Radios, flashlights, lanterns, small toys, tape recorders & players, communications equipment
3. Mercury	excellent	- high performance - high discharge capacity over long period - instantaneous and high intensive power - 1.3 volts		- tends to resist extreme temperatures - able to support sharp demands - miniature sizes	Watches, hearing aids, pacemakers, calculators, cameras, electronic thermostats, miniature radios, wireless mics, paging devices, transmitters, electronic games
4. Nickel-Cadmium	short shelf-life (when charged)	- are rechargeable but do not retain charge well - limited amount of stored energy - offer uniform power during use	very expensive	- long life-span - not recommended for emergency power	Portable tools, photoflash, calculators, tape recorders, dictation equipment, toys, measuring equipment, communication equipment
5. Lithium	- superior to mercury - unlimited	- long user life - supplies about double the voltage - high power replacement - good for low current	very expensive	- light weight - high potential - miniature sizes	Quartz watches, analog or LCD watches, calculators, cameras, semi-conductor memory, sensor circuits
6. Silver Oxide	excellent	- same as mercury - 1.6 volts - continuous source of power	more expensive than mercury	- miniature sizes	same as mercury
7. Zinc-Air	excellent	- greater capacity than mercury - quickly discharged (even when not in use) - immediate and continuous use - up to 3 volts		- miniature sizes - require frequent replacement - light weight	Watches, calculators, pocket games, hearing aids, pagers, military applications, power printed circuits

Alkaline Batteries:

Since their introduction into the market at the end of the 1950s, the market share of manganese dioxide (alkaline) batteries (also called long-lasting batteries) has continued to grow rapidly. The alkaline manganese battery has now become a familiar high power alternative to the zinc-carbon battery. Alkaline batteries were developed by replacing the active component of mercury oxide batteries (mercury oxide) by manganese dioxide (Environment Canada, 1991). Alkaline batteries have enjoyed great success during the past ten years, with the introduction of high-energy consumption devices, such as tape recorders, portable lights, photographic cameras, motorized toys, etc.

Alkaline (manganese) batteries are high performance batteries, and are much more expensive than zinc-carbon and zinc-chloride batteries (Table 2). A single alkaline battery can replace 3.5 ordinary batteries, or 2.5 high-performance batteries in the zinc-carbon/zinc-chloride family (Environment Canada, 1991). These high-performance batteries are used for high current demand applications or for devices that use current over long periods of time. Alkaline batteries operate on a completely different principle than that used by zinc-carbon batteries, and they have superior electrical qualities (Duracell, 1981; Environmental Canada, 1991). They are also resistant to high temperatures, and offer better safety where leaking is concerned.

The chemical reaction in alkaline batteries takes place in an alkaline medium. The anode of alkaline batteries consists of zinc powder, and has a surface about 100 times more efficient than that of zinc-carbon/zinc-chloride batteries. However, because of the particular shape of the anode, alkaline batteries have tended to require a much larger quantity of mercury than zinc-carbon batteries (Environmental Canada, 1991).

Zinc-Carbon/Zinc-Chloride Batteries:

Zinc-carbon batteries, also called Leclanché batteries, or ordinary batteries, are the best known, least expensive, and least powerful batteries. They are often used for low power output devices, such as flashlights and radios (Table 2) and are not suitable for sudden or sustained output demands. Zinc-carbon batteries had a tendency to leak when left in the host device after they are discharged (Environment Canada, 1991). The category also includes "high-performance" zinc-carbon or zinc-chloride batteries; these are more expensive, and better able to withstand temperature variations than regular zinc-carbon batteries. High performance batteries last about 40% longer than regular zinc-carbon batteries, they have improved discharge characteristics, as

well as a higher resistance to leaking, but they are not very suitable for applications that require high electricity outputs (Environment Canada, 1991).

The anode of zinc-carbon batteries is zinc, and the cathode contains manganese dioxide (Table 1). The electrolytes consist of an aqueous solution of zinc chloride for "high performance" batteries, and ammonium chloride for ordinary batteries. During discharge, the zinc anode is oxidized, and the manganese dioxide is reduced (Environment Canada, 1991). Small quantities of several inhibiting agents, such as mercury chloride, can be added to the electrolyte. On average, 0.01% of the total weight of zinc-carbon and zinc-chloride batteries consists of mercury.

Nickel-Cadmium Cells:

The manufacture and use of nickel-cadmium secondary cells began during the late 1950s. They are used in various industrial and communications applications. There are two types of nickel-cadmium secondary cells: gas-tight (or sealed), and open. With the exception of the electrolyte, these secondary cells are not significantly different from each other. Large nickel-cadmium cells of the open kind are rectangular in shape, and similar to lead cells, but offer better performance; they often are used for industrial applications. Small gas-tight or sealed secondary cells, commonly known as rechargeable Ni-Cad batteries, are made in the button and cylindrical formats and can theoretically be recharged more than 1,000 times and can be used for several years before disposal (USEPA, 1988; Environment Canada, 1991). Rechargeable batteries, or gas-tight nickel-cadmium secondary cells used for domestic applications offer a more economic alternative.

The composition of the electrodes in the gas-tight cells eliminates excess pressures from the emission of oxygen during overload conditions (Environment Canada, 1991). Secondary cells have a life span of about five years, and are used in hand tools, radios, tape recorders, flashlights, calculators, medical devices, measuring instruments, communications equipment, etc. Rechargeable nickel-cadmium batteries have a lower energy density than primary cells; the capacity of rechargeable Ni-Cad batteries is equivalent to about 20% that of alkaline batteries (Environment Canada, 1991). Rechargeable Ni-Cad batteries are not recommended for emergency power supply systems such as, smoke detectors, because they discharge quickly, and could be drained and ineffective at the time of an emergency. On the other hand, Ni-Cad batteries have the advantage of offering more uniform power during use.

As their name indicates, the active elements in Ni-Cad secondary cells are nickel - which forms the cathode and cadmium - which constitutes the anode (Table 1). The cadmium content is generally lower than that of nickel, and represents up to about 18% of the total weight; nickel is about 20%.

When the secondary cell is charged, the nickel hydroxide is oxidized, and the cadmium hydroxide is reduced to its metallic form.

2.2 Dry-Cell Battery Composition

Used household batteries disposed of by individual households are exempt from provincial and federal hazardous waste regulations. On the other hand, there are some toxic metals which form important components of dry-cell batteries. For all the different types, household batteries contain specific metals that may potentially impact the environment. The specific metals of concern for the dry-cells identified for this study include the following:

- Cadmium
- Manganese
- Mercury
- Nickel
- Zinc

These metals are toxic to humans and ecological receptors under certain exposure conditions or scenarios, when released into the environment. Of all the heavy metals used in the manufacture of primary and secondary cells, cadmium and mercury are of the most concern. Alkaline and Ni-Cad batteries account respectively, for a reasonable proportion of the mercury and cadmium used in dry-cell batteries.

The percentage composition of metals in the dry-cell batteries are shown in Table 3. All the types of primary cells under investigation here contain mercury, which is mainly used as a passive inhibiting element. It is present in varying quantities, depending on the type, size, and/or format of the system. Mercury is used as an inhibitor, and is added to the zinc anode (in most primary cells) to prevent corrosion mechanisms from affecting the chemical reaction through the generation of gas. The presence of mercury also prevents the batteries from self-discharging; its presence, therefore, extends the shelf life of the product considerably. Various other compounds are included with the electrodes and/or the electrolyte in each cell type, in order to increase its performance by controlling undesirable reactions, and to facilitate desirable reactions inside the battery (Environment Canada, 1991).

Table 3
Approximate Average Weight and Percent (of Total Battery Weight) of Metals in the Various Household Battery Types and Sizes

Battery			Average Metal Composition by Weight									
Type	Size	Avg. Wt. (g)	Cadmium		Manganese (as MnO2)		Mercury		Nickel		Zinc	
			%	grams	%	grams	%	grams	%	grams	%	grams
Alkaline (Manganese)												
1.5V	AAA	11 (10-11)	0.4	0.04	30 (28-32)	3.30	0.025	0.00	–		12 (10-15)	1.32
1.5V	AA	23 (23-30)	0.4	0.09	34 (32-35)	7.82	0.025	0.01	–		14 (12-15)	3.22
1.5V	C	66 (65-70)	0.4	0.26	32 (30-35)	21.12	0.025	0.02	–		14 (12-15)	9.24
1.5V	D	133 (130-140)	0.4	0.53	32 (30-35)	42.56	0.025	0.03	–		16 (14-18)	21.28
9V	9 Volt	46 (45-50)	0.4	0.18	30 (28-32)	13.80	0.025	0.01	–		10 (8-12)	4.60
Zinc-Carbon/Zinc-Chloride												
1.5V	AAA	8 (7-8)	0.01	0.00	38 (35-40)	3.04	0.01	0.00	–		38 (35-40)	3.04
1.5V	AA	17 (16-19)	0.01	0.00	28 (25-30)	4.76	0.01	0.00	–		22 (20-25)	3.74
1.5V	C	44 (44-50)	0.01	0.00	28 (25-30)	12.32	0.01	0.00	–		18 (15-20)	7.92
1.5V	D	87 (87-100)	0.01	0.01	28 (25-30)	24.36	0.01	0.01	–		18 (15-20)	15.66
9V	9 Volt	37 (37-50)	0.01	0.00	22 (20-25)	8.14	0.01	0.00	–		14 (12-15)	5.18
Nickel-Cadmium (Ni-Cad)												
1.5V	AAA	11 (10-12)	13 (10-18)	1.43	–		–		20 (15-25)	2.20	–	
1.5V	AA	23 (20-25)	13 (10-18)	2.99	–		–		20 (15-25)	4.60	–	
1.5V	C	52 (50-55)	13 (10-18)	6.76	–		–		20 (15-25)	10.40	–	
1.5V	D	62 (60-65)	13 (10-18)	8.06	–		–		20 (15-25)	12.40	–	
9V	9 Volt	38 (35-40)	13 (10-18)	4.94	–		–		20 (15-25)	7.60	–	

Note: Insignificant amounts of metals are noted as 0.00 gram for average metal composition by weight for the difference cells.

( ) = Range of approximate weights or %

Sources--Various, including:
Environment Canada, 1991
CMU, 1989
Duracell Inc., Canada (Information Sheet August, 1991)
CBMA (Personal Communication, 1991)

2.2.1 The Changing Composition of Dry-Cell Batteries

One major contributor of mercury from household batteries to MSW is from alkaline cells. Thus, reducing the amount of mercury in alkaline batteries could have significant impact on the total amount of Hg that may enter MSW due to used dry-cell battery disposal. Due to its high toxicity and its relatively high cost, manufacturers have been encouraged to attempt to reduce the amount of Hg contained in dry-cell batteries. To achieve this, a number of patented organic substitutes are being used in several makes of household batteries.

During the past decade, the total amount of Hg used in dry-cell batteries manufactured in Canada has decreased by about 95%, and this continues to decrease. It is believed that, the present concentration of mercury in alkaline batteries manufactured in North America is at most 0.8%, and less than 0.05% in the case of zinc-carbon/zinc-chloride batteries (Environment Canada, 1991). In fact, in Canada, most zinc-chloride batteries contain either no Hg or less than than 0.01% of total cell weight when present. Most of the reduction in Hg usage has occurred in alkaline batteries. Current Hg content in alkaline batteries manufactured in Canada does not exceed 0.025% by weight in most cases. The trend in Hg reduction efforts is expected to continue in pursuit of near-zero or even zero Hg alkaline batteries. On the other hand, Cd is used as the negative electrode in rechargeable Ni-Cad batteries. Since Cd is an electrode material in this case, its use cannot be reduced without proportionally reducing the energy content of the batteries (Balfour, 1990). Depending on its changing market share, Ni-Cads are therefore expected to contribute similar amounts of Cd to the MSW streams as presently exist.

2.2.2 Material Used as Substitute for Mercury

Some organic surfactants are used by the different manufacturers as a substitute for Hg in the anode production process, to help minimize the amount of Hg used in the production of alkaline batteries. These organic substitutes are used at a level of approximately 100 ppm. A typical material safety data sheet (MSDS) for the proprietary complex organic phosphate ester has been reviewed and determined to present far less toxicity effects than Hg; in addition it is not volatile and is generally a more stable compound. For instance, the organic substitute has an LD_{50} of about 5 to 15 times lower than that reported for Hg (LD_{50} is the lethal dose at which 50% of all laboratory test animals will be killed). However, the surfactant is known to be soluble in water and also to be corrosive. This could therefore possibly facilitate the degradation of batteries under landfill conditions.

2.3 Household Battery Consumption in Canada

There has been a continuous rapid increase in the number of battery-operated devices used in Canada. The driving force behind consumer demand for batteries may therefore be attributed to the battery-operated appliance market. The common battery-powered devices in use in Canada include the following:

- audio equipment (eg: radios, tape recorders)
- hearing aids
- toys and games
- lighting products (eg: flashlights, lanterns, switches)
- photographic equipment (eg: cameras, flashes)
- smoke detectors
- calculators
- clocks and watches
- remote control devices (eg: for TV and VCR sets)

Alkaline and zinc-carbon/zinc-chloride batteries hold the largest share of the domestic market in Canada--over 90% (Environment Canada, 1991; CBMA, 1991). Alkaline batteries have over 65% of the household battery market share, with zinc-carbon/zinc-chloride about 29%. Ni-Cads command the smallest share of the dry-cell battery market--about 5%. However, the rechargeable Ni-Cad battery, which has a useful life of about five years and which are discarded like all other batteries after this useful life, has a growing market.

The most popular battery size is AA. Table 4 gives an indication of the Canadian battery market estimate for household dry-cell batteries. These estimates are based on industry sales primarily through retail outlets. This may exclude relatively minor sales of batteries to medical, military, industrial and commercial users and also of batteries included in consumer devices imported/sold with batteries. A breakdown of the per capita household battery use by province is given in Table 5.

Table 4
Estimates of Canadian Household Battery Sales (1990/1991)

Battery Type	Battery Size							
	Total in Millions	AAA	AA	C	D	9 Volt	Other	% of Totals
Alkaline	100	9	60	9	8	9	5	65.4
Zinc Carbon	25	1	21	7	9	6	1	16.3
Zinc Chloride	20							13.1
Nickel-Cadmium	8	0.5	5	1	1	0.5	--	5.2
Totals (% of Total)	153 (100)	10.5 (6.9)	86 (56.2)	17 (11.1)	18 (11.8)	15.5 (10.1)	6 (3.9)	

Notes: Geographic sales would align approximately as follows:

Maritimes	6%
Quebec	23%
Ontario	36%
Manitoba/Saskatchewan	8%
Alberta	11%
British Columbia	16%

--Source of information: CBMA (Contact person M.J. Smith, Duracell Inc. Canada ; personal communication June , 1991)

Table 5
Household Battery Usage Data for Canada by Provinces 1990-91

Battery Type	Estimated National Battery Sales for 1990 for the Various Battery Sizes (Number of batteries in millions)						Provincial Data and Information (Approximate Battery Usage Figures)														
	AAA	AA	C	D	9V	Total	ALBERTA			B.C.			MANITOBA/SASK.			MARITIMES			ONTARIO			QUEBEC		
							Battery Usage (millions)	Population (millions)	Per Capita Use	Battery Usage (millions)	Population (millions)	Per Capita Use	Battery Usage (millions)	Population (millions)	Per Capita Use	Battery Usage (millions)	Population (millions)	Per Capita Use	Battery Usage (millions)	Population (millions)	Per Capita Use	Battery Usage (millions)	Population (millions)	Per Capita Use
Alkaline	8	60	9	8	9	85	10.46	2.40	4.35	18.20	2.98	6.09	7.60	2.10	3.63	5.70	2.30	2.48	94.20	8.49	8.18	21.88	6.64	3.29
Zinc-Carbon/ Zinc-Chloride	1	21	7	9	6	44	4.84	2.40	2.02	7.04	2.98	2.36	3.62	2.10	1.68	2.64	2.30	1.16	18.84	8.49	1.89	10.12	6.64	1.62
Nickel-Cadmium	0.6	6	1	1	0.8	8	0.88	2.40	0.37	1.28	2.98	0.43	0.64	2.10	0.31	0.48	2.30	0.21	2.88	8.49	0.31	1.84	6.64	0.28

Notes: Populations are based on 1990 estimated population figures from Statistics Canada.

2.4 Potential Metal Contributions from Used Household Batteries to Municipal Solid Waste (MSW) Streams

Dry-cell batteries contain heavy metals including cadmium, mercury and nickel and also an organic substitute for mercury that has been introduced to help the manufacturers reduce the mercury content of the household batteries without compromising on the effectiveness or performance of these batteries. Household batteries may therefore contribute to the hazardous waste stream appearing in municipal wastes. In fact, mercury and cadmium from discarded household batteries may be contributing up to 20% and 33 % respectively, of the respective total metal compositions appearing in municipal solid waste (MSW) streams in much of North America. Cadmium is used primarily in nickel-cadmium batteries, which make up the largest portion of the dry-cell secondary battery market.

The type and quantity of metals in batteries may vary significantly for the different systems and/or sizes/formats. Table 6 shows the potential metal contributions from the various household battery types to the MSW. The calculations are made by multiplying the average percent of metals found in each battery type by the average weight of the battery and also by the number of the particular battery types sold to consumers. An overly conservative assumption has been made that all battery sales each year equals the amount of discards, a situation that is definitely not applicable to the rechargeable nickel-cadmium batteries. However, assuming batteries used from previous years are added to the waste stream, then the assumption of sales equals discards is fair, though still conservative.

Using a similar procudeure to calculate the total amount of mercury entering the Canadian market, during 1991 the amounts for the various types of batteries were as follows: alkaline cells contributed some 870 kg of mercury, zinc-air 51.44 kg, silver oxides 9.92 kg, alkaline buttons 6.84 kg (Fig 2). No mercury was contributed from the disposal of lithium, zinc chloride/zinc carbon or NiCads. Some 3,384.94 kg or over 78% of all the mercury consumed was from mercuric oxide batteries (Fig. 3).

The trend in mercury consumption in batteries such as alkaline, zinc chloride/zinc carbon types has been steadily decreasing in Canada over the past five years (Fig. 4). Combining sales figures and known Hg compositions, the amount of mercury was just over 25,000 kg is 1986 and by 1991 the amount had been reduces to about 1000 kg.

Table 6

Estimates of Metals Contributions to MSW Streams from Household Batteries

Battery Type	Battery Size	Average Weight (grams)	Per Cent Cd Content Weight Cd per Battery (grams)		Per Cent Mn Content Weight Mn per Battery (grams)		Per Cent Hg Content Weight Hg per Battery (grams)		Per Cent Ni Content Weight Ni per Battery (grams)		Per Cent Zn Content Weight Zn per Battery (grams)	
Alkaline	AAA	11	0.4	0.04	30	3.30	0.025	0.00	0	0.00	12	1.32
	AA	23	0.4	0.09	34	7.82	0.025	0.01	0	0.00	14	3.22
	C	66	0.4	0.26	32	21.12	0.025	0.02	0	0.00	14	9.24
	D	133	0.4	0.53	32	42.56	0.025	0.03	0	0.00	16	21.28
	9V	46	0.4	0.18	30	13.80	0.025	0.01	0	0.00	10	4.60
Zinc-Carbon/ Zinc-Chloride	AAA	8	0.01	0.00	38	3.04	0.01	0.00	0	0.00	38	3.04
	AA	17	0.01	0.00	28	4.76	0.01	0.00	0	0.00	22	3.74
	C	44	0.01	0.00	28	12.32	0.01	0.00	0	0.00	18	7.92
	D	87	0.01	0.01	28	24.36	0.01	0.01	0	0.00	18	15.66
	9V	37	0.01	0.00	22	8.14	0.01	0.00	0	0.00	14	5.18
Nickel-Cadmium	AAA	11	13	1.43	0	0.00	0	0.00	20	2.20	0	0.00
	AA	23	13	2.99	0	0.00	0	0.00	20	4.60	0	0.00
	C	52	13	6.76	0	0.00	0	0.00	20	10.40	0	0.00
	D	62	13	8.06	0	0.00	0	0.00	20	12.40	0	0.00
	9V	38	13	4.94	0	0.00	0	0.00	20	7.60	0	0.00
TOTAL METAL (million grams or tonnes):				47.33		1610.88		1.08		50.70		808.92

Notes:
Estimates of metals entering MSW streams from household batteries are based on annual sales figures, percent metal compositions, and average weights for the various battery types.

Table 6 cont'd

Total Battery Sales (millions)	Battery Type	Battery Size	Provincial Data and Information (Approximate Battery Usage Figures)					
			ALBERTA					
			Battery Usage (millions)	Cd (million gm)	Mn (million gm)	Hg (million gm)	Ni (million gm)	Zn (million gm)
9	Alkaline	AAA	0.99	0.044	3.267	0.003	0.000	1.307
60		AA	6.6	0.607	51.612	0.038	0.000	21.252
9		C	0.99	0.261	20.909	0.016	0.000	9.148
8		D	0.88	0.468	37.453	0.029	0.000	18.726
9		9V	0.99	0.182	13.662	0.011	0.000	4.554
95.00			10.45	1.56	126.90	0.10	0.00	54.99
1	Zinc-Carbon/ Zinc-Chloride	AAA	0.110	0.000	0.334	0.000	0.000	0.334
21		AA	2.310	0.004	10.996	0.004	0.000	8.639
7		C	0.770	0.003	9.486	0.003	0.000	6.096
9		D	0.990	0.009	24.116	0.009	0.000	15.503
6		9V	0.660	0.002	5.372	0.002	0.000	3.419
44.00			4.84	0.02	50.31	0.02	0.00	33.99
0.5	Nickel-Cadmium	AAA	0.055	0.079	0.000	0.000	0.121	0.000
5		AA	0.55	1.645	0.000	0.000	2.530	0.000
1		C	0.11	0.744	0.000	0.000	1.144	0.000
1		D	0.11	0.887	0.000	0.000	1.364	0.000
0.5		9V	0.055	0.272	0.000	0.000	0.416	0.000
8.00			0.88	3.83	0.00	0.00	5.58	0.00
TOTAL METAL (million grams or tonnes):				5.21	177.21	0.12	5.58	88.98

Table 6 cont'd

Battery Type	Battery Size	Provincial Data and Information (Approximate Battery Usage Figures)					
		B.C.					
		Battery Usage (millions)	Cd (million gm)	Mn (million gm)	Hg (million gm)	Ni (million gm)	Zn (million gm)
Alkaline	AAA	1.44	0.063	4.752	0.004	0.000	1.901
	AA	9.6	0.863	75.072	0.055	0.000	30.912
	C	1.44	0.380	30.413	0.024	0.000	13.306
	D	1.28	0.681	54.477	0.043	0.000	27.238
	9V	1.44	0.265	19.872	0.017	0.000	6.624
		15.20	2.27	184.59	0.14	0.00	79.98
Zinc-Carbon/ Zinc-Chloride	AAA	0.16	0.000	0.486	0.000	0.000	0.486
	AA	3.36	0.006	15.994	0.006	0.000	12.566
	C	1.12	0.005	13.798	0.005	0.000	8.870
	D	1.44	0.013	35.078	0.013	0.000	22.550
	9V	0.96	0.004	7.814	0.004	0.000	4.973
		7.04	0.03	73.17	0.03	0.00	49.45
Nickel-Cadmium	AAA	0.08	0.114	0.000	0.000	0.178	0.000
	AA	0.8	2.392	0.000	0.000	3.680	0.000
	C	0.16	1.082	0.000	0.000	1.664	0.000
	D	0.16	1.290	0.000	0.000	1.984	0.000
	9V	0.08	0.395	0.000	0.000	0.608	0.000
		1.28	5.27	0.00	0.00	8.11	0.00
TOTAL METAL (million grams or tonnes):			7.57	257.76	0.17	8.11	129.43

Table 8 cont'd

Battery Type	Battery Size	Provincial Data and Information (Approximate Battery Usage Figures)					
		MANITOBA/SASKATCHEWAN					
		Battery Usage (millions)	Cd (million gm)	Mn (million gm)	Hg (million gm)	Ni (million gm)	Zn (million gm)
Alkaline	AAA	0.72	0.032	2.376	0.002	0.000	0.950
	AA	4.8	0.442	37.536	0.028	0.000	15.456
	C	0.72	0.190	15.206	0.012	0.000	6.653
	D	0.64	0.340	27.238	0.021	0.000	13.619
	9V	0.72	0.132	9.936	0.008	0.000	3.312
		7.60	1.14	92.29	0.07	0.00	39.99
Zinc-Carbon/ Zinc-Chloride	AAA	0.08	0.000	0.243	0.000	0.000	0.243
	AA	1.68	0.003	7.997	0.003	0.000	6.283
	C	0.56	0.002	6.899	0.002	0.000	4.435
	D	0.72	0.006	17.539	0.006	0.000	11.275
	9V	0.48	0.002	3.907	0.002	0.000	2.466
		3.52	0.01	36.59	0.01	0.00	24.72
Nickel-Cadmium	AAA	0.04	0.057	0.000	0.000	0.088	0.000
	AA	0.4	1.196	0.000	0.000	1.840	0.000
	C	0.08	0.541	0.000	0.000	0.832	0.000
	D	0.08	0.645	0.000	0.000	0.992	0.000
	9V	0.04	0.198	0.000	0.000	0.304	0.000
		0.64	2.64	0.00	0.00	4.06	0.00
TOTAL METAL (million grams or tonnes):			3.79	128.88	0.08	4.06	64.71

Table 6 cont'd

Battery Type	Battery Size	Provincial Data and Information (Approximate Battery Usage Figures)					
		MARITIMES					
		Battery Usage (millions)	Cd (million gm)	Mn (million gm)	Hg (million gm)	Ni (million gm)	Zn (million gm)
Alkaline	AAA	0.54	0.024	1.782	0.001	0.000	0.713
	AA	3.6	0.331	28.152	0.021	0.000	11.592
	C	0.54	0.143	11.405	0.009	0.000	4.990
	D	0.48	0.255	20.429	0.016	0.000	10.214
	9V	0.54	0.099	7.452	0.006	0.000	2.484
		5.70	0.85	69.22	0.05	0.00	29.99
Zinc-Carbon/ Zinc-Chloride	AAA	0.06	0.000	0.182	0.000	0.000	0.182
	AA	1.26	0.002	5.998	0.002	0.000	4.712
	C	0.42	0.002	5.174	0.002	0.000	3.328
	D	0.54	0.005	13.154	0.005	0.000	8.456
	9V	0.36	0.001	2.930	0.001	0.000	1.865
		2.64	0.01	27.44	0.01	0.00	18.54
Nickel-Cadmium	AAA	0.03	0.043	0.000	0.000	0.066	0.000
	AA	0.3	0.897	0.000	0.000	1.380	0.000
	C	0.06	0.406	0.000	0.000	0.624	0.000
	D	0.06	0.484	0.000	0.000	0.744	0.000
	9V	0.03	0.148	0.000	0.000	0.228	0.000
		0.48	1.98	0.00	0.00	3.04	0.00
TOTAL METAL (million grams or tonnes):			2.84	99.66	0.06	3.04	48.54

Table 6 cont'd

Battery Type	Battery Size	Provincial Data and Information (Approximate Battery Usage Figures)					
		ONTARIO					
		Battery Usage (millions)	Cd (million gm)	Mn (million gm)	Hg (million gm)	Ni (million gm)	Zn (million gm)
Alkaline	AAA	3.24	0.14256	10.692	0.00891	0	4.276
	AA	21.6	1.9872	166.912	0.1242	0	69.55
	C	3.24	0.85536	68.4288	0.05346	0	29.937
	D	2.88	1.53216	122.5728	0.09576	0	51.288
	9V	3.24	0.59616	44.712	0.03726	0	14.90
		34.20	5.11	415.32	0.32	0.00	179.96
Zinc-Carbon/ Zinc-Chloride	AAA	0.36	0.000288	1.0944	0.000288	0	1.094
	AA	7.56	0.012852	35.9856	0.012852	0	28.274
	C	2.52	0.011088	31.0464	0.011088	0	19.958
	D	3.24	0.028188	78.9264	0.028188	0	50.738
	9V	2.16	0.007992	17.5824	0.007992	0	11.188
		15.84	0.08	164.64	0.08	0.00	111.25
Nickel-Cadmium	AAA	0.18	0.2574	0	0	0.396	
	AA	1.8	5.382	0	0	8.28	
	C	0.36	2.4336	0	0	3.744	
	D	0.36	2.9016	0	0	4.464	
	9V	0.18	0.8892	0	0	1.368	
		2.88	11.86	0.00	0.00	18.25	0.00
TOTAL METAL (million grams or tonnes):			17.04	579.95	0.38	18.25	291.21

Table 8 cont'd

Battery Type	Battery Size	Provincial Data and Information (Approximate Battery Usage Figures)					
		QUEBEC					
		Battery Usage (millions)	Cd (million gm)	Mn (million gm)	Hg (million gm)	Ni (million gm)	Zn (million gm)
Alkaline	AAA	2.07	0.091	6.631	0.006	0.000	2.732
	AA	13.8	1.270	107.916	0.079	0.000	44.436
	C	2.07	0.546	43.718	0.034	0.000	18.127
	D	1.84	0.979	78.310	0.061	0.000	30.155
	9V	2.07	0.381	28.568	0.024	0.000	9.522
		21.85	3.27	265.34	0.20	0.00	114.97
Zinc-Carbon/ Zinc-Chloride	AAA	0.23	0.000	0.699	0.000	0.000	0.699
	AA	4.83	0.008	22.991	0.008	0.000	16.064
	C	1.61	0.007	19.835	0.007	0.000	12.751
	D	2.07	0.018	50.425	0.018	0.000	32.416
	9V	1.38	0.005	11.233	0.005	0.000	7.148
		10.12	0.04	105.18	0.04	0.00	71.08
Nickel-Cadmium	AAA	0.115	0.164	0.000	0.000	0.253	0.000
	AA	1.15	3.439	0.000	0.000	6.290	0.000
	C	0.23	1.555	0.000	0.000	2.392	0.000
	D	0.23	1.854	0.000	0.000	2.852	0.000
	9V	0.115	0.568	0.000	0.000	0.874	0.000
		1.84	7.58	0.00	0.00	11.66	0.00
TOTAL METAL (million grams or tonnes):			10.89	370.53	0.24	11.66	186.06

Figure 2

COMPARISON OF THE TOTAL WEIGHT OF MERCURY USED IN CANADA BY TYPE OF BATTERY IN 1991.

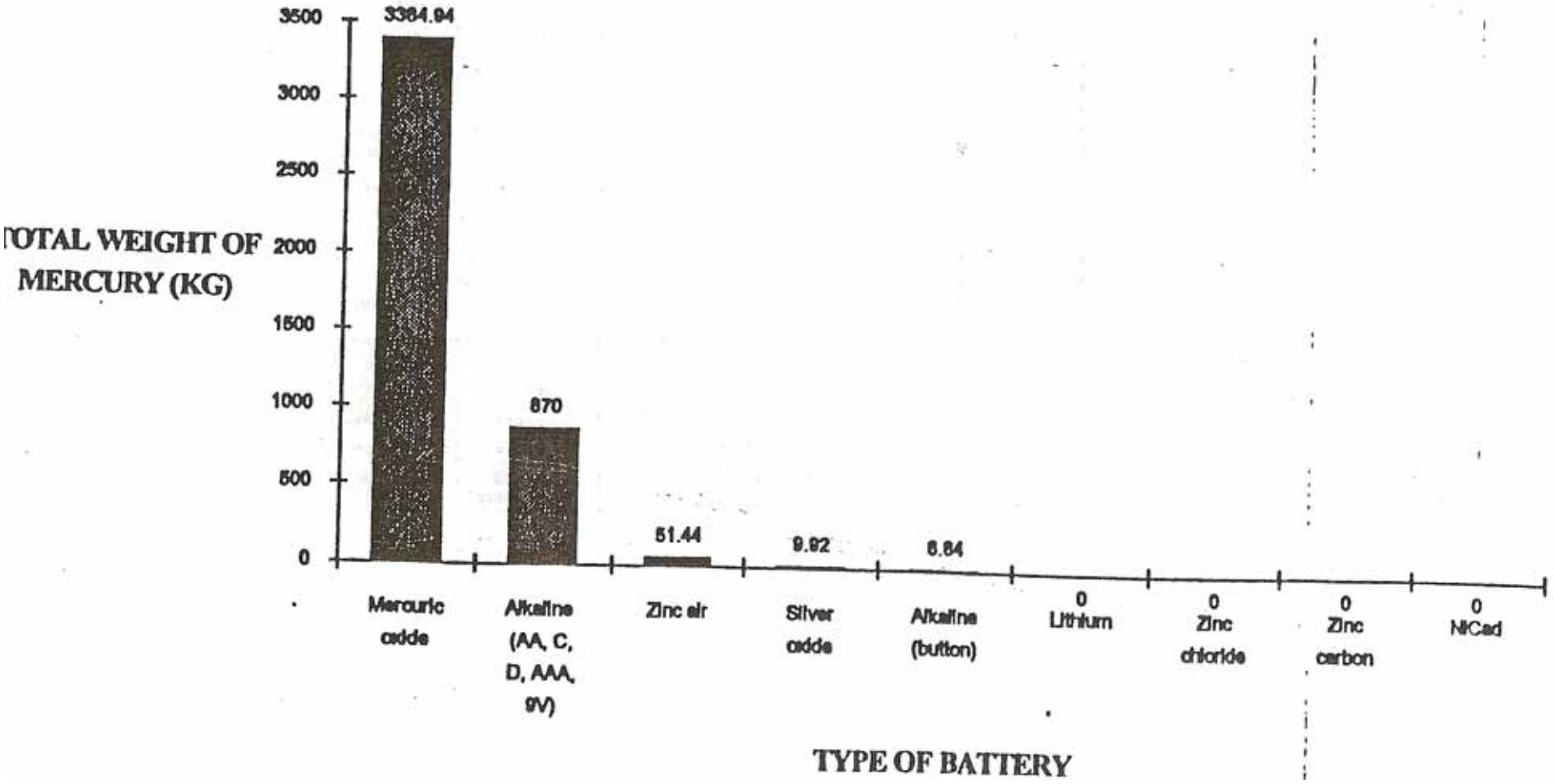


Figure 3

COMPARISON OF THE PERCENTAGE BY WEIGHT OF MERCURY USED IN CANADA BY TYPE OF BATTERY IN 1991.

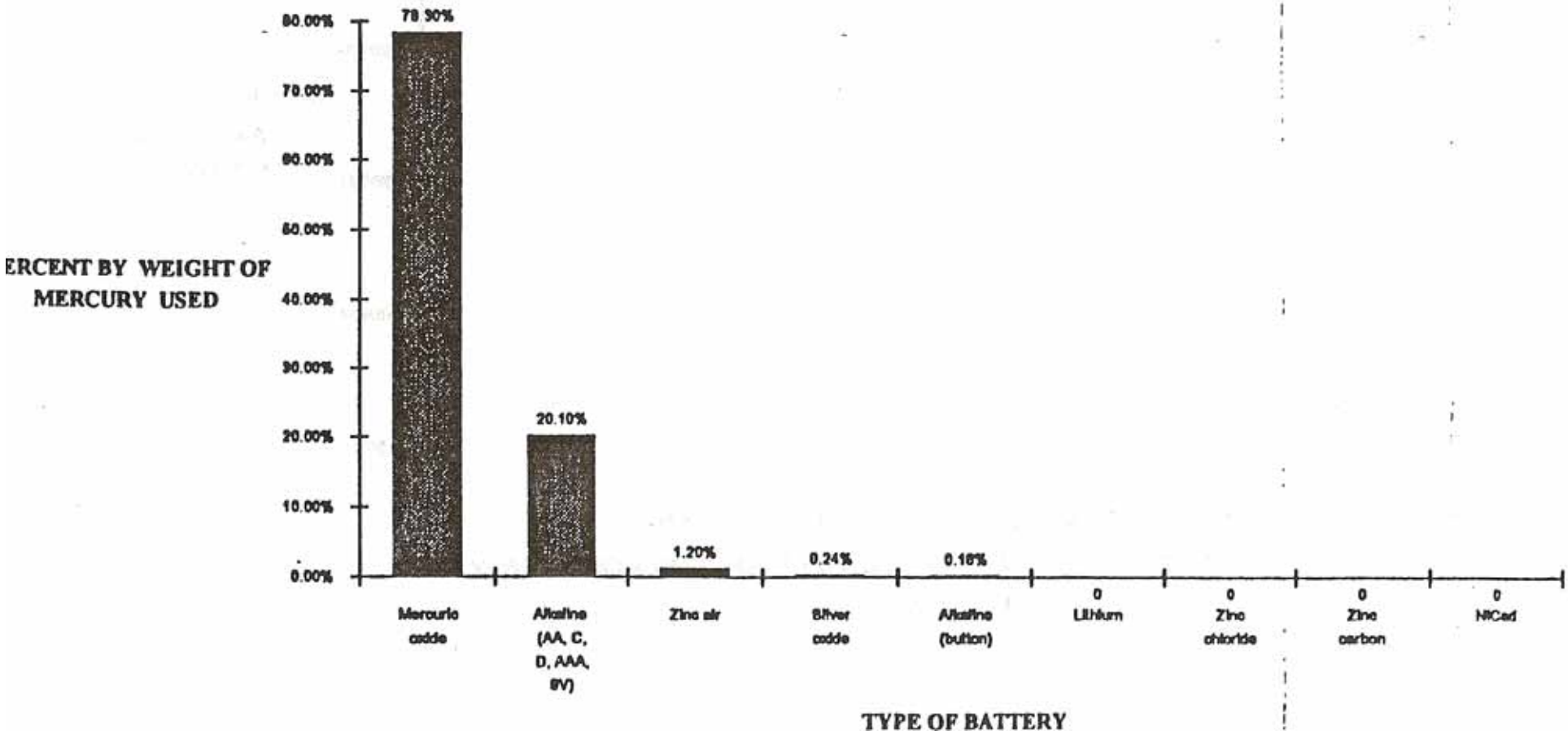
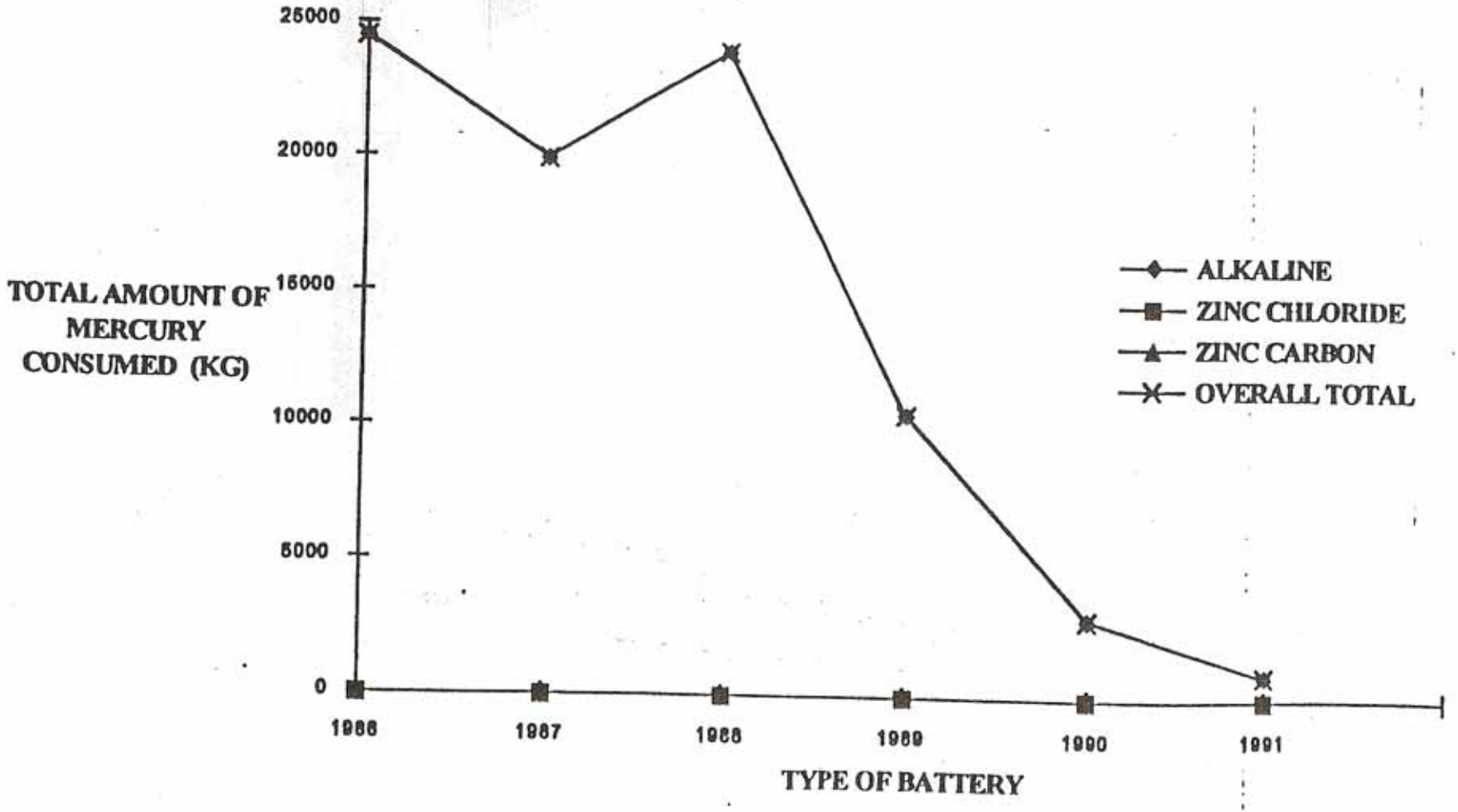


Figure 4

COMPARISON OF THE AMOUNT OF MERCURY CONSUMED IN CANADA BY TYPE OF BATTERY FROM 1986 TO 1991.



3.0 FEASIBLE DISPOSAL OPTIONS FOR USED HOUSEHOLD BATTERIES

Traditionally, landfilling has been the method of choice for disposal of municipal solid waste (MSW). With the growing stringent regulations and the lack of suitable landfill sites, disposal practices are undergoing several changes, especially with respect to the management of hazardous wastes. Incineration and recycling are becoming prominent alternatives to landfilling. These alternative waste management processes, however, are not without their own problems. Although incineration tends to reduce the volume of waste, the residual ash particularly the fly ash, which has to be disposed of is usually more toxic due to the concentration of toxic remains and byproducts. In addition there is the potential for the release of toxic contaminants into the air.

As much as recycling seems a more acceptable option, its application to the management of used household batteries is yet to be proven as efficient. In fact, the economic feasibility of recycling household batteries is a major issue -- apart from potential risks due to collection of large quantities of such batteries and the risks associated with various recycling operations. For instance, recent electrical tests carried out at the Carnegie-Mellon University (CMU, 1989) showed that a significant percentage of disposed batteries still have residual voltage. If these batteries are collected, they could short together and generate enough heat to cause a fire. Thus, care should be taken in any collection system to prevent this from happening. Sorting the batteries prior to disposal could be considered if groups of batteries cannot be separated from readily flammable materials. The residual power also makes batteries more susceptible to corrosion; a factor which should be considered while discussing the condition of batteries in landfills over long periods of time.

3.1 Landfilling with Municipal Solid Wastes as a Disposal Alternative

Recently, the general public has shown increased concern over ground and surface water contamination from landfill waste disposal (Greghian et al., 1981). The potential environmental impacts from landfilling of MSW are a major reason for many regulations enforced, the technical innovations, and the public opposition to siting of such facilities. The actual impact which a landfill has on its surrounding environment, however, is highly dependent on the practice and operation of the facility (Christensen et al., 1989). The contamination potential of a landfill depends on three factors (Miller and Mishra 1989):

- the migration of leachate out of the landfill;

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(MSW). With the growing stringent regul
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of such batteries and the risks associated w
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significant percentage of disposed batteri
collected, they could short together and gen

- the potential harm of the leachate constituents; and
- the corresponding concentrations of each contaminant in the leachate.

In addition, there should be completeness of exposure pathways due to the presence of potential human or ecological receptors that are potentially at risk. Metals in batteries will generally not be released rapidly from landfills, but, the overall load of metals into the soil and the specific soils ability to adsorb metals are critical factors to consider in deciding whether or not to landfill household batteries. The mobility of any of the metals of concern within the soil system will be controlled by the extent of fixation, adsorption, exclusion, complex formation, reaction kinetics, as well as the soils overall physical and chemical properties (MPCA, 1991). Indeed, physical and chemical properties of soils may control leachate migration. The combined physical and chemical properties of texture, particle size, hydrous oxides and organic matter, cation exchange capacity (CEC) and pH are the most significant soil properties that may determine leachate migration (MPCA, 1991). For example, the soil chemistry of cadmium (Cd) is, to a great extent controlled by pH. Under acidic conditions Cd solubility increases and very little sorption of Cd by soil colloids, hydrous oxides, and organic matter takes place. Thus, the nature of soils surrounding a landfill greatly affect the rate of heavy metal migration; for instance, clay material appears to greatly retard, if not stop, heavy metal migration, whereas sand or gravel soils are less adsorptive and also offer less resistance to water flow (Fochtman and Haas, 1975). Cd, Ni and Zn are believed to have low mobility in most clayey soils, but Hg may have moderate mobility in the same group of soils. Mercury (Hg) has become widely recognized as one of the most hazardous elements to human health. The potential for Hg contamination exists where disposal practices create conditions conducive for conversion of Hg to toxic forms, such as methyl-mercury and other organic mercury compounds. Hg moves very slowly through soils under field conditions.

When batteries are disposed of in a landfill they usually are buried together with MSW with variable moisture content. Capped landfills are expected to have a lower moisture content than uncapped landfills. Because of the compressibility of MSW, it is likely that the loads on battery cases are too small to damage them. Maintaining these conditions will limit the release of heavy metals from batteries. Batteries with broken cases present a different problem scenario. Additionally, batteries may be subjected to varying stresses between disposal in landfills, which may lead to rupture of the casing. A study of batteries in landfills has shown that some batteries are broken, corroded or otherwise damaged in a manner that would allow their contents to leach out (Little, 1989).

3.1.1 Degradation Processes of Batteries Under Domestic Waste Landfill Conditions

Batteries that are landfilled go through various physical changes after they are buried. Landfill conditions may promote corrosion and deterioration of battery cases, which consequently may release metals to leachate generated within the landfill. Kemper and Smith (1981) found that the amount of zinc leached from shredded refuse was an order of magnitude greater than from non-shredded refuse; they attributed the increase to the shredding of alkaline (and maybe Leclanché) batteries which increased the surface area and porosity of their zinc electrodes. However, Kineman and Natini (1988) reported previously that landfill waste buried for five to ten years was found to contain some damaged batteries; USEPA (1988) summarizes the results as follows:

- Many dry-cell batteries were identifiable after up to 10 years in sanitary landfills.
- The batteries were in different states of decomposition, ranging from ones looking like new, to partially corroded, to others almost totally degraded to the point that they were difficult to identify.
- Batteries were not different from several other items, which were in various stages of degradation caused by the landfill environment.
- No particular detrimental impact was noticed for any of the batteries.

Most likely the battery casing failures were from chemical corrosion. Active vertical earth loads become a more important factor for damage to batteries already structurally weakened from corrosion.The results from laboratory studies conducted at the Carnegie-Mellon University (CMU, 1989) indicated that batteries which are oriented parallel to the main compacting force of the landfill will hold together longer than those perpendicular to the force; this study concluded that dry cells are structurally strong and are not likely to be broken from the compactive forces in the normal waste disposal cycle.

3.1.2 Effects of Battery Disposal on MSW Landfill Leachate Quality

Leachate is produced in landfills as water and other liquids within the waste and any moisture that collects on top of the landfill flow through the waste. As this liquid moves down through the wastes, it flushes out and collects pieces of other materials. If a battery is cracked or corroded, the liquids in the landfill will pick up the metals contained in it. When leachate gets to the bottom of a landfill it flows along the liner to the collection pipes which carry it out to storage tanks. Most

landfill sites take the leachate, separate the water from it, and put the extracted elements back into the landfill where they may eventually leach out again. This cycle is followed while the landfill is in operation. Notwithstanding the use of liners in landfills, there is always the potential for some leachate to escape into aquifers beneath the site. Nonetheless, laboratory studies show that neither MSW leachate nor heavy metal leachate significantly affect polymeric or admix liners (CMU, 1989). Furthermore, because metals tend to be adsorbed on clay materials, where natural clay liners serve as landfill liners, the possibility of metals escaping from the landfill into groundwater below is limited. Laboratory column adsorption studies related to the attenuation of pollutants in municipal landfill leachates have found that heavy metals (including Cd, Pb, Hg, and Zn) are strongly attenuated even in columns with small amounts of clay (Griffin and Shimp, 1978); this is reflected in the low transfer coefficients shown in Table 7. These transfer coefficients give an indication of the fraction of landfilled metals that can potentially contribute to landfill leachate. These numbers are therefore important in determining the potential impacts of landfilling used batteries containing the metals of concern.

Table 7
Transfer Coefficients for Metals into Leachate in MSW Landfill.

Chemical	Transfer Coefficient
Cd	6×10^{-5} (≡ 0.006%)
Mn	Not available
Hg	6×10^{-5} (≡ 0.006%)
Ni	Not available
Zn	2×10^{-4} (≡ 0.02%)

Notes:
- Transfer coefficient, k = (specific total output in leachate at mean residence time) ÷ (specific total input by MSW)
- More than 99.9% of metals are still in residual solid after mean residence time of about 10 years.

Assumptions:
- The landfill is hydrologically stabilized after residence time of about 10 years.
- Fraction of metals which can be transferred to leachate conservatively taken to be about 0.05%

Sources:

1) "Water and Element Fluxes from Sanitary Landfills" by H. Belevi and P. Baccini. In, Sanitary Landfilling, ed. by T.H. Christensen, R. Cossu, and R. Stegmann (1989). Academic Press, London, UK.

2) "Long-Term Behavior of Municipal Solid Waste Landfills", by Belevi, H. and P. Baccini (1989). Waste Management & Research 7, 43-56.

Only limited research data are currently available regarding the actual threat posed by discarded household batteries to groundwater. A study conducted by Jones et al. (1977/78) determined that

batteries recovered from a landfill site after 7 months of burial were less corroded than those exposed to anaerobic leachate for 3 months in a laboratory. The study postulated that the reason for this result is that, in a landfill, a conducting pathway between the battery terminals may exist only intermittently, if at all. This condition reduces the ability of residual charge in the battery to contribute effectively to the corrosion process. Furthermore, the study determined that, based on analyses of the leachate from the tests, only a very small percentage, if any, of the Hg, Cd or Ni present in dry-cell batteries may be dissolved by the leachate. The study has concluded that, although Zn and Mn may be leached from large deposits of Leclanché cell wastes, there is no evidence of Hg being leached from landfilled material to any significant extent or that organo-mercury compounds are formed (Jones et al., 1977/78). Thus, provided they are well mixed with household wastes and provided recommended disposal practices are employed, the presence of spent primary dry-cell batteries in landfills may not present special threats to groundwater quality (Jones et al., 1977/78). In another study conducted by the Japanese Storage Battery Association (Oda, 1989) for Ni-Cads, it was concluded that cadmium from whole batteries would not be found (in landfill leachates) for two to four decades in a landfill under normal rainfall conditions.

In a recent laboratory investigation (CMU, 1989), a chemical analysis was carried out to determine whether batteries should be classified as hazardous or non-hazardous waste and to determine the effects of throwaway batteries on landfill liners. Both ruptured and undisturbed batteries were tested to determine how the condition of the battery affects its response to the Extraction Procedure (EP) toxicity test; the EP toxicity tests provide a measure of the potential leachability of solid waste in landfills. In the EP test an extract obtained from the solid waste is tested for the presence of specific chemicals (that includes cadmium and mercury). If the concentration of any of these contaminants in the extract exceeds the EPA limit (maximum concentration of 1.0 mg/l for cadmium and 0.2 mg/l for mercury), the solid waste is classified as hazardous. Based on information on battery constituents, the laboratory work was limited to metals that were likely to exceed the EP toxicity limits. EP toxicity tests were done for cadmium in Ni-Cad batteries and mercury in alkaline batteries. Because a significant amount of nickel is present in Ni-Cad batteries and because of the known toxicity of nickel, nickel was also analyzed, although nickel is not specified as a chemical subject EP toxicity tests. Although batteries are generally disposed of intact, they are subjected to many different stresses during the disposal cycle, potentially causing the cell casings to rupture, and allowing the contents of the battery to escape into the landfill environment. Thus, batteries were tested both with their containers crushed and in their typical unbroken condition; the results of the extraction tests (Table 8) are discussed below (CMU, 1989).

Table 8
Chemical Analysis of Household Batteries Using the EP Toxicity Method

Chemical Test	Maximum Concentration Limit (mg/l)	Battery Type	Battery Size	Battery Condition	Observed Chemical Concentration (mg/l)	Maximum Concentration Limit Exceeded?	Remarks
Cadmium (Cd)	1.0	Ni-Cad	AA	Whole	.078	No	Fully crushed batteries may pose environmental problems in MSW landfill conditions; whole cells may not be any threat due to release of Cd.
			AA	Whole	.125	No	
			Sub-C	Whole	.125	No	
			Sub-C	Whole, Corroded	.094	No	
			AA	Crushed, ends leaking	6.25	Yes	
			AA	Cut & Broken Open	37.5	Yes	
			Sub-C	Can Torn Open	90.0	Yes	
			Sub-C	Cut In Half	9.30	Yes	
Nickel (Ni)	--	Ni-Cad	AA	Whole	0	NA	Possible maximum concentration limit for nickel would be expected to be higher than for Cd. However, observed concentrations are all below limit for Cd. This means Ni poses no problem.
			AA	Whole	0.02	NA	
			Sub-C	Whole	0.26	NA	
			Sub-C	Whole, Corroded	0.02	NA	
			AA	Crushed, ends leaking	0.03	NA	
			AA	Cut & Broken Open	0.27	NA	
			Sub-C	Can Torn Open	.57	NA	
Mercury (Hg)	0.2	Zinc-Carbon		Corroded Whole	0.140 ± 0.028	No	No problems anticipated from Hg from zinc-carbon batteries.
		Alkaline	AA	Whole	0.006 ± 0.001	No	Hg in alkaline batteries may not pose significant risks, even for broken cells landfilled.
			AA	Whole	0.005 - 0.006	No	
			AA	Slightly Crushed	0.031 ± 0.006	No	
			AA	Very Slightly Crushed	0.001	No	
			AA	Broken	0.285 ± 0.057	Yes	
			AA	Broken	0.153 ± 0.031	No	

Source: Adapted from CMU, 1989

Mercury Test Results for Alkaline and Leclanché Cells:

Measured concentrations for mercury of the alkaline batteries were near the EPA test limit of 200 microgram per liter for severely damaged batteries while slightly damaged and intact batteries passed the mercury test (Table 8). Alkaline batteries passed the extraction test for mercury when their cases were intact or moderately damaged. Broken alkaline batteries provide conditions for mercury to escape to the environment, and consequently present a potential threat to landfill leachate. The Leclanché (zinc-carbon) battery type behaves differently from alkaline batteries. The case of the zinc-carbon battery is zinc while the case for the alkaline battery is either steel or plastic. Mercury is commonly a trace constituent in the zinc battery cases, as contrasted with Hg present in alkaline batteries as part of the contents inside the case. Hence, zinc-carbon batteries may more easily release their mercury content to landfill leachates than alkaline batteries.

Nickel-Cadmium Batteries Test Results:

Measured concentrations for nickel and cadmium depended on the condition of each battery (Table 8). Tests on broken batteries resulted in much higher levels of both nickel and cadmium. Tests for batteries with casings like new and those with corroded casings showed cadmium concentrations below the acceptable limits for cadmium. However, all tests with broken batteries had cadmium levels well above the limit. The concentration of the metals found in the extract appeared to be proportional to the amount of battery contents exposed to the extraction solution. Thus, broken Ni-Cad batteries failed the EP toxicity tests due to high cadmium levels while all the whole batteries passed. In conclusion, sealed and partially corroded batteries passed this modified version of the EP toxicity test for cadmium and nickel. All Ni-Cad batteries with ruptured cases failed the test based on observed cadmium concentrations; the extract concentration appeared to increase with greater damage to the case.

The laboratory study results (CMU, 1989) show that batteries are more likely to fail the EP toxicity test if their cases are damaged. It is useful to know the kinds of handling practices and loads required to damage battery cases. The U.S. EPA EP toxicity test includes a structural integrity test that consists of hitting the solid waste being tested with a 0.73 lb hammer using a six-inch free fall. This structural integrity test has been found to produce little or no damage to the battery. There is no guarantee, however, that the test is representative of conditions in the disposal cycle of dry-cell batteries in MSW. The results from the structural tests suggested levels at which batteries would fail for loads to both the top and sides of the batteries. No conclusive results could be obtained from these because of the difficulty in simulating actual landfill stress conditions. The following conclusions and policy implications are stated based on the CMU (1989) study results:

- The extraction procedure test shows that Ni-Cad batteries, if broken, fail the EP toxicity test for Cd. Even slightly damaged Ni-Cad batteries exceed the limit for cadmium. However, all whole and corroded batteries yield cadmium concentrations within the allowable limit. Since it seems that batteries would undergo some slight damage during the disposal phase, it may be concluded that landfilling Ni-Cads could lead to conditions where the quality of generated leachate is potentially detrimental to health and the environment.
- Severely damaged alkaline batteries showed levels of mercury near that allowed by the EP toxicity test. Because the probability of having a large number of heavily damaged batteries in one landfill is small, it may be concluded that mercury in alkaline batteries does not pose a problem in municipal solid waste landfills.

A new EPA test to replace the EP toxicity test is the toxicity characteristic leaching procedure (TCLP). The major difference between the TCLP and the EP toxicity test is its effectiveness in testing for volatile organics. However, the TCLP uses basically the same method as EP toxicity when testing for inorganics. In addition, the test uses the same standards as the EP toxicity test for determining whether or not inorganic wastes should be classified as hazardous. Thus the hazard classification for used batteries should not change with the use of the TCLP test.

A series of leachate characterization tests were performed in batteries readily available in the market place. Five analyses have been performed:

1. zinc carbon/zinc chloride
2. nickel cadmium
3. lithium
4. alkaline - retail sample
5. alkaline - warehouse sample

Batteries for the first four samples were purchased from several locations within the Kitchener-Waterloo area. The composition of each sample with respect to the size of battery (i.e. AAA, AA, C, D, 9v) was representative of the sales figures supplied by the Canadian Battery Market Estimates. The samples were subjected to particle size reduction as specified in Section 5.1 of the Leachate Extraction Prodedure found in the Waste Management General Regulation (Regulation 309) under the Environmental Protection Act for the Province of Ontario (For a complete description see Appendix A).

In this study the procedures followed will result in a worst-case scenario: Leachate characteristics are being derived from a monofil composed entirely of each sample type. The batteries have been

broken open and subject to particle size reduction. The results of the leachate characterizion tests for schedule 4 inorganics is presented in Table and for non-schedule 4 parameters in Table . The results as summarized in Table indicate that:

- Only Ni-Cad batteries when subjected to particle reduction would be considered leachate toxic because the sample concentration of cadmium - 2900 mg/L was greater than the criteria of 0.5 mg/L to be considered as leachate toxic or hazardous waste.

- Lithium batteries would be considered registerable solid waste because of the cadmium and fluoride contents, and is hazardous waste because of the reactivity of the lithium when exposed to the atmosphere.

- Zinc chloride/zinc carbon cells, because of the cadmium content would fall into the registeralbe solid waste category.

- Mercury levels in alkaline cells available from retail outlets was similar to those obtained from the warehouses. The levels of mercury of 0.012 and 0.014 mg/L respecitively just exceed the schedule criteria of 0.01 mg/L - the concentration above which materials are classified as registerable hazardous waste.

3.2 Incineration with MSW as a Disposal Alternative

Concern about airborne emissions from municipal waste combustors (MWCs), or incinerators, seems minimal at this time. However, there is growing concerns about ash disposal practices, which center on risks that can arise from pollution of groundwater supplies from landfills accepting incinerator ash. There is also concern about worker exposures to ash in MWC plants and at landfills and about risks to the public from wind-blown ash dust, especially during transportation of ash from MWCs to landfills (Ujihara and Gough, 1989). However, reviews and studies by Shaub (1988) and Kellermeyer (1989) conclude that risks to the public from ash dust are minimal. Although the conclusions are based on limited information, they do not appear to be as serious as possible exposures from contamination of water supplies by landfills accepting ash.

Incineration of batteries does not destroy the heavy metals contained in them. Thermal processes transfer heavy metals either to stack gas as fine particles or to bottom ash. An important characteristic of the ash is its toxicity and its status as a waste. In an incineration, the solid waste is burned, with bottom and fly ash remaining as end-products; the bottom ash falls through the grates of the incinerator, while the fly ash rises through the smokestack, and may be trapped by a collection device. Incineration reduces the volume of waste by about 80-95 percent, and the

Table 9

BATTERY LEACHATE STUDY
SUMMARY OF RESULTS - SCHEDULE 4 INORGANICS
REGULATION 309 LEACHATE EXTRACTION TEST

Parameter	Ontario Regulation * 309 Criteria (mg/L)	Concentration in Sample (mg/L) Sample Type 1 (Zinc Carbon/Chloride)	Sample Type 2 (Nickel/Cadmium)	Sample Type 3 (Lithium)	Sample Type 4 (Alkaline-retail)	Sample Type 5 (Alkaline-warehous
Arsenic	0.05	ND (0.005)	ND (0.005)	0.035	ND (0.005)	ND (0.005)
Barium	1.00	5.36	0.15	0.06	0.12	0.13
Boron	5.00	0.4	0.05	0.16	0.06	ND(0.05)
Cadmium	0.005	0.067	2900	0.31	ND(0.032)	ND(0.032)
Chromium	0.05	0.04	0.06	0.27	ND(0.02)	0.08
Fluoride	2.40	ND (0.05)	0.06	43.0	0.16	0.13
Lead	0.05	ND (0.10)	ND (0.10)	ND (0.10)	ND (0.10)	ND (0.10)
Mercury	0.001	0.003	0.0009	ND (0.0005)	0.012	0.014
Nitrate + Nitrite	10.00	ND (1.2)	ND (1.2)	ND (0.24)	ND (2.4)	ND (2.4)
Nitrite	1.00	ND (0.30)	ND (0.30)	ND (0.06)	ND (0.60)	ND (0.60)
Selenium	0.01	ND (0.005)	ND (0.005)	ND (0.005)	ND (0.005)	ND (0.005)
Silver	0.05	ND (0.02)	ND (0.02)	ND (0.02)	ND (0.02)	ND (0.02)

* Note: A leachate toxic waste is a waste which produces any of these contaminants in excess of 100 times the listed criteria. A registerable waste is a waste which produces any of the listed contaminants at a level of between 10 and 100 times the criteria

Table 10

BATTERY LEACHATE STUDY
SUMMARY OF NON-SCHEDULE 4 PARAMETERS
REGULATION 309 LEACHATE EXTRACTION TEST

Parameter	Ontario Drinking Water Criteria (mg/L)	Concentration in Sample (mg/L) Sample Type 1 (Zinc Carbon/Chloride)	Sample Type 2 (Nickel/Cadmium)	Sample Type 3 (Lithium)	Sample Type 4 (Alkaline-retail)	Sample Type 5 (Alkaline-warehouse)
Alumium	0.050 (1)	ND (0.10)	ND (0.10)	23.6	ND (0.10)	ND (0.10)
Beryllium		ND (0.005)	ND (0.005)	ND (0.005)	ND (0.005)	ND (0.005)
Calcium		16.9	0.87	ND (0.20)	1.8	1.06
Cobalt		ND (0.09)	ND (0.09)	ND (0.09)	ND (0.09)	ND (0.09)
Copper	1.0 (2)	0.31	ND (0.02)	ND (0.02)	ND (0.02)	ND (0.02)
Iron	0.3 (2)	ND (0.05)	0.28	ND (0.05)	ND (0.05)	0.24
Magnesium		4.50	10.4	0.50	0.18	0.16
Manganese	0.05 (2)	170	ND (0.01)	0.01	17.2	15
Vanadium		0.02	0.01	0.02	ND(0.01)	ND(0.01)
Zinc	5.0 (2)	1970	0.94	0.40	1190	1300
Nickel	0.350(3)	0.66	21.5	0.05	0.13	0.07
Strontium		0.43	ND (0.01)	ND (0.01)	ND (0.01)	ND (0.01)
Sodium	200 (2)	33.7	44.0	4.36	130	100
Antimony		ND (0.005)	ND (0.005)	0.008	ND(0.005)	ND(0.005)
Chloride	250 (2)	2050	ND (0.80)	ND (0.16)	ND (1.6)	ND (1.6)
Bromide		ND (1.90)	ND (1.90)	ND (0.38)	ND (3.8)	ND (3.8)
Phosphate		ND (5.50)	ND (5.50)	ND (1.10)	NA	NA
Sulphate	500 (1)	ND (5.90)	ND (5.90)	64.0	ND(11.8)	ND(11.8)

Notes: (1) This is a U.S.E.P.A. secondary maximum contaminant level (SMCL)
(2) Maximum desirable concentrations - parameters related to aesthetic quality
(3) This is a U.S.E.P.A. lifetime health advisory for a 70 kg adult.

Table 11
Battery Leachate Study
Summary of Waste Classifications
Based on Regulation 309 Criteria

Sample Type	Parameter(s)	Registerable Solid Waste (If Greater Than) (mg/L)	Leachate Toxic Waste or Hazardous Waste (If Greater Than) (mg/L)	Sample Concentration (mg/L)	Comment
1. Zinc Chloride/ Zinc Carbon	I) cadmium	.05	.5	0.067	Registerable Solid Waste
2. Nickel/Cadmium	I) cadmium	.05	.5	2900	Leachate Toxic
3. Lithium	I) cadmium	.05	.5	0.31	Registerable Solid Waste
	II) fluoride	24.0	240	43.0	Registerable Solid Waste
	III) Reactivity			NA	Hazardous Waste Sample caught fire on exposure to air, indicating a potentially reactive waste
4. Alkaline - retail	I) mercury	.01	.1	0.012	Registerable Solid Waste
5. Alkaline - warehouse	I) mercury	.01	.1	0.014	Registerable Solid Waste

weight by about 50-75 percent. Table 12 shows typical partitioning concentrations during incineration for the chemicals of concern found in dry-cell batteries; it gives an indication of how much of the metals present in MSW become available in the different forms and media.

Table 12
Typical Concentration for the Chemicals of Concern in MSW, MWC Ash and in Other Incinerator Effluents

Chemical Parameter	Household Waste (µg/g)	Fly Ash (µg/g)	Bottom Ash (µg/g)	Combined Bottom and Fly Ash (µg/g)	Incinerator Flue Gas (mg/m^3)	
					Raw Gas	Clean Gas
Cadmium (Cd)	10 - 40	<5 - 2,210	1.1 - 46	0.18 - 100	0.5 - 3.5	0.02 - 0.2
Manganese (Mn)	Not available	171 - 8,500	50 - 3,100	14 - 3,130	Not available	Not available
Mercury (Hg)	1 - 10	0.9 - 35	ND - 1.9	0.05 - 17.5	0.1 - 1.0	0.02 - 0-.2
Nickel (Ni)	50 - 200	9.9 - 1,966	9 - 226	13 - 12,910	0.5 - 5.0	0.1 - 1.0
Zinc (Zn)	500 - 3500	2,800-152,000	200 - 12,400	92 - 46,000	10 - 100	1 - 10

Note: All values given in µg/g (ppm), unless otherwise stated.
ND - Not detected at the detection limit.

Sources: 1) U.S. EPA, Characterization of MWC Ashes and Leachates from MSW Landfills, Monofills, and Co-Disposal Sites. Summary, Vol. 1, EPA/530-SW-87-028A (Washington, DC: EPA, October).
2) Ujihara, A.M. and M. Gough, 1989. Managing Ash from Municipal Waste Incinerators: A Report. Resources for the Future, Washington, DC.
3) Lorber, K.E. (1987)
4) Fochtman and Haas (1975)

3.2.1 Effects of Battery Incineration on Stack Emissions from MSW Incinerators

Most of the metals in incinerator emissions may be captured by the pollution control systems on an incinerator. It is believed that lime scrubbing systems using filter fabric typically remove over 99.9% of the fly ash; over 99% of cadmium and zinc, and 90-95% of mercury are removed by the filters (statement by Ray Klicius of Environment Canada, cited in CMU, 1989). Conservatively estimated, USEPA (1989) also indicates air pollution control devices (APCDs) efficiencies for controlling Cd of up to 99%, and up to 98% for Hg. MPCA (1991) agrees that existing air pollution control equipment are very effective at removing particulates and gases containing metals from household batteries; current dry and semi-dry air pollution control systems are believed to be able to collect 95% or more of the metals found in MSW, except for Hg. A wet scrubber or a wet/dry scrubber can achieve higher collection efficiencies for Hg (Brna, 1991; Donnelly, 1991).

Mercury creates the most concern in the incineration process; because of its low boiling point, it may not always condense on the fly ash and therefore it might be released into the atmosphere. The control of mercury emission to the atmosphere from incineration of household batteries in domestic waste is an important pollution control issue. Different air pollution control devices vary in their effectiveness in capturing mercury. Some studies show that only 25 to 30 percent of the mercury is collected by the collection systems (cited in CMU, 1989); this issue is presently being studied by the EPA (CMU, 1989). A spray dryer and baghouse system can achieve 75 to 85 percent mercury removal, thus capturing the mercury in the ash; a spray dryer plus electrostatic precipitator can achieve about 35 to 45 percent removal (Ellison, 1986, 1991; MPCA, 1991). If it is determined that mercury collection is a problem, then removing high-content mercury batteries from the MSW stream could assist in abating the problem. Other metals from the incineration of household batteries, including cadmium, are largely collected in the ash from the incineration process; about 92.4 per cent of the cadmium in incinerated waste stream is captured in the total ash (Vogg et al., 1986; MPCA, 1991).

3.2.2 Effects of Battery Incineration on Incinerator Ash Toxicity

Fly ash consists of the airborne particles that are captured by filters in the incinerator stack. Bottom ash is the heavy residue formed at the bottom of the incinerator after the municipal solid waste is burned. In the process of burning wastes, some of the heavy metals present may volatize, and then condense on particulate matter as it rises through the smokestack. Consequently, the fly ash will have a high concentration of heavy metals. Part of the metals do not volatize though, and these contribute to the bottom ash generated in the process. Of the ash created through the

combustion process, about 10% is fly ash and 90% is bottom ash (CMU, 1989; Environmen Canada, 1991). The partitioning of metals between bottom and fly ash varies between incinerators and may be influenced by such factors as the operating temperature of the incinerator and whethe the material being burned is suspended in the air or on the floor of the incinerator. In fact, the partitioning of metals between the bottom and fly ash is a poorly understood aspect of incineration. Since most incinerator facilities mix the bottom and captured fly ash for disposal, the total amount of metals contributed by batteries to the total ash will approximately be the same, regardless of how it is partitioned between the bottom and fly ash, assuming none or only minimum amounts of the metals escape through the smokestack. Results of EP toxicity tests of ash samples have been generalized as follows (Ujihara and Gough, 1989):

- almost all fly ash samples exceed EP toxicity limits for cadmium (and also lead);
- most combined ash (i.e., bottom and fly ash) samples surpass EP toxicity limits for lead, but not for cadmium; and
- some bottom ash samples are below the limit for lead and all are below the limit for cadmium.

Laboratory studies show that cadmium, one of the toxic substances of concern associated with household batteries, is one of two metals (together with lead) that tends to be more concentrated in the fly ash than in the bottom ash (Ujihara and Gough, 1989; Sawell, Constable and Klicius, 1991). Since Cd is carcinogenic by the inhalation pathway, it is crucial that adequate scrubbers are used in MSW incinerators, that will capture as much of the fly ash as possible and to minimize the amounts that could eventually reach potential human receptors. In the absence of that, Ni-Cads which may be a significant contributor of Cd in MSW may have to be removed from the wastes to be incinerated.

Indeed, this is supported by results from an evaluation conducted at Carnegie-Mellon University (CMU, 1989) which showed that removal of Ni-Cad batteries from incinerators could reduce ash carcinogenic toxicity by about 2 percent and ash non-carcinogenic toxicity by about 0.5 percent. The significance of these values may be investigated using detailed methods of risk assessment. For example, assuming that percent reduction in ash toxicity is on the order of percent reduction in risk from exposure to ash (i.e., assuming a linear model), it can be inferred that for every 100 cases of cancer due to status quo incinerator emissions of ash, an average of 2 of those cancer cases will not occur if Ni-Cads are removed from incinerators and disposed of in a manner in which humans are less likely to be exposed to their contents. Since the components used in the calculations are not the only pollutants released from MSW incinerators, the calculated values should be interpreted as the upper limit for risk abatement. Again, based on an analysis of a comprehensive risk assessment for a hazardous waste incinerator (cited in CMU, 1989) in which

an extensive list of both organics and inorganics were used, the majority of the total carcinogenic risk from ash was due to cadmium, arsenic and chromium. Similarly, the major metals contributing to noncarcinogenic risk was associated with manganese (and also lead).

A number of caveats are associated to these generalizations, however, due to inherent problems with testing ash. Also, limited field data do reveal that concentrations of Cd (and Pb) in leachates from "real-world" ash disposal sites are generally below EP toxicity limits, but few samples have approached the limit for Pb (Ujihara and Gough, 1989). It should be cautioned, however, that field data exist only for a few years; this limited data makes conclusive statements difficult to make.

In the meantime several options have been suggested to manage the ash and reduce the risks:

1. attempt to remove the metals from the waste stream prior to combustion.
2. dispose of the ash in properly designed landfills so that the materials do not contaminate groundwater.
3. heat the ash prior to disposal so as to remove the metals.
4. reuse the ash in a way that human exposure does not occur. Some examples include the use of ash in cement-based products, as fill in base in roadbeds, and as a component in asphalt paving (Ujihara and Gough, 1989).

3.3 Recycling as an Option for the Management of Used Batteries: Logistics and Viability of Recycling Household Batteries

This section consists of a review of the literature pertaining to battery collection systems, the current status of battery recycling technology for the various battery systems, and the costs of battery recycling, in order to evaluate the feasibility of implementing battery recycling programs in Canada. The collection and recycling of used household batteries poses several unanswered technical and economic problems, and until these are resolved, it would be premature to recommend widespread implementation of such a program.

The mercury and silver oxide (button) batteries are the only battery types for which recycling technologies have been developed and have the potential to operate in a cost-effective manner (subject to fluctuations in the market value of mercury) due to the high concentration of mercury and silver. Mercury batteries are the source of seven tonnes of a total of 48 tonnes of mercury emitted in Canada per year. Therefore, a collection program for mercury batteries alone would not significantly reduce the emissions of mercury from discharged batteries (Environment Canada, 1991). Furthermore, while source reduction in the mercury content of these batteries is not likely,

there is a trend towards a declining market share of this type of battery as they are increasingly replaced by zinc-air cells which contain only 1 percent mercury by weight.

While there is no established industry for recycling/processing mixed collections of household batteries, some recycling technologies have been tested. However, commercial implementation of mixed battery recycling appears to have basically poor economics due to the readily available, low cost, high purity, virgin materials. Further barriers to the development of large scale household battery collection and recycling programs are that the recycling processes are not fully developed (i.e. separation of zinc and manganese) and secondary markets for recovered metals do not exist in all cases. The collection rate for used batteries is generally low, even in countries where the public awareness has been high (Environment Canada, 1991). The public is likely to be unwilling and/or unable to distinguish between alkaline cells and non-alkaline cells, or between zinc-carbon cells and nickel-cadmium cells. Thus recycling plants would need to be designed to sort a complete mix of consumer cells (Little, 1989). However, there are currently no cost-effective methods to automate sorting of cells.

None of the most prevalent types of batteries (alkaline, carbon-zinc and heavy-duty carbon-zinc) are being commercially recycled due to the fact that their major components are metals with low value. While a Japanese pilot project demonstrated satisfactory recovery and marketing of mercury and iron, separation and the identification of economical markets for the major components zinc and manganese dioxide has not been resolved. Whether a reutilization of the recovered secondary materials will be possible in the future is uncertain. The recovered zinc and manganese dioxide are currently of a low purity and as such cannot be marketed as raw materials.

Ni-Cad batteries can be smelted to extract metals for reuse in the manufacturing of steel. The smelting process is expensive and must be subsidized by the provider of the batteries. The impact that a Ni-Cad battery collection and recycling program would have on the diversion of cadmium from the waste stream would be limited given that the consumer does not have access to 80% of Ni-Cad cells which are manufactured into cordless rechargeable appliances.

While cost-efficient recycling processes for battery reclamation do not currently exist, it is not unreasonable to suggest that technological advances will make this option more feasible in the future (CMU, 1989). Recycling of batteries could be developed if clean technologies are found and viable end-use markets for the reclaimed materials are secured. Proposals for establishing battery recycling facilities to handle all types of batteries have been developed or are in the process of investigation by Recytec, Voest-Alpine, Bronx 2000 Development Corporation and J.L. Recycling. However, it may take several years to establish recycling collection systems,

processing plants and infrastructure. The current environmental concerns associated with household battery disposal may be alleviated or have changed in that time frame. For example, the decline in the mercury content of alkaline manganese batteries, the changing market share of the mercury button cell with the introduction of the zinc-air battery, and the recent introduction of alkaline rechargeables which will compete with disposable alkaline batteries and rechargeable Ni-Cads, show that approaches designed to solve current problems could become less effective or even obsolete in a short time.

3.3.1 Current Situation

In most Canadian municipalities, household batteries are largely treated as other household waste and are disposed of in incinerators or landfills. For instance, in Toronto, batteries of all types are also collected at transfer stations through a voluntary drop-off system and can be collected by the "Toxic Taxi" service, which will pick up household hazardous waste (must be a minimum amount) for subsequent disposal at a hazardous waste landfill.

In 1991 the Citizens' Clearinghouse on Waste Management and the Victoria Environmental Network, with funding from Environment Canada, undertook a pilot project to collect used dry-cell batteries from households. Over 4,000 batteries were collected by school children, hospital personnel and the general public (Citizens' Clearing house on Waste Management, 1991).

In Waterloo region batteries are included in "Household Hazardous Waste Days" held once or twice a month in which residents voluntarily drop off hazardous waste at a designated depot in Waterloo. This waste is then disposed of at the Sarnia hazardous waste landfill by Laidlaw Environmental Services. Household batteries are classified into a broad waste classification called 'miscellaneous/inorganic household hazardous waste"; therefore statistics on the volume of household batteries collected are not available. There are no immediate plans to expand the existing household battery collection program, however plans to create a permanent depot for household hazardous waste will make it more convenient for residents to participate in the existing program. In Hamilton-Wentworth region, household batteries can be disposed of at the hazardous waste depot. Batteries are currently being stored while the feasibility of conducting a pilot project for recycling is investigated.

At present, there are no established facilities for recycling of household batteries of any type in Canada. J.L. Recycling in Ottawa has been operating as a third-party broker for recycling of household batteries for the past year and a half. The company accepts mercury-oxide, silver-oxide, zinc-air, zinc-carbon, and nickel-cadmium (Ni-Cad) batteries which are then sent to recycling

facilities in the U.S. and Europe. While their primary customers are currently government agencie and private companies, additional clients are being sought through market surveys. The company is proposing to begin a recycling operation next year that will process all types of household batteries, including alkalines. The technology is being developed by a company in Switzerland, using a closed loop, chemical process. They are proposing that 95% by weight of materials will be recycled and the reclaimed metals will be of 99% purity. The company expects to be granted approval for the facility by the Province fairly readily as a public review process is not required. It was suggested that it would be much more difficult to gain approval for a recycling facility that uses a thermal process, as a public review process would be required (Duron, 1991).

3.3.2 Collection Systems

Battery collection programs have been implemented in several communities in the U.S. and Europe for storage or separate disposal in hazardous waste landfills, in order to divert batteries from municipal incinerators. The basic elements of used battery collection programs include: public education, installation of collection containers or organization of curbside pick-up, the organization of a system to retrieve containers, storage and/or disposal or recycling of the collected batteries (Environment Canada, 1991). Overall cost effectiveness is a function of participation and cost to the collection agency. Participation is dependent upon the cost and convenience to the consumer (CMU, 1989). Two options have been investigated for the collection of household batteries. These options include: retail drop-off centers and curbside collection programs (CMU, 1989).

Retail drop-off Centers:

In the U.S. a small number of retail outlets have set up their own button battery collections, primarily in jewelry shops (watch batteries) and hearing aid supply outlets. The New Hampshire/Vermont Solid Waste Project began collecting batteries of all types in retail stores in 1987. They placed 5-quart silver buckets with a battery recycling logo in about 70 stores and recycling centers. The area included 26 municipalities with a total population of approximately 60,000. The startup costs were less than $1000, and collection was done by volunteer groups, thus they were able to keep operating costs of collection low. Although the centralized collection option is easy to set up and has low capital costs, it will not by itself greatly reduce the amount of batteries in the municipal solid waste stream due to low participation rates. The project reported a 6 percent recovery rate (CMU, 1989). While the original plan was to sort the batteries to recover button cells and Ni-Cads, sorting was never carried out. The mixed batteries were sent to a hazardous waste landfill. It cost the New Hampshire/Vermont Solid Waste Project $350 for each 55 gallon drum (approximately 650 pounds of batteries) or about $1,075 per ton to dispose of the

spent batteries as hazardous waste (MIT, 1990). With all batteries going to a hazardous waste landfill, costs for the program would rise significantly with increased participation in the collection program.

Curbside Collection:

While curbside collection of batteries can be costly, the higher participation rates and ease of combining it with existing curbside collection programs can make it the most cost effective alternative (CMU, 1989). In 1989, Hennepin County, Minnesota conducted a study designed to investigate the most effective methods for removing household batteries from the waste stream. The county initiated pilot battery collection programs in two towns with similar population characteristics - one using curbside collection of household batteries, the other collecting household batteries through a retail drop-off program. To date, this project has generated the most useful information on participation in, and success of, household battery collection programs in the U.S.

Six to seven times more batteries were collected in the curbside method of collection. The number of batteries collected per household was also about 50 percent higher than in the retail collection method (MPCA, 1990). Ninety-four percent of the batteries were alkaline or zinc/carbon, three percent were Ni-Cads or button batteries. This suggests that curbside collection of household batteries is a more effective collection strategy. It was also suggested however, that part of the difference may have been attributable to the fact that residents of the curbside pickup town received a more intensive advertising campaign (MIT, 1990). After storing the batteries for nearly a year, Hennepin County ended up sending them to a hazardous waste landfill at a cost of about $1000 per ton.

Both HennepinCounty, Minnesota and the New Hampshire /Vermont Solid Waste Projects, as reported, have attempted to remove household batteries from MSW. On the basis of data collected and experiences gained, both have recently recommended that the most efficient way to direct mercury and cadmium is to reduce the amount of mercury used in batteries such as alkaline cells, to focus attention on the data collection and recycling of batteries that contain mercuric oxide, require small cordless appliances be supplied with recyclable batteries which can be readily removed, and to look for reductions of metals from other sources (Amos, 1991; Johnson and Hirth, 1990).

3.3.3 Feasibility of Recycling by Battery Types

Several factors are involved in determining the economic feasibility of household battery recycling:

- Certain recycling processes require a homogeneous supply of batteries, such that the batteries to be processed are of one type. This makes the consideration of separation processes important.
- The cost of reclamation may exceed the cost of extraction from ore.
- Reclaimed elements must be of a purity that will be marketable to industries using the elements.
- Subsidization may be needed to encourage or initiate reclamation.

Despite the existence of technology to recover some types of batteries, its overall economic feasibility remains controversial. In general, the recovery of used batteries meets technological and economic obstacles that limit the effectiveness of organized collection programs. In many cases, these have caused projects to fail after a short period of time (Environment Canada, 1991).

The obstacles cited in recycling dry batteries include :

- Insufficient control over battery supply and low or unstable market value of the materials recovered (Environment Canada, 1991).
- Insufficient purity of recovered metals; raw materials are in plentiful supply commercially and available at low cost
- Changing composition of battery systems, i.e. decreasing content of mercury in alkaline batteries and decreasing market value of mercury
- Poor environmental record of mercury refining facilities
- Different battery systems require different recycling technologies, therefore the various battery systems must be collected separately or manual or mechanical separation is required at the recycling facility, adding to the cost of the operation. Consumers cannot readily distinguish between different battery systems.

The closest mercury recycling plant to Canada is located in Albany, New York. Mercury Refining (MERECO) Company is the only company in the U.S. that accepts button batteries. After the mercury and silver are recovered, remaining material (about 80% by weight of the original) is sent to a secure landfill. There is no charge for sorting. A fee is charged to dispose of the lithium cells. Nickel cadmiums are accepted at no charge and are sent to a recycler in France. Alkaline and carbon-zinc batteries are also accepted for a fee and are sent to a secure landfill operated by Chemical Waste Management in Model City, N.Y. (Reutlinger, 1990). MERECO has had in the past, contamination problems associated with mercury, although reports now suggest that facility improvements will address previous environmental problems (Reutlinger and DeGrassi, 1991).

Nickel-Cadmium Batteries:

Sweden and Switzerland both passed Ni-Cad battery recycling laws in 1986. Battery collection has been under way on a voluntary basis in West Germany, the Netherlands and Austria since 1989. Ni-Cad batteries are collected by the manufacturers and sent to a recycling or controlled disposal centre. The cathode is sold for its nickel content, and the anode for its cadmium content. Research across Europe has demonstrated that sealed Ni-Cad cells can be reprocessed but recycling may be hampered by adverse economics. Few facilities in Europe are capable of reprocessing sealed Ni-Cad batteries. Until recently, SNAM in France and SAB-Nif in Sweden were the only commercial operations. They were joined in 1989 by SAVAN, a joint venture between SNAM and cadmium producer Vielle Montagne, also in France (ENDS, 1990). These recycling plants are mainly interested in the nickel used in secondary cells which is sold to steel manufacturers (Environment Canada, 1991). Some small recovery firms are stockpiling the cadmium anode and treating only the nickel cathode, while waiting for an increase in the market price of cadmium (Environment Canada, 1991).

Nickel-cadmium batteries are collected by J.L. Recycling Ltd. in Ottawa and are shipped to Europe for processing. Sanyo Energy U.S.A. corporation will offer a recycling program later this year that will allow consumers to return used Sanyo Ni-Cads for recycling at a company plant in Japan. There are several metal reprocessors in the U.S. that accept industrial Nickel-Cadmium batteries. Until recently however, none of these companies accepted consumer Ni-Cads (Reutlinger, 1990). Nickel-cadmium batteries are now being recycled in the U.S. by Inmetco in Ellwood, Pennsylvania, which operates a secondary smelting furnace for the high-temperature recovery of nickel, chromium and iron. Inmetco fumes off zinc, lead and cadmium, which are collected in a wet scrubber and in the dust. These byproducts are sent to Horsehead Resource Development Co. Inc. in Palmerton, Pennsylvania, a residue processor (American Metal Market, 1991; Schweers et al., 1991).

Experience at the existing plants and several research establishments have shown conclusively that recycling of sealed Ni-Cad batteries is technically feasible. Recovery and reprocessing techniques are far from being optimized, however, and recent research has yielded a number of important advances. The Dutch research group TNO is testing a hydrometallurgical process in which cadmium is leached from shredded batteries by hydrochloric acid and then recovered by solvent extraction. Unlike thermal processing, TNO's method recovers pure nickel as well as cadmium.

Collection and sorting of Ni-Cad batteries continues to be a problematic. Mercury batteries should not be present in the lot of nickel-cadmium batteries to be recycled. The presence of 1% of mercury may cause impurities and considerably increase the cost of recycling (Environment Canada, 1991). In Germany, 50% of batteries collected to date have been primary cells, not Ni-Cad batteries. Hand-sorting of Ni-Cad cells represents approximately one-half of the total costs, excluding reprocessing. At current world cadmium prices, these costs make recycling uneconomical (ENDS, 1990).

The amount of cadmium produced in Canada has been estimated to be 1570 tonnes in 1989 and 1463 tonnes in 1990 (Cadmium Association, 1990). The rechargeable battery market dominates the world consumption of cadmium (Table 13). These estimates compare very favourably with estimates provided by the U.S. Bureau of Mines data for cadmium demand in 1984 (A.D. Little, 1989).

Table 13
Estimated Market Share for Various Cadmium Uses in 1980 and 1990

Uses	1980 %	1990 %
Batteries	23	55
Pigments	27	20
Stabilizers	12	10
Coatings	34	8
Alloys	3	3
Miscellaneous	1	4

Source: Cadmium Assoc., 1990

Approximately 80% of Ni-Cad cells sold are sealed inside cordless rechargeable appliances. If consumers recycled all of the nonsealed nickel-cadmium batteries, the result would be recycling of 19% of the originally reproduced nickel-cadmium batteries. This figure equates to a recycling of approximately 150 to 185 tons of cadmium annually or about 4 to 5 percent of the U.S. total cadmium usage (Europile, 1986). Given that achievable recycle rates for recyclable materials (e.g., aluminum cans, plastic bottles, newsprint) tend to be less than 50 percent on a national basis, the reduction of cadmium in the waste stream that could be effected by recycling of consumer cells is likely to be proportionately very small (Little, 1989).

Recent legislation in Minnesota and Connecticut requires that by July 1993, nickel-cadmium batteries in rechargeable household appliances must be designed to allow for removal of the battery.

Zinc-Carbon and Alkaline Batteries:

In several U.S. cities, concern about non-recyclable alkalines has led to the development of alkaline collection-disposal systems. The motivation for these programs was to remove these batteries from the waste stream to reduce the mercury content in incinerator emissions and ash. The collected batteries are sent for disposal to a hazardous waste facility or stockpiled until a recycling option develops (Reutlinger, 1990).

The declining mercury content in alkaline batteries is key to the discussion of whether to separate these batteries from the waste stream. There is no consensus among U.S. recycling program co-ordinators as to whether the expense of collecting and disposing of alkalines is justified in light of recent mercury reductions. Some recyclers feel that any amount of mercury in the waste stream is too much, and also raise concerns about the large amounts of zinc and other metals in batteries. Others feel that by the time a municipality could implement a substantial collection program, the manufacturers will have further reduced the mercury content (Reutlinger, 1990).

There is currently no established industry for reprocessing alkaline-manganese and zinc-carbon battery systems. Environmental Pacific Corporation of Lake Oswego, Oregon, has indicated that they grind up the dry cells and export them to processors in the U.S., Europe, and Asia. None of the U.S. battery recycling or collection-disposal programs send batteries to this company as they were not knowledgeable about the environmental and worker safety records of the foreign processors (Reutlinger, 1990). More recently Environmental Pacific has been charged by the Oregon Dept. of Environmental Quality, is facing increasing hostility from nearby residents, and has been found to lack the appropriate permits to store used batteries (Watson, 1991).

The collection of carbon-zinc and alkaline batteries is costly, and their recovery is not profitable because they contain only a small percentage of mercury. The cost of recycling exceeds the value of the materials recovered (Environment Canada, 1991). The trend toward mercury reduction in alkaline batteries would reduce the economic feasibility of recycling this type of battery system as mercury is the most valuable component.

For standard alkaline zinc/manganese and non-alkaline/zinc carbon cells (non-button cells only), Voest-Alpine Environmental Engineering of Austria has developed a pilot process in which the

batteries are crushed and screened magnetically. In this process, all ferrous scrap, plastic and pape is removed, leaving a crude black mass which is composed of manganese dioxide, zinc flakes and carbon. The black mass is then thermally treated allowing the mercury and zinc to be recovered (either as the metals or their oxides) and separated in a stage called the "vapour" phase. To remove any possible increased amounts of potassium and sodium, the black mass can be leached, before or after thermal treatment, and this black mass can be used as a primary ore substitute in the manufacture of electrolytic manganese dioxide. Despite the available European technology, markets currently do not exist for the recovered metals (Taylor et al., 1988).

In 1985, with the financial backing of the government, a non-profit organization, Clean Japan Center, built an experimental plant to recycle used dry batteries and fluorescent tubes containing mercury. Operation of the plant, for the purpose of technical and economic demonstration, continued until September 1987. The project aimed at recovering mercury, zinc and manganese (in combination) and iron. The plant operation consisted of the following processes:

1. Pretreatment in which batteries are sorted into five groups according to shape and size, with further separation of zinc-carbon batteries and alkaline-manganese batteries by taking advantage of the weight difference. Hand removal of foreign materials is still required to a certain extent. Iron jackets of the pre-sorted batteries are detached and recovered as iron scrap. The batteries then undergo a process of crushing and mixing.

2. Roasting using a rotary furnace heated to 600 to 800 degrees Celsius to decompose and vaporize volatile substances such as mercury. Combustion fuel for the furnace is liquefied petroleum gas (LPG), which is injected against the flow of feed material.

3. Magnetic separation is used to sort out the calcine generated by roasting into iron scrap and zinc residue.

4. Gaseous emission containing vaporized mercury from the roasting process is sent through dust collecting equipment to a condenser where it is cooled to below the dew point of mercury for the recovery of metallic mercury. Dewdrops of the metallic mercury condensed on the internal surface are periodically recovered by water flushing and refined to the purity of 99.99% or above for sale.

5. A gas treatment process removes and neutralizes completely trace amounts of gaseous mercury and chloride by chemical cleansing, wet electrostatic precipitation and gaseous mercury adsorption.

6. The condensing process and the gas treatment process as well as a waste water treatment process are all connected serially in a closed circuit. Rinse wastes and drainage discharged from those processes are all collected. After mercurial substances are stabilized and removed, the waste water is vaporized to crystalize salts. The collected waste water is agitated with mercury stabilizing agent and the pH value is adjusted before being filtrated. The filtrated water is vaporized and the residues are all returned to the roasting furnace.

Levels of exhaust gases were in compliance with the emission standards provided in the Air Pollution Prevention Act. Sulfuric oxide was less than 1% of the emission standard and hydrogen chloride did not reach the minimum concentration for quantitative analysis. Mercury concentration was below the World Health Organization's guideline value. Industrial water is recycled in a closed circuit so that no water is discharged to public sewers.

Mercury is refined to a purity of 99.99% or above. Zinc content in the recovered calcine is approximately 35% on the average. Other major components of the calcine are manganese (23-30%) and carbon (14-18%). Two kinds of iron scraps are recovered, one with the peeling machine and the other is recovered from calcine through the magnetic separator. Iron scrap recovered with the peeling machine has an average quality of 98% while that recovered with the magnetic separator has a quality of approximately 94%.

Recovered mercury was applied to the material for batteries, inorganic chemical agents, measuring instruments, electric relays, lamps, amalgams and others. Iron scraps were supplied to iron and steel manufacturers through their contracted vendors. In spite of the favourable recovery, zinc residues which made up 54% by weight of the incoming batteries were not marketed due to the collapse of international price and the rate of foreign exchange.

It was concluded that the plant demonstrated favourable performance, resulting in the establishment of technically stable conditions for recycling battery wastes. The demonstration suffered from uneven quality and unstable delivery of used batteries, while marketing of the products was adversely affected by drastic changes in the rate of foreign exchange and in the price of metals at international market. Income from tipping fees and revenue derived from the sale of recovered materials, failed to cover expenditures.

Processes for recycling mixed batteries including alkalines are in the development stages. Proposals for recycling plants have also been put forward in Austria, Switzerland, Germany , Canada and the Netherlands. Experience to date is not very positive. It has been found to be difficult to produce recovered material of salable quality and at a reasonable cost. It appears

uncertain whether any commercial recycling plants for alkaline manganese and/or zinc carbon batteries will become available in the future (University of Lund, 1990).

Some exploratory work has been initiated in Canada by Falconbridge Ltd. to recover contained metals of interest - including Hg, Cd, Ag, etc., from used batteries collected from domestic garbage (Freeman, 1990). Conventional mineral dressing techniques which are commonly used in the mining and metallurgical industry were employed to separate and concentrate components into fractions acceptable for recycling. The batteries were collected from a pilot recycling program in Toronto. Although the recovered iron and zinc concentrates were assumed to qualify for recycling; nevertheless, some 75% of the remaining mass was unsuitable for recycling. The authors suggested that the unrecyclable mass could be disposed in pond tailing of existing mines. A related issue which arose with the various approaches listed was what to do with the wash water - it would require neutralization and metal recovery before discharge.

Battery Separation Systems:

The separation of household batteries by type is an important consideration in the feasibility of recycling as different battery systems require different processing technology for reclamation. For example, the cadmium in Ni-Cad batteries acts as a contaminant to mercury, reducing the yield of mercury during recycling. Better identification and separation of batteries is a key to economical battery reclamation. At present however, there is no efficient way of distinguishing among the different types of button cells which include mercury, silver, zinc-air and lithium systems.

The possibility of mechanical sorting of collected batteries has been studied at TEM, University of Lund in Sweden. Mechanical sorting appears to be feasible only when all battery types are to be collected. From the point of view of the treatment technology, the batteries should be sorted into the following groups:

1. Button cells
2. Nickel-cadmium
3. Alkaline manganese and carbon-zinc

Button cells can be separated in a relatively straight-forward way due to their size. While further sorting of button cells by weight is technically possible, it was concluded that the stringent requirements in the size and weight sorting steps and the small quantities of nickel-cadmium and zinc carbon button cells in Swedish battery collections made it infeasible to develop equipment for the sorting of button cells. Carbon-zinc batteries can be separated from other batteries according to weight with a relatively high efficiency. Alkaline manganese and nickel-cadmium batteries can be

separated by measurement of resistance between a pole and the inner casing, or by ultrasound (University of Lund, 1990).

The Swedish report concluded that the mechanical separation plant would not be economically viable, purely on the basis of low battery volumes collected in Sweden (approximately 100 tons per year). Countries with more dense populations that decide to collect all battery types and have shorter transport distances may find the economics more favourable. Alternatives to mechanical sorting involve markings that are invisible under normal circumstances including; markings that become visible under ultra-violet light, markings that become visible under infra-red light or a magnetic bar code (University of Lund, 1990).

Reclamation Processes:

One of the objectives of recycling primary batteries is the recovery of the contained metals including mercury (Hg), silver (Ag), zinc (Zn), manganese (Mn) and iron (Fe). The proportion by weight in the various battery systems amounts to almost 30% for Hg or Ag in the relevant button cells, 25 to 30% for Mn in cylindrical cells, Zn is present in all cells - with the exception of lithium on a scale of 10 to 20%, Fe is also present in all cells, as the protective container, accounting for 20 to 30% by weight. Many other metals are present in lesser quantities and variable amounts depending on specific technology: copper, cadmium, calcium, magnesium, titanium, aluminum, tin, lead, and others. Some inorganic salts are also present as well as plastic, paper, and graphite in various forms (Bartolozzi, 1990). Nickel-cadmium cells contain 15% cadmium by weight.

Since batteries are made of several metals, they require expensive and complex processing for successful extraction of the recyclable elements. The installations must meet strict environmental standards when mercury and cadmium are involved. The reclamation process separates metals for reuse and wastes that can be appropriately disposed. Separation of batteries by type is usually required by reclamation plants, however facilities that mechanically separate mixed batteries are in the development stages. Several reclamation processes including pyrometallurgical and hydrometallurgical have been tested and are described below.

Pyrometallurgical processes remove battery elements by crushing and roasting the batteries. Roasting is a process where the batteries are heated to evaporate off the desired elements. In principle, all mercury containing batteries can be treated in this way. The batteries are ground and the mercury is distilled at a high temperature and low pressure. Through further heating, crude zinc and cadmium can be removed. If the process is continued, manganese and iron and nickel can be

recovered in crude form. The processing of used batteries results in residues that must be disposed of in hazardous waste facilities (CMU, 1989).

Hydrometallurgical processes operate either at room temperature or at temperatures not exceeding 100 degrees Celsius. The spent batteries, usually first crushed, are then leached with hydrochloric or sulphuric acids, before treatments to precipitate MnO2, Fe(OH)3, Zn(OH)2 or, alternatively, the purified solutions are electrolyzed. Often the mercury is removed in advance by heating (Bartolozzi, 1990).

The hydrometallurgical process could be favourable to the pyrometallurgical process, both economically and environmentally. The process could be less energy intensive and therefore less costly, and results in substances of a purer form. The self-contained nature of the process and potential recyclability of the pH solutions, which may be able to be fed back into the process would lead to less production of hazardous waste byproduct. The roasting process used in pyrometallurgy has higher emissions due to the burning than would a solution based process (CMU, 1989).

Costs:

Arthur D. Little Inc. developed an estimate of the costs of reprocessing household batteries through a plant designed to handle alkaline, carbon-zinc and nickel-cadmium batteries. Based on the annual volume of consumer cells sold (excluding button-cells), the total processing cost would be approximately 4.08 cents per cell or nearly $50 per cubic foot ($1695 per cubic meter). This does not include the costs of collection. Against these costs, the value of the plants crude products (mercury, zinc, cadmium, iron and manganese dioxide) might supply on the order of 30 percent of the overall costs of recovery (1.45 cents per cell). These credits were based on pricing estimates resulting from interviews with scrap dealers, chemical companies and metallurgical companies. The highest valued material was mercury, and the declining content of mercury in alkaline batteries would tend to significantly reduce the value of the recovered products. The study concluded that in all instances, the cost of virgin materials was substantially less than would be required to recycle the key materials (Little, 1989).

In the six month battery collection program conducted in NH/VT, 135 gal. of non-recyclable batteries were collected. The cost to dispose of these batteries in the hazardous waste landfill was $750.00. The maximum revenue obtained from the recycling of mercury and silver button batteries was estimated at $105.00.

3.3.4 Policies on Recycling Programs

For purposes of comparison, it is interesting to examine whether or not other industrialized nations have regulated the disposal of household batteries. Many have either mandatory or voluntary programs for separating batteries out of the municipal waste stream. Sweden and Switzerland require that batteries containing more than 0.025% mercury or cadmium be marked with both an "environmentally hazardous" symbol and a "recyclable" symbol. In Sweden, manufacturers must pay a levy on these batteries, equal to 15-20% of their retail price. In addition, batteries must be separated from other household waste prior to disposal. These batteries are then stored of landfilled as hazardous waste. Both Norway and Denmark are considering legislation similar to that of Sweden. Italy and Holland also have enacted legislation requiring that batteries be disposed of separately from organic household waste (Primary Batteries in Waste and the Environment, NEMA Position Papers I and II, 1989).

In January of 1989, the entire European Community (EC) proposed a management program for all Alkaline, Mercury, Ni-Cad, and Lead-Acid batteries that are manufactured, bought, or sold within EC countries. Their proposal requires that these batteries display a recycling symbol, and that they be subject to recycling and/or special disposal. The proposal also mandates the start of a consumer information program on battery composition and disposal.

Compared with its North American counterpart, the European battery industry is in general more closely monitored. Faced with the possibility of an increase in regulation, European manufacturers have in some cases begun voluntary collections programs. It is important to note that these collection programs do not focus on reclaiming battery components; rather, their goal is to insure that batteries are disposed of as hazardous waste. This is particularly important for the many European countries that incinerate large portions of their waste. Similar voluntary recovery programs also exist in Japan.

A study has been done in Canada on the feasibility of separating and recycling batteries as a means for reducing the amount of heavy metals in the environment. Rather than emphasizing collection and separation schemes for used batteries, the recommendations of the study centered on improving technology in battery design in order to reduce the levels of heavy metals used. It also advocated the adoption of wet gas scrubbers in all municipal incinerators (Feasibility Study of the Recycling of Used Dry Cells in Canada, March, 1989).

Before examining possible changes in policy, it is worthwhile to speculate on how technology and market mechanisms alone could produce change. In the past, changes in technology produced the

alkaline battery - a product that has virtually replaced the zinc-carbon cell. Originally these cells contained high amounts of mercury, yet technological changes and market forces have resulted in the development of an alkaline cell that contains less than 0.025% mercury. In addition, new technology could appear that would make the alkaline battery obsolete. Any new policy will impose costs and benefits on various interest groups, as well as introduce various inefficiencies into the market.

4.0 RISK ASSESSMENT OF BATTERY DISPOSAL ALTERNATIVES

Heavy metals can have detrimental effects on ecosystems. It is apparent that, ecosystems have naturally occurring levels of heavy metals; however, additional amounts of such metals can stress the delicate balance necessary for the efficient performance and/or survival of the ecological community. Household battery usage and disposal practices currently in place for used dry-cell batteries results in some additional metal loadings on the environment, with potentials for detrimental effects. This section examines the risks associated with various disposal options for used dry-cell batteries that becomes part of household waste streams. The concepts of risk assessment are employed in evaluating and comparing the feasible disposal options identified for the management of used household batteries.

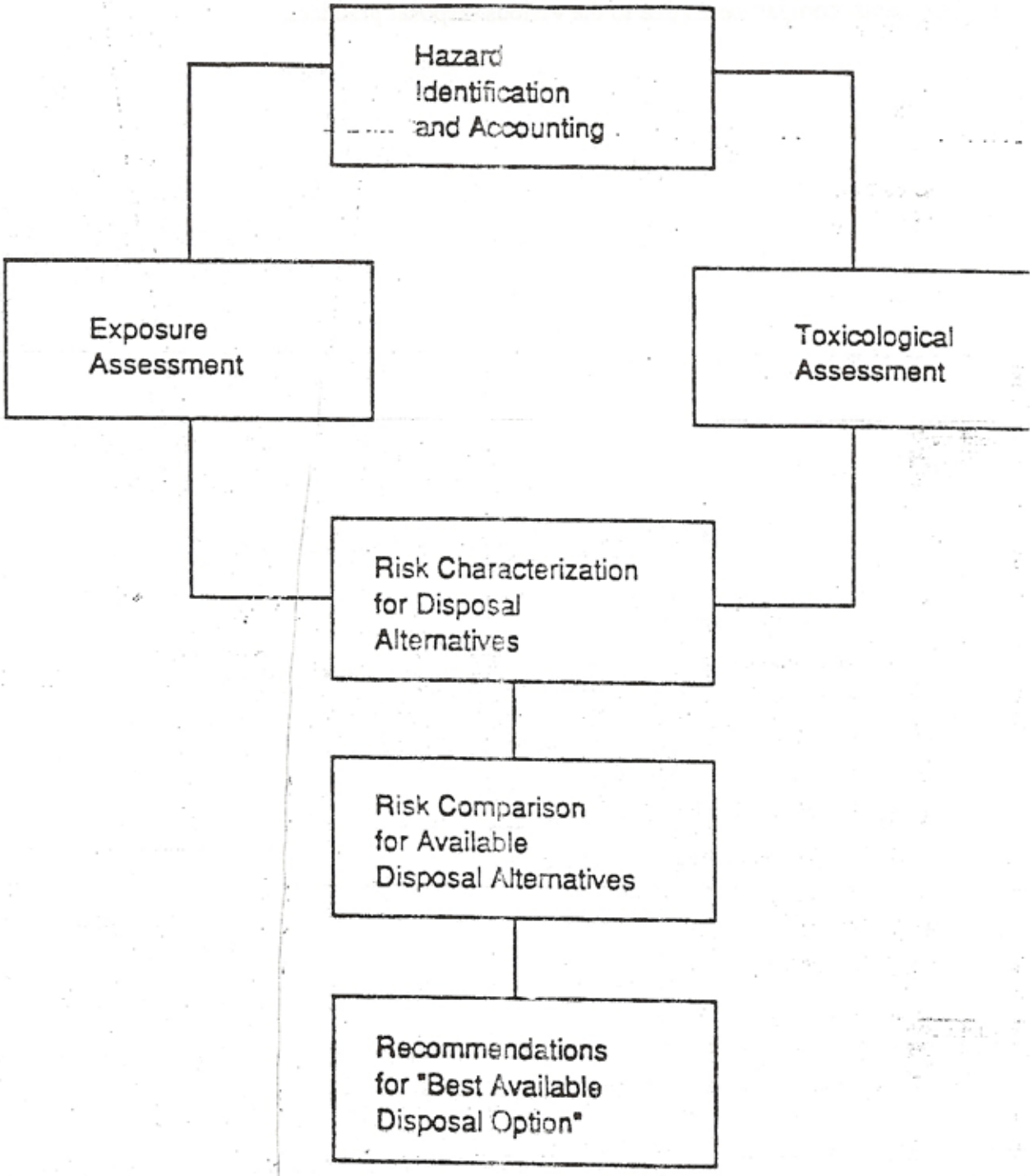
4.1 The Risk Assessment Methodology and Case Model Description

The risk assessment process consists of several steps (Figure 5). Hazard identification is the qualitative evaluation of the potential adverse health/environmental impacts of a chemical on potential receptors. The exposure assessment phase of the risk assessment includes the identification of significant migration pathways and potential receptors, the development of exposure scenarios (including the analysis of environmental fate and persistence), the estimation/modeling of exposure point concentrations, and the estimation of chemical intakes for all potential receptors and significant pathways of concern. Exposure assessment is a process used to estimate the rates at which chemicals are absorbed by organisms through all mechanisms--including, ingestion, inhalation, and by dermal absorption. The absorbed dose differs significantly from the externally applied dose, called exposure or intake. The exposure estimates are used to determine whether any threats exist based on existing exposure conditions from a potentially contaminated media. The exposure assessment of a hazardous chemical involves a characterization of the major pathways of contaminant transport leading from the sources to the points of exposure. Following a risk assessment, risk management plans can be developed to help reduce existing or anticipated risks from a given action or activity.

A conservative assumption is made in this evaluation, that all batteries purchased in a given year enter MSW in that year. Since lesser quantities than this will in practice be disposed of, estimated quantities of heavy metals purported to enter the MSW and which is used in characterizing risks for used battery disposal options will represent over-estimates that will be more protective of human health and the environment. This helps to build a factor of safety that can counteract uncertainties potentially present in the analytical processes. Data on total sales (conservatively assumed to equal

total amount of disposal) of dry-cell batteries in Canada are used to indicate the potential risks of the battery disposal alternatives. The risk assessment model utilized in this evaluation incorporates methods for estimating the amounts of the metals of concern potentially entering the various environmental compartments due to the various disposal practices.

Figure 5
Steps in the Risk Assessment



Hazard Identification and Accounting for Dry-Cell Battery Disposal:

The metals of concern in the battery types being investigated in this study may be sources of potential risks to humans and the environment. The potential risks are the results of potential releases of the metals into the environment following disposal of used batteries in the MSW stream. In general, exposure to certain levels of the metals of concern in household batteries can result in acute and/or chronic health effects. Acute effects may be due to short-term exposures to large doses of the toxic metals, while chronic effects are due to long-term exposures to low doses. Cd and Hg are recognized as the critical metals of concern. Selected properties of the chemicals of concern in household batteries that affect the hazard potential of the disposal practices are presented in Appendix B (Tables B1 and B2).

Estimation of Potential Receptor Intakes and Doses in the Exposure Assessment:

Different exposures may result due to the method of choice in used household battery disposal practices. The generic equation for calculating chemical intakes is given by (USEPA, 1989a):

$$I = C \times CR \times CF \times FI \times ABS_S \times EF \times ED \times 1/BW \times 1/AT \quad (1)$$

where:

I	=	intake, adjusted for absorption (mg/kg-day)
C	=	chemical concentration in media of concern (e.g., mg/kg)
CR	=	contact rate (e.g., mg soil/day; liters water/day)
CF	=	conversion factor (to cater for consistency in units used)
FI	=	fraction of intake from contaminated source (unitless)
ABS_S	=	bioavailability/absorption factor (%)
EF	=	exposure frequency (days/years)
ED	=	exposure duration (years)
BW	=	body weight (kg)
AT	=	averaging time (period over which exposure is averaged -- days).

This equation is adapted to estimate actual exposure intakes by potential receptors as a result of releases of metals into various environmental media from the battery disposal practices.

A conservative mass balance approach is additionally employed in this evaluation, according to the following relationship:

$$I = (\text{Mass/year}) \times (1/BW) \times (CF) \quad (2)$$

where mass/year refers to the amount of chemical available per year to potential receptors and all other terms are same as defined above. This equation is used to estimate potential intakes by receptors potentially impacted from the method of choice for the disposal of used household batteries.

Toxicological Evaluation of the Chemicals of Concern:

A toxicity assessment is conducted as part of a risk assessment to quantitatively and qualitatively assess the potential for adverse human health and environmental effects from exposure to the chemicals of potential concern. Typical and important potential health and environmental effects of the metals of concern used in the dry-cell batteries are presented in Table 14. A more elaborate discussion of the metals' health and environmental effects is further presented in Appendix C of this report.

Table 14
Potential Health and Environmental Effects of the Metals of Concern in Dry-Cell Batteries

Metal	Typical Health Effects and Toxic Manifestations
Cadmium (Cd)	Developmental disabilities, Kidney damage, Pulmonary edema, Lung cancer, Neonatal death, Bioaccumulation in aquatic organisms
Manganese (Mn)	Bronchitis, Influenza (metal fume fever), Liver cirrhosis, Pneumonia
Mercury (Hg)	Ataxic gait, CNS Symptoms, Developmental disabilities, Minamata disease, Kidney and Liver damage, Bronchitis, Gingivitis; Bioaccumulation of methyl-mercury
Nickel (Ni)	Asthma, CNS effects, Gastrointestinal effects, Lung and nasal Cancer, Dermatitis
Zinc (Zn)	Corneal ulceration, Esophagus damage, Pulmonary edema

The quantitative portion of the toxicity assessment entails identifying the relevant toxicity indices against which exposure point intakes and doses can be compared during the risk characterization stage of the overall assessment. This aspect of the evaluation consists of the determination of appropriate toxicity indices (such as acceptable daily intakes (ADIs), reference doses (RfDs), and slope factors (SF), or cancer potency factors (CPFs)). The qualitative aspect of the assessment may include summaries of the adverse human health effects, typical environmental levels or background concentrations, toxicokinetics, toxicodynamics, and ecotoxicology associated with each chemical of potential concern.

Risk Characterization and Consequence Determination:

Risk characterization is the final step in the risk assessment process and the first input to the risk management process. Its purpose is to present the risk manager with a synopsis and synthesis of all the data that should contribute to a conclusion with regards to the nature and extent of the risk. In the risk characterization, the exposure and toxicity assessments are used to characterize risk levels, both quantitatively and qualitatively. The risks to potentially exposed populations from exposure and subsequent intake of the chemicals of potential concern are characterized by the calculation of noncarcinogenic hazard quotients and indices and/or carcinogenic risks. These parameters are then compared with applicable standards for risk decisions.

For potential carcinogens, risks are estimated as probabilities. The carcinogenic risks may be estimated by multiplying the cancer slope factor (SF), which is the upper 95% confidence limit of the probability of a carcinogenic response per unit intake over a lifetime of exposure, by the estimated intakes–yielding incremental risk values. The carcinogenic effects of the chemicals of concern are calculated according to the following relationship (USEPA, 1989a):

$$\text{Risk, CR} = \text{CDI} \times \text{SF} \qquad (3a)$$

where

CR = probability of an individual developing cancer (dimensionless)
CDI = chronic daily intake averaged over 70 years (mg/kg-day)
SF = slope factor (1/[mg/kg-day]).

This represents the linear low-dose cancer risk model, and is valid only at low risk levels (i.e., below estimated risks of 0.01). The aggregate cancer risk equation for multiple chemicals is subsequently obtained by summing the risks calculated for the individual chemicals using the above relationship(s). Thus, for multiple compounds,

$$\text{Total Risk} = \sum_i (CDI_i \times SF_i) \qquad (3b)$$

CDI_i = chronic daily intake for the i^{th} contaminant

SF_i = slope factor for the i^{th} contaminant.

or multiple compounds and multiple pathways, the overall total cancer risk for all exposure pathways and all contaminants considered in the risk evaluation will be:

$$\text{Overall Total Risk} = \sum_i \sum_j (CDI_{ij} \times SF_{ij}) \qquad (3c)$$

where

CDI_{ij} = chronic daily intake for the i^{th} contaminant and j^{th} pathway

SF_{ij} = slope factor for the i^{th} contaminant and j^{th} pathway.

The CDIs are estimated from the equations given previously for chemical intakes, whereas the SF values are obtained from various sources/databases, including the Integrated Risk Information System (IRIS) and the Health Effects Assessment Summary Tables (HEAST) available through the U.S. EPA (Environmental Protection Agency). As a rule-of-thumb, incremental risks of between 10^{-4} and 10^{-7} are generally perceived as acceptable levels for the protection of human health and the environment.

The overall potential noncarcinogenic effects posed by the chemicals of concern is usually expressed by the Hazard Index (HI). The noncarcinogenic effects of the chemicals of concern are calculated according to the following relationship (USEPA, 1989a):

$$\text{Hazard Quotient, HQ} = E / RfD \qquad (4a)$$

where

E = chemical exposure level or intake (mg/kg-day)

RfD = reference dose (mg/kg-day).

The sum total of the hazard quotients for all the chemicals of concern gives the hazard index for a given exposure pathway. The applicable relationship is:

$$\text{Total Hazard Index, HI} = \sum_i E_i / RfD_i \qquad (4b)$$

where

E_i = exposure level (or intake) for the i^{th} contaminant

RfD_i = acceptable intake level (or reference dose) for i^{th} contaminant.

For multiple compounds and multiple pathways, the overall total noncancer risk for all exposure pathways and all contaminants considered in the risk evaluation will be:

$$\text{Overall Total Hazard Index} = \sum_i \sum_j E_{ij} / RfD_{ij} \qquad (4c)$$

where

E_{ij} = exposure level (or intake) for the i^{th} contaminant and j^{th} pathway

RfD_{ij} = acceptable intake level (or reference dose) for i^{th} contaminant and j^{th} pathway.

The E values are estimated from the equations given previously for chemical intakes, whereas the RfD values are obtained from databases such as IRIS and HEAST available through the U.S. EPA. RfDs have been established by the U.S. EPA as thresholds of exposure to toxic substances below which no adverse or insignificant health impact is anticipated; it is the suggested maximum amount (in mg) of the chemical of concern for every kg of a person's body weight that an individual can be exposed to daily without noticeable health impacts. These thresholds have been established on a substance-specific basis for oral and inhalation exposures, taking into account evidence from both human epidemiologic and laboratory toxicologic studies. In accordance with the U.S. EPA guidelines on the interpretation of hazard indices, for any given chemical, there may be potential for adverse health effects if the hazard index exceeds unity (i.e. HI > 1). The "acceptable level" itself (i.e., the RfD) incorporates a large margin of safety (of a factor of between 100 and 1,000), so that it is possible that no toxic effects may occur even if the "acceptable level" is exceeded. However, in interpreting the hazard index (HI) values, a reference value of HI less than or equal to 1 should be taken as the acceptable reference or standard. For HI values greater than unity (i.e. HI > 1), the higher the value, the greater is the likelihood of adverse health impacts.

Case Models Description:

Case simulations are used to demonstrate the potential effects of potential receptor exposures to metals of concern contributed by dry-cells to MSW. Three different case scenarios are evaluated, including the following:

- Landfilling disposal only for MSW (Case Model 1)
- Incineration disposal only for MSW (Case Model 2)
- Combined landfilling and incineration disposal for MSW (Case Model 3)

The model utilizes specific information pertaining to the following:

n results that landfilling even concentrated volumes of the used household batteries
significant health and environmental impacts, since the hazard indices are <1 for all

- Total annual amount of batteries entering MSW streams
- Total annual amount from batteries going to landfill and/or incinerator used for MSW management

Additional information necessary for a complete evaluation are identified in the subsequent sections.

4.2 Risk Characterization of the Landfill Disposal Alternative for Household Batteries

Landfilling of used household batteries together with MSW is a popular disposal option for dry-cell batteries. The landfilling of used batteries leads to consequential decomposition of the batteries. Subsequently, the metals of concern present in the batteries are released into leachates generated at the landfills. The migration of such leachates into underlying aquifers can result in the contamination of potential water supply sources.

Additional information on exposure modeling parameters for the landfilling option necessary for a complete evaluation include the following:

- For leachate generated, it is assumed that = 0.05% of the metals from batteries entering a landfill will escape in the form of leachate to contaminate groundwater. This represents a conservative estimate, compared with typical transfer coefficients (Table 7) for leachate in MSW landfills.
- It is assumed that there is ingestion intake from potentially contaminated groundwater, following the migration of leachate into a water supply aquifer.

A complete listing of the exposure modeling assumptions are included in Table 15, together with the estimates of potential receptor exposures based on equation (2). A summary of the risk characterization computations is given in Table 16; this is based on results from Table 15 and equations (3) and (4). Hazard index values of 0.4 and 0.1 were estimated for the child and adult populations, respectively. No carcinogenic risks are anticipated, since none of the chemicals is classified as a carcinogen by the oral exposure route. Thus, *it is apparent from the risk characterization results that landfilling even concentrated volumes of the used household batteries will present no significant health and environmental impacts*, since the hazard indices are <1 for all potential receptors.

TABLE 15

EXPOSURE MODELING AND EVALUATION RESULTS
CASE MODEL I--LANDFILLING DISPOSAL ONLY FOR MSW

Modeling Assumptions:

1) All MSW generated in region are disposed of at municipal landfills.
2) Conservatively assumed that up to 0.05% of all landfill waste will become leachate annually.
3) Populations potentially impacted chosen to be the size of residential population in case region of landfill location; population of case region is taken at 100,000 and that for the Province of Ontario approximated to 10 million.
4) Child average weight is 16kg, and adult average weight is 70kg; these potential receptors may be impacted via ingestion of contaminated groundwater.
5) Contribution of metals (in household batteries) to MSW in case region is proportional to the population.

Battery Constituent	Total Annual Amount from Batteries into MSW Streams (National--million gm)	Total Annual Amount from Batteries into MSW Streams (Ontario--gm)	Total Annual Amount from Batteries into MSW Streams (Case Region 1--gm)	Total Annual Amount from Batteries into Landfill (Case Region 1--mg)	Leachate	Potential Ingestion Intake (mg/person/yr)	Potential Receptor Expos Via Ingestion of Leache Releases into Ground Wa CHILD | | ADULT (mg/kg/day)	
Cadmium (Cd)	4.73E+01	1.70E+07	1.70E+05	1.70E+08	8.52E+04	8.52E-01	1.46E-04	3.33E-0
Manganese (Mn)	1.61E+03	5.80E+08	5.80E+06	5.80E+09	2.90E+06	2.90E+01	4.97E-03	1.13E-0
Mercury (Hg)	1.06E+00	3.80E+05	3.80E+03	3.80E+06	1.90E+03	1.90E-02	3.25E-06	7.44E-0
Nickel (Ni)	5.07E+01	1.83E+07	1.83E+05	1.83E+08	9.13E+04	9.13E-01	1.56E-04	3.57E-0
Zinc (Zn)	8.09E+02	2.91E+08	2.91E+06	2.91E+09	1.46E+06	1.46E+01	2.49E-03	5.70E-0

TABLE 16

RISK CHARACTERIZATION
CASE MODEL I–LANDFILLING DISPOSAL ONLY FOR MSW

Baterry Constituent	Potential Ingestion Intake (mg/person/yr)	Potential Receptor Exposure Via Ingestion of Leachate Releases Into Ground Water CHILD (mg/kg/day)	ADULT (mg/kg/day)	Oral RfD (mg/kg/day)	Oral SF (1/[mg/kg/day])	Hazard Index for Child Exposure	Hazard Index for Adult Exposure	Carcinogenic Risks for Child Exposure	Carcinogenic Risks for Adult Exposure
Cadmium (Cd)*	8.52E-01	1.46E-04	3.33E-05	5.00E-04		0.29	0.07	0.00E+00	0.00E+00
Manganese (Mn)	2.90E+01	4.97E-03	1.13E-03	1.00E-01		0.05	0.01	0.00E+00	0.00E+00
Mercury (Hg)	1.90E-02	3.25E-06	7.44E-07	3.00E-04		0.01	0.00	0.00E+00	0.00E+00
Nickel (Ni)	9.13E-01	1.56E-04	3.57E-05	2.00E-02		0.01	0.00	0.00E+00	0.00E+00
Zinc (Zn)	1.46E+01	2.49E-03	5.70E-04	2.00E-01		0.01	0.00	0.00E+00	0.00E+00
TOTAL						0.4	0.1	0.00E+00	0.00E+00

*Cadmium is a B1 carcinogen (i.e., a probable human carcinogen) by the inhalation pathway only.
All the other chemicals of concern are considered to be noncarcinogens.

Disposal Practices in the Regional Municipality of Waterloo as an Approximate Example:

To validate results obtained from the case model, field information is used to confirm the conclusion reached. A landfill site is chosen in the Regional Municipality of Waterloo. The Waterloo landfill site is located in the extreme southwest corner of the City of Waterloo, Regional Municipality of Waterloo, Ontario. The area is underlain by a major sand and gravel aquifer which supplies about 45 million gallons per day (mgpd) of groundwater to the Kitchener-Waterloo area. In fact, this is one of the most significant groundwater resources in Ontario.

Over 100 monitoring wells have been installed at, and in the vicinity of the landfill since 1971. Table 17 shows a summary for typical monitoring information associated with operation of the Waterloo landfill site area. None of the monitoring data for the upper unit aquifer or the aquifer unit No. 1 exceeds the MOE objective. Nonetheless, the higher of these contaminant levels (worst-case scenario) is used to estimate the level of potential risks posed by aquifer contamination from all sources of the metals of concern. It should be noted that the dry-cell batteries of concern are contributing only a fraction of this total amount of metals monitored as present in the aquifer; for instance, Hg and Cd from dry-cell batteries contribute only about 20% and 33%, respectively, to the quantities appearing in MSW. Tables 18 and 19 respectively, show the potential receptor exposures based on equation (1) and the risk characterization based on equation (3) and (4) for the impacts of the Waterloo landfill on underlying aquifers.

The hazard index for the adult population is 0.7 (i.e., <1), indicating no problem. On the other hand, child exposure shows a hazard index >1. However, this value of 1.6 is not very much above the acceptable index of unity. Considering the conservative assumptions used in the evaluation, it may be concluded that the landfill disposal of household batteries are not posing any risks due to the presence of any of the metals of concerns identified for these batteries. No carcinogenic risks are anticipated, since non of the chemicals is a carcinogen by the oral exposure route.

Thus, it is apparent from the risk characterization results that, landfilling MSW of which used household batteries may be a part is presenting no significant health and environmental impacts. It is also worth mentioning that these levels of metals present in the aquifer would have been contributed from all sources other than dry-cell batteries only.

Table 17
Representative Maximum Concentrations (mg/L) of the Chemicals of Concern in Leachate and Groundwater Near the Waterloo Landfill Site in the Regional Municipality of Waterloo

Parameter	Upper Fine-Grained Unit Aquifer	Aquifer Unit No. 1	Typical Leachate Quality	Worst-Case Leachate Quality	Collector-Case Leachate Quality	MOE Objective
Cadmium (Cd)	<0.01	<0.01	0.024	0.065	0.013	0.005
Manganese (Mn)	0.01	0.18	12	39	–	0.05
Mercury (Hg)	<0.00001	0.00003	–	–	–	0.001
Nickel (ni)	<0.01	<0.01	0.23	7.0	0.43	–
Zinc (Zn)	0.35	0.55	6.5	50	2.60	5.0

Notes: • Collector-case characterization of leachate quality is based on average value from leachate pumping station.
• Worst-case leachate quality represents the maximum contaminant level from leachate generated from landfill.
• MOE objective is maximum acceptable contaminant levels.

Sources: 1) CRA, 1990. Waterloo Landfill Site Hydrogeologic Investigation (Final Report, June 1990). Prepared for the Regional Municipality of Waterloo.
2) CRA, 1990. Waterloo Landfill Site Progress Report. Report to the MOE, Hamilton.

Parameter	Upper Fine-Grained Unit Aquifer	Aquifer Unit No. 1	Typical Leachate Quality	Worst-Case Leachate Quality	Collector-Case Leachate Quality	MOE Objective
Cadmium (Cd)	<0.01	<0.01	0.024	0.065	0.013	0.005
Manganese (Mn)	0.01	0.18	12	39	–	0.05
Mercury (Hg)	<0.00001	0.00003	–	–	–	0.001
Nickel (ni)	<0.01	<0.01	0.23	7.0	0.43	–
Zinc (Zn)	0.35	0.55	6.5	50	2.60	5.0

pumping station.
Worst-case leachate quality represents the maximum contaminan
generated from landfill.
MOE objective is maximum acceptable contaminant levels.

1990. Waterloo Landfill Site Hydrogeologic Investigation (
Prepared for the Regional Municipality of Waterloo.
1990. Waterloo Landfill Site Progress Report. Report to th

TABLE 18

EXPOSURE ASSESSMENT
CHRONIC DAILY INTAKES FROM LANDFILL RELEASES
WATERLOO LANDFILL FACILTY, WATERLOO, ONTARIO

Modeling Assumptions:

1) Potential receptors are potentially exposed via ingestion of contaminated groundwater.
2) Child average weight is 16kg, and adult average weight is 70kg.
   These potential receptors may be impacted via ingestion of groundwater potentially impacted by landfill leachate.
3) Ingestion rates assumed to be 2 liters/day and 1 liter/day for adult and child respectively.
   chemical bioavailability factors for ingested water is conservatively taken to be 100%, and water is taken for 365 days/yr.

Chemical of Concern	Absorption Factor (%)	Exposure Concentration*	Chronic Daily Intake (CDI)–CHILD (mg/kg-day)		Chronic Daily Intake (CDI)–ADULT (mg/kg-day)	
			Carcinogenic Effects	Noncarcinogenic Effects	Carcinogenic Effects	Noncarcinogenic Effects
Cadmium (Cd)	100	1.00E-02	-	6.25E-04	-	2.86E-04
Manganese (Mn)	100	1.80E-01	-	1.13E-02	-	5.15E-03
Mercury (Hg)	100	3.00E-05	-	1.88E-06	-	8.58E-07
Nickel (Ni)	100	1.00E-02	-	6.25E-04	-	2.86E-04
Zinc (Zn)	100	5.50E-01	-	3.44E-02	-	1.57E-02

* Units are mg/l.

TABLE 19

RISK CHARACTERIZATION
FOR LANDFILL LEACHATE RELEASES INTO AQUIFERS
WATERLOO LANDFILL FACILTY, WATERLOO, ONTARIO

Battery Constituent	Potential Receptor Exposure Via Ingestion of Landfill Releases (carcinogenic effects) CHILD | | ADULT (mg/kg/day)	Potential Receptor Exposure Via Ingestion of Landfill Releases (noncarcinogenic effects) CHILD | | ADULT (mg/kg/day)	Oral RfD (mg/kg/day)	Oral SF (1/[mg/kg/day])	Hazard Index for Child Exposure	Hazard Index for Adult Exposure	Carcinogenic Risks for Child Exposure	Carcinogenic Risks for Adult Exposure
Cadmium (Cd)*		6.25E-04 2.66E-04	5.00E-04		1.25	0.57	0.00E+00	0.00E+00
Manganese (Mn)		1.13E-02 5.15E-03	1.00E-01		0.11	0.05	0.00E+00	0.00E+00
Mercury (Hg)		1.88E-06 8.58E-07	3.00E-04		0.01	0.00	0.00E+00	0.00E+00
Nickel (Ni)		6.25E-04 2.66E-04	2.00E-02		0.03	0.01	0.00E+00	0.00E+00
Zinc (Zn)		3.44E-02 1.57E-02	2.00E-01		0.17	0.08	0.00E+00	0.00E+00
TOTAL					1.57	0.71	0.00E+00	0.00E+00

*Cadmium is a B1 carcinogen (i.e., a probable human carcinogen) by the inhalation pathway only.
All the other chemicals of concern are considered to be noncarcinogens.

4.3 Risk Characterization of the Incineration Disposal Alternative for Household Batteries

Incineration of used household batteries generally will result in the release of metals as fumes and particulates. Exposure to such contaminants can consequently occur via direct inhalation by humans, or contamination of other environmental media may occur with subsequent impacts on humans (such as through consumption of contaminated foods and water).

The incineration process yields fly and bottom ash, which both may contain some metals. Based on an assumption of reported figures that 10% of incinerated wastes become fly ash and 90% become bottom ash (Goldstein, 1989), and that fly ash has a smaller unit weight than bottom ash, it is estimated that about 70% by weight of the metals are in the bottom ash and 30% by weight in the fly ash (CMU, 1989). It is further assumed that 99% of the fly ash is captured by the incinerator's air pollution control device, and this part of the fly ash is added to the bottom ash for disposal at a hazardous waste facility.

A complete listing of the exposure modelling assumptions are indicated in Table 20, together with the estimates for potential receptor exposures based on equation (2). A summary of the risk characterization calculations for the incineration option is given in Table 21; this is based on results from Table 20 and equations (3) and (4). Hazard index values of 3.5 and 0.8 were estimated for the child and adult populations, respectively. Also, carcinogenic risks of 5.3 x 10-3 and 1.2 x 10-3 were estimated for the child and adult groups, respectively. *By comparing these numbers with an acceptable hazard index of 1 and an acceptable carcinogenic risk range of 1.0×10^{-4} to 1.0×10^{-7}, it is apparent that, incineration of concentrated amounts of the dry-cell batteries may present both carcinogenic and noncarcinogenic risks to potential receptors. Much of this risk is contributed by cadmium, known to be a probable human carcinogen by the inhalation pathway. This cadmium will be from concentrated sources such as the disposal of Ni-Cads. Removal of large amounts of Ni-Cads from MSW streams to be incinerated will therefore remove the potential risks posed by the incineration alternative for managing the used dry-cell batteries.*

Disposal Practices in the Regional Municipality of Hamilton-Wentworth as an Approximate Example:

To validate the results obtained for the case model presented, field information is used to confirm the conclusion reached for the incineration management of used dry-cell batteries. A solid waste incinerator facility located in Hamilton with a population of about 430,000 is chosen for this purpose.

The Solid Waste Reduction Unit (SWARU) in Hamilton, Ontario, processes and incinerates municipal solid waste (MSW). The facilities are owned by the Region of Hamilton-Wentworth and operated by Laidlaw Waste Systems. Table 22 shows a summary of typical monitoring/modeling data associated with the operation of the SWARU incinerator facility in Hamilton. It should be noted that the dry-cell batteries of concern are contributing only a fraction of the total amount of metals present in wastes incinerated at the facility. Tables 23 and 24 respectively show the potential receptor exposures (based on equation (1)), and the risk characterization (based on equations (3) and (4)) for the potential impacts of the Hamilton SWARU due to atmospheric emissions. *There is no significantly measurable noncarcinogenic risks experienced by potential human receptors, since hazard indices approaching zero values were estimated for both child and adult population. On the other hand, carcinogenic risks of about* 1.6×10^{-7} *and* 1.4×10^{-6} *were estimated for the child and adult groups, respectively. However, these numbers fall within the acceptable carcinogenic risk range of* 1.0×10^{-4} *to* 1.0×10^{-7}*. Hence, it may be concluded that metal emissions from the Hamilton-Wentworth facility presents no risks of concern.*

TABLE 20

EXPOSURE MODELING AND EVALUATION RESULTS
CASE MODEL II--INCINERATION DISPOSAL ONLY FOR MSW

odeling Assumptions:

1) All MSW generated in region are disposed of at municipal incinerators.
2) It is assumed that 10% of incinerator wastes go to fly ash, and 90% into bottom ash; also, 30% by weight of metals become fly ash and 70% by weight become bottom ash.
3) Assumed that Pollution control efficiency of up to 99% attained, so that only 1% of the available fly ash is released through the stack into air
4) All captured fly ash and bottom ash are sent to a hazardous waste landfill elsewhere.
5) Populations potentially impacted chosen to be the size of residential population in case region of incinerator location; population of case region is taken at 100,000 and that for the Province of Ontario approximated to 10 million.
6) Child average weight is 16kg, and adult average weight is 70kg; these potential receptors may be impacted via inhalation of air emissions.
7) Contribution of metals (in household batteries) to MSW in case region is proportional to the population.

Battery onstituent	Total Annual Amount from Batteries into MSW Streams (National million gm)	Total Annual Amount from Batteries into MSW Streams (Ontario gm)	Total Annual Amount from Batteries into MSW Streams (Case Region 2 gm)	Total Annual Amount from Batteries into Incinerator (Case Region 2 mg)	Bottom Ash (mg)	Incinerator Fly Ash Component (mg)	Total Fly Ash Captured by Pollution Control Devices (mg/yr)	Total Fly Ash Released into Air (mg/yr)	Potential Inhalation Intake (mg/ person/yr)	Potential Receptor Exposure Via Inhalation of Incinerator Releases into Air (mg/kg/day)	
										CHILD	ADULT
lmium (Cd)	4.73E+01	1.70E+07	1.70E+05	1.70E+08	1.19E+08	5.11E+07	5.06E+07	5.11E+05	5.11E+00	8.75E-04	2.00E-04
ganese (Mn)	1.61E+03	5.80E+08	5.80E+06	5.80E+09	4.06E+09	1.74E+09	1.72E+09	1.74E+07	1.74E+02	2.98E-02	6.81E-03
cury (Hg)	1.06E+00	3.80E+05	3.80E+03	3.80E+06	2.66E+06	1.14E+06	1.13E+06	1.14E+04	1.14E-01	1.95E-05	4.46E-06
kel (Ni)	5.07E+01	1.83E+07	1.83E+05	1.83E+08	1.28E+08	5.48E+07	5.42E+07	5.48E+05	5.48E+00	9.38E-04	2.14E-04
(Zn)	8.09E+02	2.91E+08	2.91E+06	2.91E+09	2.04E+09	8.74E+08	8.65E+08	8.74E+06	8.74E+01	1.50E-02	3.42E-03

TABLE 21

RISK CHARACTERIZATION
CASE MODEL II–INCINERATION DISPOSAL ONLY FOR MSW

Baterry Constituent	Potential Inhalation Intake (mg/person/yr)	Potential Receptor Exposure Via Inhalation of Incinerator Releases into Air (mg/kg/day) CHILD	ADULT	Inhalation RID (mg/kg/day)	Inhalation SF (1/[mg/kg/day])	Hazard Index for Child Exposure	Hazard Index for Adult Exposure	Carcinogenic Risks for Child Exposure	Carcinogenic Risks for Adult Exposure
Cadmium (Cd)*	5.11E+00	8.75E-04	2.00E-04	5.00E-04	6.10E+00	1.75	0.40	5.34E-03	1.22E-03
Manganese (Mn)	1.74E+02	2.98E-02	6.81E-03	2.00E-01		0.15	0.03	0.00E+00	0.00E+00
Mercury (Hg)	1.14E-01	1.95E-05	4.46E-06	3.00E-04		0.07	0.01	0.00E+00	0.00E+00
Nickel (Ni)	6.48E+00	9.38E-04	2.14E-04	2.00E-02		0.05	0.01	0.00E+00	0.00E+00
Zinc (Zn)	8.74E+01	1.50E-02	3.42E-03	1.00E-02		1.50	0.34	0.00E+00	0.00E+00
TOTAL						3.6	0.8	5.34E-03	1.22E-03

*Cadmium is a B1 carcinogen (i.e., a probable human carcinogen) by the inhalation pathway only.
All the other chemicals of concern are considered to be noncarcinogens.

AVERAGE STACK GAS METALS EMISSION DATA				DISPERSION MODELING RESULTS			
Chemical of Concern	Actual Concentration (mg/m^3)	Emission Rate (mg/s)	Average Baghouse Efficiency for Metal Removal (%)	Emission Rate (mg/s)	Maximum Ground Level Impingement Concentration ($\mu g/m^3$)	Allowable Concentration ($\mu g/m^3$)	Fraction Allowa Concentr (%)
Cadmium	0.011	0.42	98.3	0.84	0.0019	5	0.0
Manganese	0.048	1.92	98.5	3.84	0.0087	30	0.0
Mercury	0.026	1.05	91.6	2.10	0.0048	5	0.1
Nickel	0.028	1.11	88.4	30.44	0.0691	5	1.3
Zinc	0.312	12.43	98.7	24.86	0.0564	100	0.0

Notes: Average baghouse efficiency for particulate removal estimated at about 98%.
Source: "Review of Emission Studies at the SWARU Facilities, Hamilton". A Report to Tricil Ltd., Hamilton, Prepared by ORTE International, Mississauga, August 1990.

EXPOSURE ASSESSMENT
CHRONIC DAILY INTAKES FROM INCINERATOR RELEASES
HAMILTON SWARU FACILTY, HAMILTON, ONTARIO

Modeling Assumptions:

1) Potential receptors are potentially exposed via inhalation of particulates from stack emissions.
2) Child average weight is 16kg, and adult average weight is 70kg;
   these potential receptors may be impacted via inhalation of air emissions.
3) Inhalation rates assumed to be 0.25m3/hr and 0.83m3/hr for child and adult respectively;
   retention rate of inhaled air is conservatively taken to be 100% for 365 days/yr.

Chemical of Concern	Absorption Factor (%)	Exposure Concentration*	Chronic Daily Intake (CDI)--CHILD (mg/kg-day)		Chronic Daily Intake (CDI)--ADULT (mg/kg-day	
			Carcinogenic Effects	Noncarcinogenic Effects	Carcinogenic Effects	Noncarcinogenic Effects
Cadmium (Cd)	100	1.90E-06	2.55E-06	3.57E-07	2.24E-07	2.70E-07
Manganese (Mn)	100	8.70E-06	1.17E-07	1.64E-06	1.03E-06	1.24E-06
Mercury (Hg)	100	4.80E-06	6.43E-08	9.02E-07	5.66E-07	6.82E-07
Nickel (Ni)	100	6.91E-05	9.26E-07	1.30E-05	8.15E-06	9.81E-06
Zinc (Zn)	100	5.64E-05	7.56E-07	1.06E-05	6.66E-06	8.01E-06

* Units are mg/m^3 for Air Respirable Concentrations.

TABLE 24

RISK CHARACTERIZATION
FOR INCINERATOR RELEASES
HAMILTON SWARU FACILTY, HAMILTON, ONTARIO

Battery Constituent	Potential Receptor Exposure Via Inhalation of Incinerator Releases (carcinogenic effects) (mg/kg/day)		Potential Receptor Exposure Via Inhalation of Incinerator Releases (noncarcinogenic effects) (mg/kg/day)		Inhalation RfD (mg/kg/day)	Inhalation SF (1/[mg/kg/day])	Hazard Index for Child Exposure	Hazard Index for Adult Exposure	Carcinogenic Risks for Child Exposure	Carcinogenic Risks for Adult Exposure
	CHILD	ADULT	CHILD	ADULT						
Cadmium (Cd)*	2.55E-08	2.24E-07	3.57E-07	2.70E-07	5.00E-04	6.10E+00	0.00	0.00	1.55E-07	1.37E-06
Manganese (Mn)	1.17E-07	1.03E-06	1.64E-06	1.24E-06	2.00E-01		0.00	0.00	0.00E+00	0.00E+00
Mercury (Hg)	6.43E-08	5.66E-07	9.02E-07	6.82E-07	3.00E-04		0.00	0.00	0.00E+00	0.00E+00
Nickel (Ni)	9.26E-07	8.15E-06	1.30E-05	9.81E-06	2.00E-02		0.00	0.00	0.00E+00	0.00E+00
Zinc (Zn)	7.56E-07	6.66E-06	1.06E-05	8.01E-06	1.00E-02		0.00	0.00	0.00E+00	0.00E+00
TOTAL							0.0	0.0	1.55E-07	1.37E-06

*Cadmium is a B1 carcinogen (i.e., a probable human carcinogen) by the inhalation pathway only.

4.4 Risk Characterization of the Combined Landfilling and Incineration Disposal Option for Household Batteries

Part of dry-cell batteries present in MSW may be landfilled, while the remaining is incinerated. Tables 25 and 26 respectively show the exposure assessment (based on equation (2)) and risk characterization (based on Table 25 and equations (3) and (4)) of a management option that involves both the incineration and landfilling of used household batteries. *Hazard index values of 1.9 and 0.4 were estimated for the child and adult populations respectively. Also, carcinogenic risks of* 2.7×10^{-3} *and* 5.1×10^{-4} *were estimated for the child and adult groups, respectively. By comparing these numbers with an acceptable hazard index of 1 and an acceptable carcinogenic risk range of* 1.0×10^{-4} *to* 1.0×10^{-7}*, it is noted that risks of concern are presented, due to cadmium (from Ni-Cads) being incinerated. In reality, these metals will appear in diluted forms due to mixing with large volumes of MSW. Concentrated amounts of Ni-Cads are less likely to be found. This means that the levels of risks estimated here will actually be lower. Nonetheless, it is recommended that Ni-Cads be landfilled or recycled if in concentrated forms.*

Test analyses of ash removed from SWARU are presented in Table 27.While bottom ash loss was found to meet Regulation 309 criteria; fly ash samples failed to meet the criteria because of high cadmium levels. As a result, the fly ash from SWARU is no longer co-mingled with bottom ash and is instead disposed separately in an approved hazardous waste landfill site near Sarnia.

4.5 Risk Characterization of Recycling as a Management Alternative for Household Batteries: Risks Related to Separate Collection

Ingestions, fires and explosions, and environmental problems are some of the negative consequences associated with the separate collection, storage and recycling of household batteries. Although the risks associated with dry-cell battery recycling are not quantified, a qualitative evaluation of such risks is presented below.

Ingestions:

To characterize the magnitude of this problem in the U.S. the number of battery ingestions was compared to ingestions of other objects (CMU, 1989). Data on ingestions of coins, Christmas ornaments, and toys, compiled by the American Association of Poison Control Centers (AAPCP) was compared to battery ingestions (alkaline cells and button cells). The number of battery-related ingestions was on the same order of magnitude but slightly less than that of the other groups. Battery ingestions made up nearly one-third of the total ingestions of the compared groups for the years 1986-1988. Of ingestions of all types, battery related ingestions represented only 0.3%.

Data on the severity of health consequences of battery ingestions revealed that only 0.12% of outcomes were life threatening, 1.71% were self-limiting (patients are fine; however, the patients place restrictions on themselves due to their own perceptions of their symptoms), while 30.60% have short-lived self-limiting symptoms and 67.57% have no symptoms.

NEMA (1988) reports that 1,065 batteries were accidently ingested between 1982 to 1987 in the U.S. causing the death of two young children as a result of internal chemical burns (0.02 % of the ingested batteries proved fatal) (MPCA, 1990).

There is no data to suggest that battery collection programs increase the risk of ingestions.

Explosion/Fire:

Placing large quantities of batteries together in containers could possibly increase the risk of fire or explosion, particularly if the batteries are not fully discharged. Research has shown that a significant number of discarded batteries do have residual voltages and can still supply current. (CMU, 1989). When charged batteries contact, they begin to discharge. This generates heat and hydrogen gas, creating potentially hazardous conditions in the home or to collection and disposal workers if the containers are not ventilated.

NEMA (1988) reports one instance in which a sealed barrel of waste batteries exploded, causing minor injuries to the person unloading the barrel. This was not a barrel of batteries from a household collection program. In addition, used dry-cell batteries kept in a separate container can continue to discharge, leading to leakage which can cause skin irritation if handled – a threat to the home and workplace.

Environmental Impacts:

Additional risks resulting from reclamation processes must be taken into consideration, including the possibilities of air and ground pollution.

The Mercury Refining Company (MERECO) in Albany, N.Y. has past and current mercury contamination problems. Prior to 1989, MERECO had been operating with antiquated equipment and without modern air pollution control devices. The New York Department of Environmental Conservation ordered MERECO to install a new burner and air pollution control equipment, improve operations at the facility, minimize surface runoff and enclose the facility in one building after finding that considerable amounts of mercury had migrated off-site from the plant grounds. A second study following a September 1989 fire at the facility concluded that severe contamination of soil and creek sediments had occurred. It was concluded that mercury contamination at and near the

MEREGO facility was due to teflon stack emissions and on-site dust, as well as the fire (Reutlinger, 1990).

TABLE 25

EXPOSURE MODELING AND EVALUATION RESULTS
CASE MODEL III–COMBINED LANDFILLING AND INCINERATION DISPOSAL FOR MSW

Modeling Assumptions:

1) All MSW generated in region are disposed of at municipal landfills and incinerators--50% to each.
2) Conservatively assumed that up to 0.05% of all landfill waste will become leachate annually.
3) It is assumed that 10% of incinerator wastes go to fly ash, and 90% into bottom ash; also, 30% by weight of metals become fly ash and 70% by weight become bottom ash.
4) All captured fly ash and bottom ash are sent to a hazardous waste landfill elsewhere.
5) Populations potentially impacted chosen to be the size of residential population in case region of incinerator location; population of case region is taken at 100,000 and that for the Province of Ontario approximated to 10 million.
6) Child average weight is 15kg, and adult average weight is 70kg; these potential receptors may be impacted via ingestion of contaminated groundwater and/or via the inhalation of air emissions.
7) Contribution of metals (in household batteries) to MSW in case region is proportional to the population.

Battery Constituent	Total Annual Amount from Batteries into MSW Streams (National million gm)	Total Annual Amount from Batteries into MSW Streams (Ontario-gm)	Total Annual Amount from Batteries into MSW Streams (Case Region 3 gm)	Total Annual Amount from Batteries into Landfill (Case Region 3 mg)	Leachate	Potential Ingestion Intake (mg/person/yr)	Potential Receptor Exposure Via Ingestion of Leachate Release into Ground Water CHILD | | ADULT (mg/kg/day)		Total Annual Amount from Batteries into Incinerator (Case Region 3 mg)	Bottom Ash (mg)	Incinerator Fly Ash Component (mg)	Total Fly Ash Captured by Pollution Control Devices (mg/yr)	Total Fly Ash Released into Air (mg/yr)	Potential Inhalation Intake (mg/person/yr)	Potential Receptor Exposure Via Inhalation of Incinerator Release into Air CHILD | | ADULT (mg/kg/dpy)	
Cadmium (Cd)	4.73E+01	1.70E+07	1.70E+06	8.52E+07	4.26E+04	4.26E-01	7.29E-05	1.07E-05	8.52E+07	5.96E+07	2.56E+07	2.53E+07	2.56E+06	2.56E+00	4.38E-04	1.00E-04
Manganese (Mn)	1.61E+03	5.80E+08	5.80E+06	2.90E+09	1.45E+06	1.45E+01	2.48E-03	5.67E-04	2.90E+09	2.03E+09	8.70E+08	8.61E+08	8.70E+06	8.70E+01	1.49E-02	3.40E-03
Mercury (Hg)	1.00E+00	3.60E+05	3.60E+03	1.80E+06	9.60E+02	9.60E-03	1.63E-06	3.72E-07	1.80E+06	1.33E+06	5.70E+05	5.64E+05	5.70E+03	5.70E-02	9.76E-06	2.23E-06
Nickel (Ni)	8.07E+01	1.93E+07	1.93E+06	9.13E+07	4.56E+04	4.56E-01	7.81E-05	1.78E-05	9.13E+07	6.39E+07	2.74E+07	2.71E+07	2.74E+06	2.74E+00	4.69E-04	1.07E-04
Zinc (Zn)	8.09E+02	2.91E+08	2.91E+06	1.46E+09	7.28E+06	7.28E+00	1.25E-03	2.85E-04	1.46E+09	1.02E+09	4.37E+08	4.32E+08	4.37E+06	4.37E+01	7.48E-03	1.71E-03

TABLE 26

RISK CHARACTERIZATION
CASE MODEL III–COMBINED LANDFILLING AND INCINERATION DISPOSAL FOR MSW

Battery Constituent	Potential Ingestion Intake (mg person/yr)	Potential Receptor Exposure Via Ingestion of Leachate Releases Into Ground Water CHILD (mg/kg/day)	ADULT (mg/kg/day)	Potential Inhalation Intake (mg/ person/yr)	Potential Receptor Exposure Via Inhalation of Incinerator Releases Into Air CHILD (mg/kg/day)	ADULT (mg/kg/day)	Oral RfD (mg/ kg/day)	Oral SF (1/[mg/ kg/day])	Inhalation RfD (mg/ kg/day)	Inhalation SF (1/[mg/ kg/day])	Hazard Index for Child Exposure	Hazard Index for Adult Exposure	Carcinogenic Risks for Child Exposure	Carcinogenic Risks for Adult Exposure
Cadmium (Cd)*	4.26E-01	7.29E-06	1.67E-06	2.56E+00	4.38E-04	1.00E-04	5.00E-04		5.00E-04	6.10E+00	1.02	0.23	2.67E-03	6.10E-04
Manganese (Mn)	1.45E+01	2.48E-03	5.67E-04	8.70E+01	1.49E-02	3.40E-03	1.00E-01		2.00E-01		0.10	0.02	0.00E+00	0.00E+00
Mercury (Hg)	8.50E-03	1.63E-06	3.72E-07	5.70E-02	9.76E-06	2.23E-06	3.00E-04		3.00E-04		0.04	0.01	0.00E+00	0.00E+00
Nickel (Ni)	4.56E-01	7.81E-05	1.79E-05	2.74E+00	4.69E-04	1.07E-04	2.00E-02		2.00E-02		0.03	0.01	0.00E+00	0.00E+00
Zinc (Zn)	7.28E+00	1.25E-03	2.85E-04	4.37E+01	7.48E-03	1.71E-03	2.00E-01		1.00E-02		0.75	0.17	0.00E+00	0.00E+00
TOTAL											1.9	0.4	2.67E-03	6.10E-04

*Cadmium is a B1 carcinogen (i.e., a probable human carcinogen) by the inhalation pathway only.
All the other chemicals of concern are considered to be noncarcinogens.

Table 27

SWARU Hamilton

Summary of Regulation 309 Analytical Results on Fly and Bottom Ash Sample - August 1990

Contaminent	Minium Detection Limits	Units	Bottom Ash	Fly Ash Sample 1	Fly Ash Sample 2	Fly Ash Sample 3	Toxic Concentration*
Arsenic	1	mg/L	4	140	<	<	5
Barium	.001	mg/L	1.3	.79	.54	.39	100
Cadmium	.002	mg/L	.003	1.4	1.6	2.4	.5
Chromium	.004	mg/L	.028	.012	.014	.009	5
Lead	.02	mg/L	<	4.9	1.5	2.3	5
Mercury	.2	mg/L	<	3.1	<	.79	.1
Selenium	1	mg/L	5	15	<	<	5
Silver	.01	mg/L	<	<	<	<	5

* Toxic Concentration - as described by Reg. 309 criteria.

4.6 Risk Comparisons for Used Household Battery Disposal Alternatives

It is necessary to determine the extent to which risks can be reduced, as well as the pathway for implementation of risk reduction policies. Table 28 summarizes the relative level of risks associated with the preferred disposal alternatives for used household batteries.

Table 28
Risk Comparison for Disposal Alternatives

Disposal Option	Quantitative Risk Measure*	
	Hazard Index	Carcinogenic Risk
Landfilling	0.4 (1.6)**	0.0 (0.0)
Incineration	3.5 (0.0)	5.3×10^{-3} (1.4×10^{-6})
Combined Landfilling and Incineration	1.9	2.7×10^{-3}
Recycling	Not Quantified	Not Quantified

* Shows value for the most sensitive potential receptor, i.e., population indicating highest risk measure.
** Numbers in parentheses show values for typical/actual case studies for selected disposal options; these are represented by Waterloo Landfill Site (Waterloo) and Tricil SWARU incinerator facility (Hamilton), both in Ontario.

Notes: Acceptable Hazard Index ≤ 1.
Acceptable Carcinogenic Risk Range is 10^{-4} to 10^{-7}.

Theoretically, incineration of the batteries of concern in this study will present the greatest risks; in practice, mixed with MSW, these batteries may safely be incinerated with MSW without any significant risks. Landfilling of the dry-cell batteries with MSW will generally present no significant risks of concern. Although the recycling of the household batteries have not been quantified, the qualitative indicators are that it is not the best disposal option for the alkaline (manganese) and the zinc-carbon/zinc-chloride cells; Ni-Cads recycling programs may, however, be a worthwhile effort.

5.0 CONCLUSIONS AND RECOMMENDATIONS

Waste disposal and management practices are shaped in part by federal and provincial regulation and legislation. Regulations governing waste disposal practices attempt to distinguish between hazardous and non-hazardous materials. Materials are determined to be hazardous based on a set of tests that examine their toxicity, flammability, explosivity, corrosivity, and/or infectiousness. Despite the toxicity of some of the composition of household batteries, dry-cell batteries are themselves not affected by hazardous waste regulations, since all household wastes entering the MSW stream are generally classified as non-hazardous.

Concerns about battery disposal practices stem from the possibility of hazardous materials/chemicals leaching from landfills or entering the atmosphere through incineration of MSW. On the one hand, the amount of household battery usage seems to be going up which augments the concern about its impact when disposed together with MSW. On the other hand, some of the amounts of more toxic chemicals used in some of the batteries are going down and/or being substituted with potentially less toxic ones, thus minimizing potential impacts of the presence of dry-cell batteries in MSW.

5.1 Conclusions

Most used household batteries become an integral part of the municipal solid waste stream. In practice, all solid waste is either landfilled or incinerated, with recycling becoming an integral component. Several conclusions may be drawn based on this investigation:

- The dry-cell batteries investigated (i.e. the alkaline, zinc-carbon/zinc chloride and Ni-Cads) do not generally represent a concentrated source of heavy metals in MSW
- There is no clear evidence to suggest that the co-disposal of dry-cell batteries with MSW via incineration or landfilling presents environmental or health problems.
- Risks to the environment from battery disposal by landfilling and incineration are not likely to be significant. Thus, most household batteries may be safely disposed of in municipal landfills or municipal incinerators; Ni-Cads are better landfilled.
- At present "recycling" of non-rechargeables is more likely to present significant risks. There appear to be significant health-related problems associated with the separate collection, storage and disposal of most household batteries. However, recycling for Ni-Cads may be a more viable and desirable measure to adopt. With the currently reduced levels of Hg in most primary

cells (especially the alkaline and zinc-carbon/zinc-chloride batteries), recycling of alkaline and zinc-carbon/zinc-chloride cells is not necessary or needed.

5.2 Risk Management Programs for Used Household Batteries

Table 29 presents the recommended methods for the management of spent dry-cell batteries. The preferred management option refers to the best available method of disposal that is recommended for use, whereas the alternative option is what can be called the second best method to adopt when necessary.

Table 29
Recommended Management Methods for Used Dry-Cell Batteries

Battery Type	Preferred Management Option	Alternative Management Option	Comments
Alkaline (manganese)	Landfilling	Incineration	Neither landfilling or incineration of even concentrated forms appear to present any significant risks
Zinc-carbon/zinc chloride	Landfilling	Incineration	Neither landfilling or incineration of even concentrated forms appear to present any significant risks.
Ni-Cads	Recycling	Landfilling	Separate collection and recycling of Ni-Cads preferred due to potential risks from Cd.

5.3 Recommendations for Action

Until the uncertainty regarding the recycling feasibility issues are resolved, municipalities should increase public awareness of their household hazardous waste collection programs in which residents voluntarily drop off batteries at a depot for disposal in a hazardous waste landfill, particularly in communities which rely on municipal waste combustion as the primary waste disposal method. In conjunction with this effort, the battery manufacturing industry should play a larger role in educating the consumer about the various battery systems and the environmental implications associated with their composition. The use of less toxic systems (i.e. zinc-air, lithium

cells) should be encouraged while the more toxic alternatives are gradually removed from the market. Better labelling of battery types and their contents would result in a more environmentally aware and generally informed battery consumer and perhaps alleviate public misperceptions regarding the toxicity of the household battery systems. Further source reduction of mercury in alkaline batteries should be continued among Canadian battery manufacturers, particularly in light of anticipated increases in alkaline sales over the next several years.

Specifically, the following recommendations are made regarding household battery disposal practices:

- There is the need to educate the general public with respect to the distinction between lead-acid automotive batteries and various types of dry-cell household batteries.
- Considerable research needs to be conducted to determine the effect of household battery disposal on landfill leachate quality, and the potential impacts on groundwater resources.
- All municipal incinerators should be equipped with wet gas scrubbers. So that the mercury emitted during combustion of household waste would be removed.
- Future directions for industry should include:
  - Public education
  - Encourage recycling where appropriate (eg. used Ni-Cads)
  - Leadership in terms of waste management
  - Continue research and development (e.g. battery substitution)
  - Establish communication strategies with municipalities and government agencies

Rather than impose policies, one could consider the option of letting market pressure and changes in technology lead to the possibility of a reduction in the use of heavy metals. In fact,batteries themselves are not the largest constituent in HHW, yet they are a ubiquitous and diffuse source of heavy metal waste. It is possible that managing their disposal efficiently could reduce the overall environmental threat posed by such metals.

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