ABSTRACT

KENNETH J. HAUSLE. A Decision Framework to Assist Local Communities in Managing Troublesome Solid Waste. (Under the Direction of Dr. DEBORAH A. AMARAL)

Certain solid wastes; tires, batteries, etc. present potential health risks if they are improperly managed. A methodology has been implemented for evaluating available options for managing these materials. Landfilling and incineration are the management options focussed upon but the framework can be expanded to more fully include other options such as recycling and banning. Potential human exposures from each option are compared to risk related health guidelines or standards to determine health risks. A case study evaluates management of polybrominated flame retardant materials in municipal solid waste in Wilmington, N.C. Aerometric and ground water models are utilized for estimating probability related exposures. The aerometric model is driven by a Gaussian plume model, and the migration of toxic material in ground water is estimated from a two dimensional analytical model sponsored by the Electric Power Research Institute. Exposure ranges in air, water, and food are developed by assigning probabilities to uncertain input parameters such as stack emission rates and landfill leachate concentrations. The data produced is for illustrative purposes in order to demonstrate the methodology. Frequency versus concentration plots are generated from which levels of exposure derived from different management options can be compared and the option which presents the lowest health risk to the community determined. Consideration is given to the cost of risk reduction to the community in order to implement the management option which poses the least health risk.

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Introduction

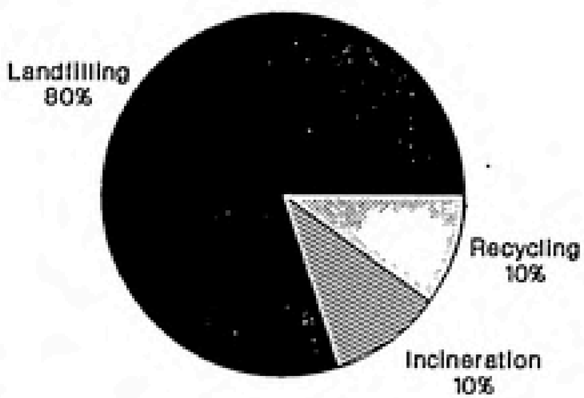
Managing solid waste has become a complex issue for local communities. Growing environmental consciousness, concern for health risks, and rising disposal costs have all increased the attention given to solid waste management. Local communities are having to make important decisions regarding how to deal with their solid waste. Some materials, primarily products of modern society such as certain plastics, tires, and batteries, are particularly troublesome to manage safely. These materials often do not readily biodegrade, pose health hazards if improperly disposed, and are not easily recycled. Local decision makers need a conceptual framework with which to determine the best solid waste management option for troublesome materials. For the purpose of this paper, management options will include all methods available to a community to manage solid waste such as disposal, material and energy recovery, as well as alternatives such as banning particular materials.

A wide array of information is needed by community decision makers to make management choices regarding a troublesome solid waste. This information includes among other items the quantity of the troublesome waste produced, the transport properties of the waste or its by-products in environmental media as it is being managed, the health impacts of exposure to the waste or its by-products, and the costs of the various options for managing the waste. For the majority of waste materials, such a wide array of information is not available, and few if any communities have the resources necessary to ascertain all of the unknown parameters.

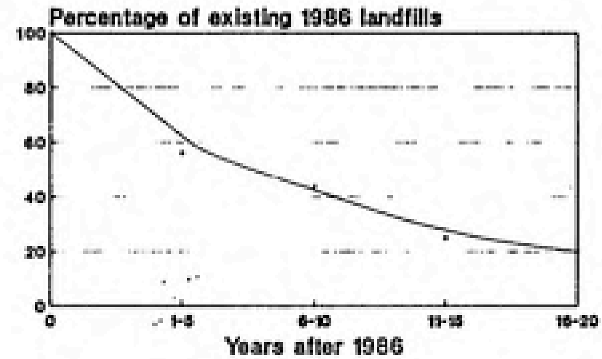
To make a decision amidst this uncertainty and complexity it is very difficult to be objective. Some subjective judgments are often needed to make estimates for uncertain parameters. In order to estimate uncertain parameters in a rational manner, a decision analysis approach is used. As stated by Ronald A. Howard (17), decision analysis is a methodology for making logical decisions in complex, dynamic and uncertain situations. It treats uncertainty effectively by encoding informed judgement in the form of probability assignments to events and variables. An important benefit of decision analysis is that it provides a formal language for communication among the people involved in the decision making process.

The components of municipal solid waste (MSW) are presented in Figure 1. Many potentially harmful metals and organic chemicals are components of products and packaging that are used at residences and offices and then discarded as MSW. When MSW is landfilled, incinerated, recycled or otherwise managed, these components have the potential to contaminate the environment and threaten public health. In this paper, these components are referred to as troublesome

Estimated Use of MSW Management Options, 1986



Estimated Decline in Existing Permitted Landfills



Estimated Portions of Materials and Products in MSW, 1986, by Weight

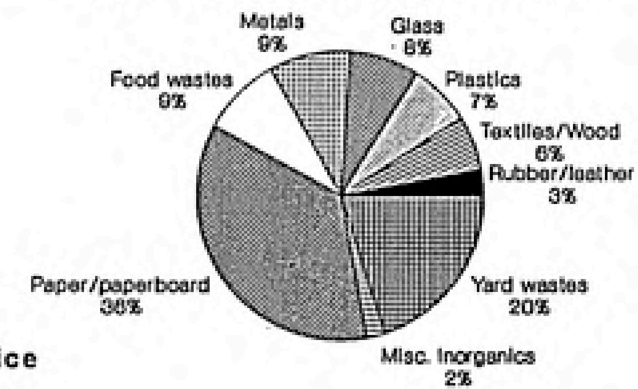


Figure 1. General data on MSW management. Source: Office of Technology Assessment, *Facing America's Trash: What Next for Municipal Solid Waste,* Washington D.C., 1989.

waste. In the report Facing America's Trash (31), published by the Office of Technology Assessment in 1989, troublesome wastes are discussed. Information from this report is briefly summarized in the following three paragraphs.

Mercury, lead and cadmium are the metals which have been focussed upon as posing potential health risks. Sources of mercury include most household batteries, fluorescent light bulbs, thermometers, and mirrors. The primary source of lead in MSW is automobile batteries, but it is also found in solder in steel cans and electronic components, paint pigments, and plastics. Cadmium is found in metal coatings and platings, rechargeable household batteries, paints, and as a heat stabilizer in plastics. Approximately 98% of the lead and 64% of the cadmium are in noncombustible materials, suggesting that separation of these materials from waste to be incinerated would reduce the amount of these metals in emissions and ash.

Household hazardous waste are another component of MSW that contains potentially toxic substances. Over 100 substances listed in RCRA as hazardous are present in household products. Household hazardous waste includes cleaning products, automobile products, home maintenance products, personal care products and yard maintenance products.

Plastics in 1986 made up 7% of MSW. It is estimated by the year 2000 that 10% of all MSW will be plastics (30).

Most plastics contain additives to give them specific useful properties. Over 4,000 individual types of additives exist and they can be classified into four major types: reaction controls, processing additives, stabilizers, and performance additives. Concern over the fate of additives when plastics are discarded has focussed primarily on heavy metal additives and organic and halogenated chemical additives. Heavy metal additives, particularly lead and cadmium are used as heat stabilizers in wire and cable insulating material, furniture film, floor tiles, and pressure pipes and colorants in a wide variety of thermoplastics. Organic chemical additives are used for example as "plasticizers" which impart flexibility and as flame retardants. Flame retardant additives are made up of a variety of highly brominated organic compounds and are the troublesome waste evaluated in the application of the decision framework outlined in this paper.

In Figure 1, the estimated use of MSW management methods is presented. Landfills have traditionally been the disposal method of choice for communities, however capacity is declining for the following reasons: 1) older landfills are reaching their capacity; 2) increased Federal and State regulation has resulted in the closure of substandard landfills and reduced the number of potential sites available for landfills; and 3) the public is extremely opposed to the siting of new landfills (29,31,40). Figure 1 displays a graph of the estimated decline in existing permitted landfills.

Incineration, which is used extensively in Europe, was seen as the ultimate solution for disposal problems, but it too has problems as follows: 1) the technology is unproven in the U.S. (American facilities have a history of operating problems some of which are thought to be caused by the higher percentage of plastics in American waste, leading to corrosion and unplanned shutdowns); 2) incineration can be a disincentive for recycling if the plant is oversized; 3) emission and regulatory standards have not been clearly defined; 4) there is a lack of operator training in facilities; 5) many toxic constituents have been measured in incinerator ash and emissions (7,29,30,31). These problems suggest that incineration is not the cure all for MSW management needs.

Despite their problems, the use of landfills and incinerators is likely to be relied upon in the future as the primary disposal methods of most communities, and when it is carried out appropriately can be the best method to manage particular wastes (29,40,46). Landfills will be needed to dispose of ash from incinerators and in some cases where the waste presents a minimal threat to groundwater, landfills may be the most economical means of managing the waste. New technologies are also being developed in which a landfill is run more with the philosophy of a chemical plant. The waste is the raw material and the products for example are energy in the form of methane collected as it is produced in the landfill, or fertilizer from a compost heap made up of organic waste (29,34).

To have a successful solid waste management program, recycling must play a central role and options such as composting which can convert organic waste into a useful product should be coordinated into the overall management scheme if possible. Recycling should be a top priority in managing solid waste because of its materials conservation benefits and its energy savings compared with manufacturing using virgin materials. In a community which has a comprehensive solid waste management program, several disposal and recycling alternatives should be available. Two reasons for this are that it enables manufacturing of products specifically for a particular management option (e.g. by designing products for recyclability) and it enables solid waste management to be approached on a material-by-material basis where waste material is diverted to the most appropriate management method based on its physical and chemical characteristics. To make informed decisions regarding solid waste management alternatives, local communities need to be aware of what the health risks and costs are for each alternative. There is considerable uncertainty in evaluating the health risks that result from how a particular waste is disposed, and to a lesser extent there is uncertainty in determining the entire cost of a management alternative. The health risks depend on the magnitude of exposure to individuals from various environmental media, and the potential harm this exposure causes. The costs are very specific to the local community and depend on many factors such as local wages, land values, transportation systems, and the size of the community (13,29,43,44).

When confronted with such uncertainty there are many differing opinions as to what values should be assigned to parameters of concern such as the concentration of a substance leaving an incinerator stack. One approach is to assign a best estimate to an uncertain parameter. A problem with this approach is that it masks the inherent uncertainty in the parameter by assigning it only one value. In the event that the best estimate is incorrect and results in an underestimate of risk, this approach can have disastrous consequences. Another approach is to assign the most conservative estimate to the uncertain parameter so as to minimize the possibility of underestimating risk. This approach however also masks the parameter uncertainty and tends to overestimate the risk. The conservative approach assumes that the value of avoiding a negative outcome such as one additional cancer case is extremely high. This can result in spending a considerable amount of resources to avoid a risk that is likely to be overestimated in the first place, and can lead to financial expenditures that may be better utilized elsewhere.

The approach outlined in this paper is to use a range of values to accurately represent the uncertainty of a parameter. The range is based upon the best information available. In the event of little information, the range of values for a parameter is greater to reflect the high uncertainty. Models are used to represent the physical processes occurring, and assumptions made are explicitly stated. A specific effort is put forth to prevent the models from becoming tools to hide assumptions and cloud the uncertain nature of the input parameters.

The decision framework presented in this paper attempts to maximize reduction of health risks posed by a troublesome solid waste while minimizing the cost of implementing the risk reducing management option. The focus will be on landfills and incinerators, but management options such as recycling, composting and banning of materials are considered and can enter the decision framework if they are alternatives for managing the troublesome waste. Landfills and incinerators are the primary focus of this study because they are the predominant solid waste management alternatives used by local communities today, and are a starting point for the development of the framework. This by no means suggests they are the best management alternatives available, and as previously stated a goal of local communities should be to develop comprehensive solid waste management policies providing many options for managing waste. The framework developed can be extended to comprehensively include all feasible management options, providing a consistent comparison of expected performance as measured by attributes such as risks and costs.

Method

The decision framework follows the outline shown in Figure 2. Initially the management options for a troublesome waste and the resulting exposures that occur for each option are determined. Next, the health impacts of the exposure for each management option are assessed and it is determined which option or combination of options poses the least health risk. The management option that poses the least health risk is called the preferred management option. Finally, the costs of managing the waste using the preferred management option are assessed and a recommendation as to what action should be carried out is made.
If the costs are too high for the reduction in health risk, a new option is evaluated to see if it has a more favorable cost to risk reduction ratio. In a situation where several risk reducing options exist, the costs can be evaluated for each option. These steps will each be described in more detail and an application of the framework will be made in a case study evaluating brominated fire retardants in

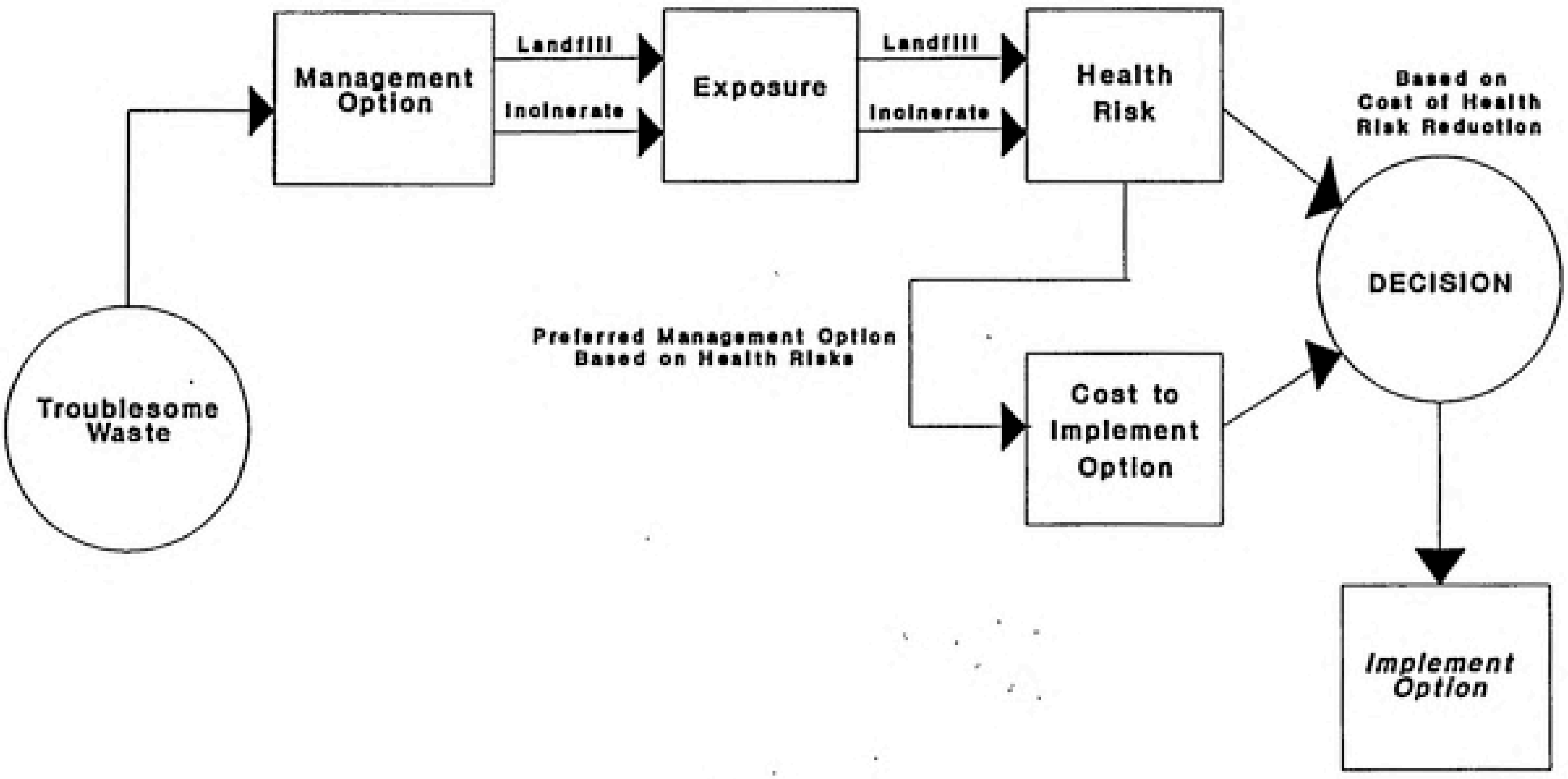


Figure 2. A decision framework for managing a troublesome MSW.

Wilmington, North Carolina.

Models play a crucial role in the decision making process. They are used to avoid the potentially expensive cost of actually obtaining data as well as to make predictions concerning future events. Decision analysis is based on choosing the course of action which results in the greatest likelihood of obtaining the most desirable future. Typically a decision analysis is performed with a sequence of progressively more realistic models. These models can be referred to as the pilot model, the prototype model and the production model (17). The pilot model is an extremely simplified representation of the problem and is utilized to determine important parameters and their relationships. The prototype model is a more detailed but not entirely complete representation of the problem. It gives an indication of how the final model will appear and perform. The production model is the most accurate representation of reality that a decision analysis can produce given spending limitations.

Throughout the modelling sequence, sensitivity analysis is used to determine the most important parameters. Sensitive parameters are those that highly influence the output of a model. These parameters are included in the production model if possible. In certain cases, developing a model to include an uncertain parameter is expensive. If the analyst can calculate the value of perfect information about the uncertain parameter, he can evaluate if the cost of additional modelling is merited. This practice is referred to as the value of perfect information technique and may be applied at several points within the decision framework.

Exposure/Intake

Individuals can be exposed to contaminants through air, water and food. What actual exposure levels are and who is exposed as a result of managing a troublesome waste are a function of several factors. These factors include how much waste is generated, what management methods are available for the troublesome waste, environmental conditions in the community such as average wind velocity and direction, and the population characteristics of the community. Processes which lead to human exposure to troublesome components of MSW are summarized in Figure 3. For landfills, the typical environmental medium that is contaminated is groundwater, but contamination of surface water is also possible. For incinerators, the contaminated media include air from emissions and groundwater from incinerator ash disposed in a landfill. Air emissions may subsequently lead to deposition in surface water and onto soil and vegetation which then is passed up the food chain. Thus, by carrying out groundwater modelling and air modelling with consideration for deposition onto soil, vegetation and surface water, overall exposures can be determined.

The first step in carrying out an analysis is to

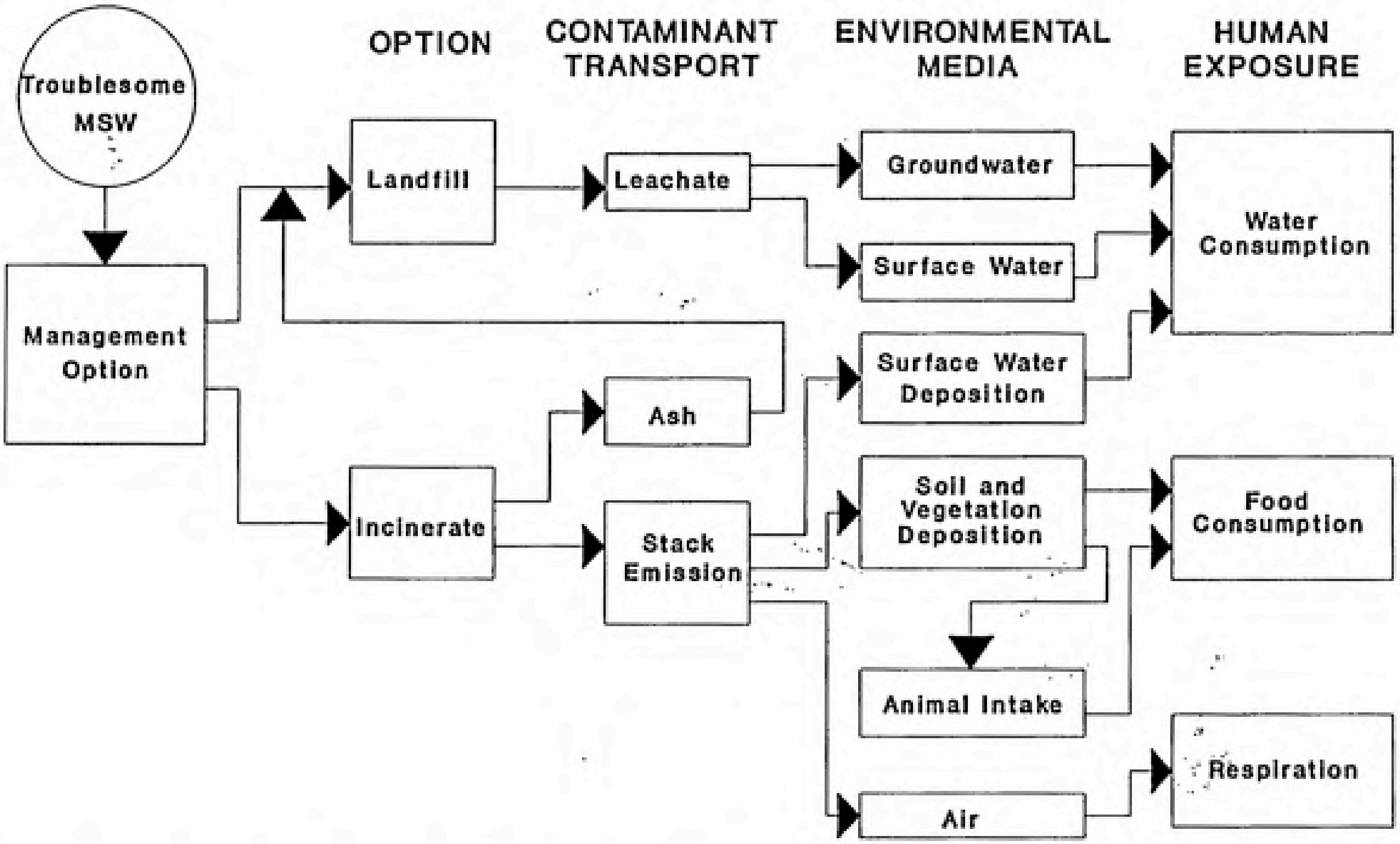


Figure 3. Flow chart for evaluating human exposure for the management options of landfilling and incinerating a troublesome MSW.

determine which input parameters are uncertain and which can be determined with a high level of confidence. Once those parameters of uncertainty are determined, each must be evaluated for its sensitivity to the final exposure concentration. If the parameter is uncertain, but changes in the parameter have little effect on the overall concentration, then it is an efficient use of analytical resources not to perform detailed analysis to determine the range of values for the parameter. The parameter must be adjusted under several different conditions before being considered non-critical in concentration determination. This can be done by keeping several combinations of parameters constant and adjusting the parameter of interest for each combination. Those parameters that cause considerable fluctuation in the final concentration must be further analyzed. This analytical process is called sensitivity analysis and is at the heart of an application of this framework.

Groundwater modelling, air modelling and food exposure evaluation require several input parameters. These parameters as well as the quality of information regarding them and their sensitivity to the model prediction are listed in Tables 1A and 1B and 1C. The ratings for uncertainty/sensitivity are very general and in specific cases may not apply.

Tables 1A-1C list the important parameters in

TABLE 1A: Parameters needed in groundwater modelling (9,23,36)

Parameter	Method of Determination	Relative Uncertainty/ Sensitivity
-saturated hydraulic conductivity, K, (m/yr)	-measure at site -base upon soil type	MODERATE/ MODERATE
-hydraulic gradient, I, (m/m)	-measure at site -base on local characteristics	LOW/ MODERATE
-porosity, p (unitless)	-measure at site -base upon soil type	LOW-MODERATE/ MODERATE
-seepage velocity, V, (m/yr)	-estimate with equation V = (K * I)/p	MODERATE-HIGH/HIGH
-Longitudinal dispersivity coefficient, Cl, (m)	-estimate as 0.1 times distance of interest	MODERATE/ LOW
-Longitudinal dispersion, Dx, (m/yr)	-estimate with equation Dx = Cl * V	MODERATE/ LOW
-Transverse dispersion, Dy, (m /yr)	-estimate as 0.1 - 0.3 times Dx	MODERATE/ LOW
-Vertical dispersion, Dz, (m /yr)	-estimate as 0.1 - 0.01 times Dx	MODERATE/ LOW
-Retardation, R, (unitless)	-refer to literature -use estimation technique based on solubility or octanol water coeff. -measure in lab using soil from site	HIGH/HIGH HIGH/HIGH
-Decay rate, T, (1/unit of time) Note: Applies to organic chemicals	-refer to literature -use estimation technique	HIGH/HIGH
-Source Concentration, Co, mg/liter	-measure at site -refer to literature	MODERATE-HIGH/HIGH

TABLE 1A (cont.)

Parameter	Method of Determination	Relative Uncertainty/ Sensitivity
-Penetration depth, P, (m)	-use estimation technique -measure at site	MODERATE/ LOW
-Aquifer thickness, d, (m)	-measure at site -base on local characteristics	MODERATE/ LOW

TABLE 1B: Parameters needed in air modelling (3,42,45)

-Stack emission concentration, Cs, (mg/m)	-measure -refer to literature -calculate based on input and destruction ratio	LOW-HIGH/ HIGH
-Stack height, H, (m)	-measure	LOW/MODERATE
-Stack exit gas velocity, Vs, (m/s)	-measure -calculate	LOW/MODERATE
-Weather/wind profile for community of interest	-base on data for actual or nearby community	LOW-MOD./ HIGH
-Degradation	-refer to literature	HIGH

TABLE 1C: Parameters needed for food exposure modelling (15,38) NOTE: Relative sensitivity not rated for food exposure parameters

-air concentration	-air modelling	HIGH
-soil deposition	-refer to literature -estimation technique	MODERATE
-vegetation deposition	-refer to literature -estimation technique	MODERATE
-decay in soil and vegetation	-refer to literature -measure/estimate	MODERATE

TABLE 1C (cont.)

Parameter	Method of Determination	Relative Uncertainty
-animal intake	-refer to literature -measure for site	LOW-MODERATE
-animal uptake	-refer to literature -measure/estimate	MODERATE-HIGH
-human intake	-refer to literature -measure for community	LOW-MODERATE

evaluating exposure as a result of managing a troublesome MSW. The levels of uncertainty are based on the amount and quality of information for a particular parameter. The level of uncertainty for retardation, decay rates, contaminant intake and uptake by biota, and to a lesser extent deposition rates are uncertain primarily because of the complexity of the physical process. These parameters are a function of many variables making them site specific and difficult to estimate. The parameters source concentration, stack emission, and to a lesser extent seepage velocity may be uncertain due to the lack of measurement data rather than the inherent complexity of the physical process.

As previously stated, the sensitivity ratings are very general and may not apply to all cases. For example, if a community was interested in the potential contamination of groundwater at a specific well obtaining water at a given point away from a landfill, then the dispersion values (Dx, Dy, and Dx) may be highly sensitive to the contamination concentration. However, when considering general levels of groundwater contamination, changes in the dispersion values do not have a great impact on the contamination level. On the other hand, under most circumstances changes in seepage velocity, retardation, and decay rates, have a significant impact on concentration levels. Seepage velocity regulates the distance the groundwater plume will travel; retardation essentially reduces the seepage velocity and when it has a high value can dominate the other variables; and the decay rate causes reduction in concentration as the plume spreads away from the troublesome waste source. The source concentration and emission rate are directly correlated with the contaminant level, and thus, the model prediction is highly sensitive to their value.

The sensitivities for parameters used in evaluating food exposure are case specific and not cannot accurately be generalized. If a substance has a high rate of decay on vegetation, than the vegetation deposition and uptake will have little impact on overall food exposure. Of course the exact opposite impact would result if there was little decay on vegetation and vegetation was the primary source for human consumption. In this way, all of the parameters in food exposure have the potential to highly effect the model prediction.

Additional factors very important in evaluating exposure impact are population characteristics such as location relative to source of contamination, density of population and future projections for population characteristics. If an exposure is very high but effects only a few individuals, its impact may be less than a low exposure effecting many.

Once the parameters of greatest uncertainty and sensitivity have been determined, a statistical distribution is used to represent the uncertainty. This can be carried out at different levels of complexity. For the case study in this paper, the uncertainty is represented by establishing a range of discrete values for each parameter and assigning each value a corresponding probability. A more complex approach is to establish a distribution such as normal or Poisson that reflects the uncertainty as a continuous function. Both approaches are based on the same concept of expressing the uncertainty over a range of values. The continuous range simply provides more detailed data and should be applied if this detail is considered important in making the final decision.

The following is an example of using discrete values to represent a parameter's uncertainty. For groundwater seepage velocity, the expected range may be from 10 m/year to 100 m/year, and this could be expressed as 20% likely that flow is 10 m/year, 60% likely that flow is 55m/year and 20% likely that flow is 100 m/year. The actual number of values used depends on the level of uncertainty for the parameter. For a parameter that is highly sensitive to the final outcome with an extremely wide range of potential values, it is appropriate to assign more discrete values to reflect its uncertainty. In assessing a likelihood for each value in the range, as much information as is readily available should be used. It is recommended that in order to understand and make explicit the assumptions underlying how the numbers are arrived at, each value and its corresponding likelihood must be defended.

There are many sources of information to obtain values for the parameters. Ideally the actual values can be found in the literature, but this often is not the case. There are chemical estimation techniques which enable one to calculate various characteristics of a particular chemical (1,21). The estimation techniques can be based upon information from a similar chemical, and/or data on the chemical such as the octanol water coefficient or water solubility. Other sources of information include engineering studies such as groundwater boring at a landfill site and design or operating conditions for a facility such as the exit temperature from an incinerator stack. For some parameters, current data such as lab analysis of groundwater or incinerator ash can be useful. Finally when there is no information in the literature or in order to further defend parameter values, direct communication with experts in the field can be a source of information. In the event that no information can be obtained, then the range for the parameter will have to reflect the great deal of uncertainty. For parameters such as this, if appropriate and possible it may be worthwhile to perform actual research to obtain a better understanding of its potential value. The value of perfect information technique mentioned previously is useful to evaluate the appropriateness of investing time and money in research.

Once values and their corresponding likelihoods have been established, the probability of environmental media concentration for the contaminant can be determined. When discrete values are assigned to uncertain parameters, there are a given number of resulting scenarios to be input into the model. For example, if there are four parameters whose value is uncertain, and each is expressed with a range of three values, there would be 3 to the 4th or 81 possible scenarios. The probability of each scenario is the multiplication of each of its parameter value's likelihoods. Thus, the contaminant concentration in a particular environmental medium, as calculated through the use of a model, will have a range of possible overall values each with a corresponding overall likelihood. This contaminant concentration range for exposure can be converted to a range of human intake by using a standard factor (i.e. 2 liters of water consumed per day). These factors themselves also contain uncertainty. A cumulative distribution curve for the probability that the intake of a contaminant is lower than a given value can be generated. This is accomplished by arranging the intake values from lowest to highest and adding their corresponding likelihoods. The probability that an individual receives less than a given intake, X, is equal to the summation of all the probabilities for each scenario resulting in an intake value below X.

<u>Health Risk</u>

Once the range of intake through air, water, and food has been determined, the next step is to assess what effect this intake has on human health. Typical communities may not have the resources to carry out a full fledged health risk analysis. This will be the case particularly for those wastes for which health effects are unknown. For each management method, the corresponding intakes (i.e. air, water, food) need to be evaluated for their health risks. Because the intake amounts are over a range, the health effects are also over a range. The final output is a curve of likelihood of a particular health risk (e.g. increase in lung cancer or expected number of increased cases of heart attacks, etc.) for the community as a result of a particular management option. Each management option has a different range of possible effects, and that option which minimizes health risks is called the preferred management option. For many substances, there exist health guideline values such as a No Observed Adverse Effects Level (NOAEL) or a Virtually Safe Dose (VSD) value which can be used to assess risk. The NOAEL level is based on the assumption that there must be a threshold before a substance has a harmful effect and is usually determined in animal experiments involving lifetime exposure (10). The NOAEL dose is divided by a safety factor to allow for increased human sensitivity and varying sensitivities amongst humans to calculate an Acceptable Daily Intake (ADI). VSD's are used by United States regulatory agencies to regulate chemical carcinogens and represent a daily dose which correlates to an additional cancer case per million individuals over a lifetime (14). Health guideline values are calculated and published by several federal agencies, primarily the Environmental Protection Agency, but also the Food and Drug Administration, the Center for Disease Control and other concerned agencies. The values are based on cellular, animal and/or human exposure studies and are best estimates as to risk and are available for a wide variety of substances including metals and organic compounds. There is considerable uncertainty in these values, and this uncertainty will be briefly described. Nevertheless they are often the only measures available to assess what health risk a substance poses, and although relying on a single value for health risks masks the uncertainty in the value itself, it is an indicator of risk.

Health guideline values for a chemical (or values for similar substances if none exist for the actual chemical of interest) can be used to carry out simple analysis of potential risk. The range for intake previously calculated can be divided by the guideline value to determine the level of risk the intake poses. This is demonstrated in Figure 4.

As previously mentioned, assessing health risks for exposure to a substance is laden with considerable uncertainty. For many substances, there simply is no information available pertaining to their health risks. For those in which health risks have been evaluated, many factors contribute to uncertainty in the results. Several key contributors to uncertainty in assessing health risk are briefly described below.

Extrapolating data from animal studies to humans is a difficult process. Two major extrapolations are: interspecies adjustments for differences in size, lifespan and basal metabolic rate and extrapolation of the dose-response relation observed at doses used in animal experiments to lower doses to which humans are likely to be exposed (10). Chemical agents vary widely in extent of absorption among animal species and ideally this should be taken into account, but there are limited data on absorption for most chemicals (10,27). Another difficulty in making interspecies adjustments is that it is not infrequent for the route of exposure given to the study animals to not

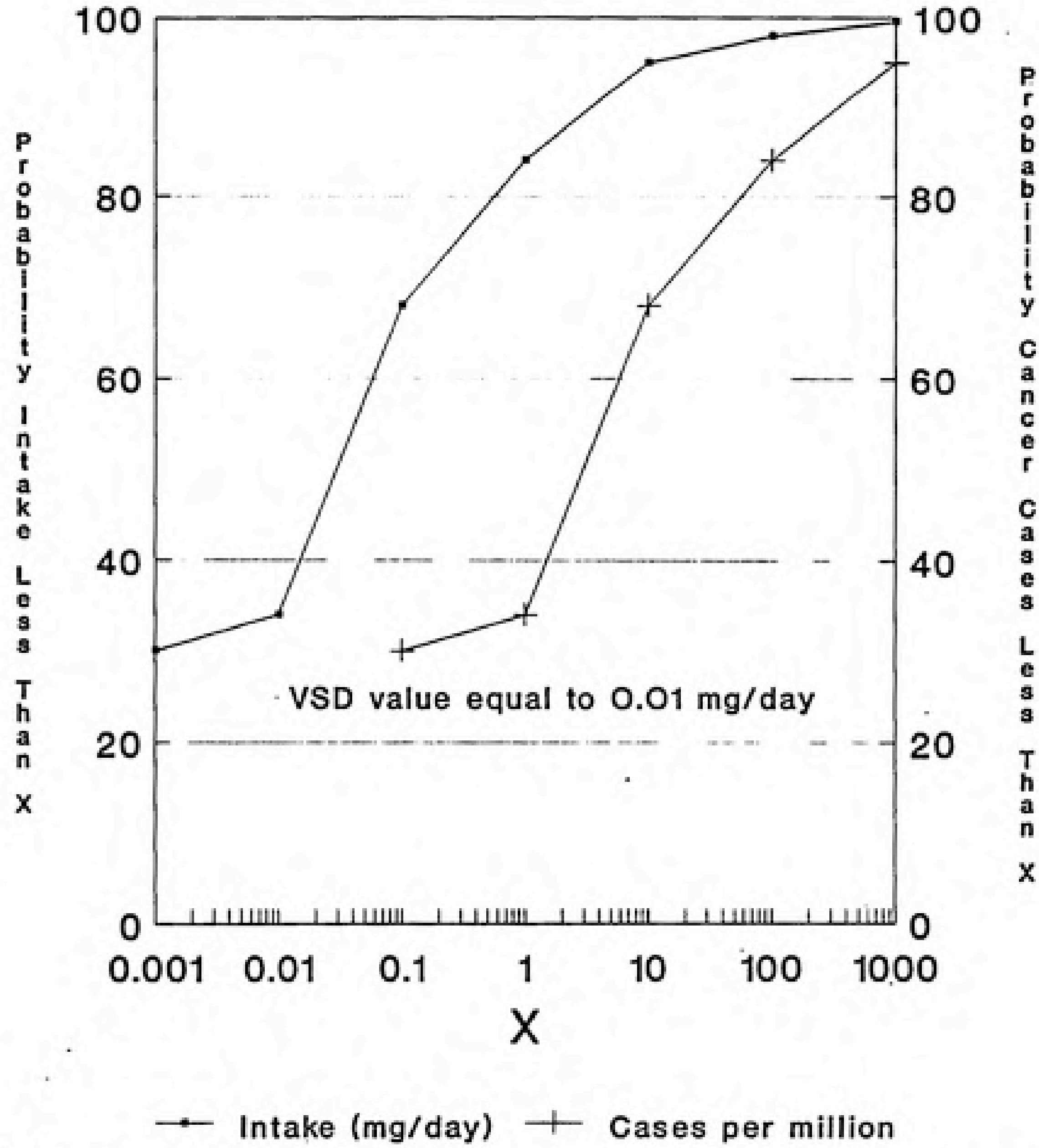


Figure 4. Example of a cumulative confidence curve for intake, and conversion of the intake to the health risk of cancer cases through the use of a hypothetical very safe dose (VSD) value. The VSD value represents risk of one additional cancer case per million lifetimes.

accurately represent human exposure. For instance, the animal dose may be given in their food whereas the typical human dose occurs by inhalation. High doses are used in animal studies in part to account for the small number of experimental animals used. Animal studies are very expensive and costs limit the number of animals that can be studied. It is hoped that by giving high doses statistically significant results can be obtained. However, this creates the problem of extrapolating from high to low doses. Scientist have developed several mathematical models to estimate low-dose carcinogenic risks from observed high-dose risk. These models tend to fit the experimental, high dose data, but the predicted risks at low doses may vary significantly (2,10). Knowledge of actual biological mechanisms of a substance and how these lead to harmful effects is important to truly understand the impact a substance has on human organs and tissues. This knowledge facilitates risk extrapolation from animals to humans and guides researchers in what is the best method to study a substance's toxicity (2,27). However, Even for highly studied substances such as dioxin (2378-TCDD), biological mechanisms are not clearly understood (14). Even when biological mechanisms are understood, varying sensitivities among humans of different ages, for example, further complicates the extrapolation process and risk determination.

When human exposure data are available the problems of extrapolation can sometimes be avoided. However, human exposure data are often unavailable, missing or imprecise, and once a substance is suspected of being harmful it is usually too late to obtain human data. Efforts to reconstruct past exposure levels have not been extremely successful and have led to conclusions which were later refuted (20).

There are several additional problems in general when attempting to determine health risk. Present studies do not attempt to account for multiple and mixed exposures which are common in the environment and workplace and may play a role in health risks (20). Most risk assessments do not even consider health risks other than cancer and results are essentially unverifiable without using epidemiological techniques, which due to methodological limitations cannot be done (20).

Awareness of these uncertainties is important so that a decision maker understands the limitations of his findings. Amidst all of this uncertainty a decisionmaker in a local community can only be expected to obtain as much information as possible and to act in an appropriate manner. This paper describes a simple and relatively conservative method to carry out a health risk analysis. If after using conservative health risk assumptions there is no significant risk, then further analysis of uncertainty for health risks is not needed. A more detailed analysis may be appropriate when the costs and risks involved are high. Substances whose health risks have been more thoroughly studied lend themselves to a more detailed analysis. These substances may have an actual range of potential health risks per unit of intake and can be combined with the range for intake previously calculated to determine a final range of health risks for the community. In cases where it is possible to obtain better information on health risks, value of perfect information techniques can again be utilized to determine if it is worth the cost to acquire the information.

Once health risks are assessed the preferred management option can be chosen. Difficult comparisons such as low exposures for many in the near future versus high exposure for few many years into the future need to be made. The best choice may be to utilize one management option only such as incineration, several management options, or on the other hand to exclude one management option which presents high risks. One community's preferred option(s) may be different than another even for the same troublesome waste.

<u>Costs</u>

Once health risks are established, the final step is to determine if action should be taken to change the management method for a troublesome waste. This decision is based upon the cost of changing the present management method and on what reduction in health risks this cost will achieve. Note that costs of changing management methods are evaluated with respect to a single troublesome waste and not solid waste in general. The following list of questions, which will be discussed in more detail can be used as a guide to estimating costs:

1. How much total waste and how much troublesome waste does the community discard, how is the waste distributed amongst management alternatives, what factors determine the distribution, and is the preferred management option the same or different than currently being practiced?
2. If the preferred management option is different than currently practiced, what factors unique to the community may affect their willingness to implement the preferred management option?
3. For various separation methods, what will the cost be, and what separation will be achieved?
4. Once the waste is separated, can it be recycled?
5. What will be the additional cost (or savings) in changing the management method after the waste is separated?
6. What is the total cost and is there a separation and management option that reduces health risks at a cost the community is willing to pay?

Question 1: A basic piece of information is how much total solid waste a community discards and what proportion of that waste is the troublesome waste. Knowledge of how the troublesome waste of concern is distributed amongst management options and what the factors are that dictate the distribution is needed when making a management decision concerning the troublesome waste. Many factors dictate how waste is distributed amongst different management options. These include capacity of disposal sites, waste pickup location within the community, contractual agreements to deliver specified amount of waste, or existence of a central collection facility or transfer station from where waste is divided among management alternatives (44). Some distribution schemes result in consistent waste makeup sent to management alternatives of the community (i.e. distribution based solely upon location) whereas others may result in daily fluctuation in amount and content of waste sent to alternatives (distribution after waste arrives at central facility). Those schemes that tend to be consistent will simplify implementing separation procedures.

An example of carrying out step 1 follows. It is determined that for a troublesome waste in Community X, 60% is sent to an incinerator and the remaining 40% is sent to a landfill. The distribution is based on the following facts: 1) Location within the community determines whether waste is sent to the landfill or incinerator; 2) 60% of all the solid waste goes to the incinerator and 40% goes to the landfill; 3) it is assumed that the troublesome waste is the same proportion of total waste across the entire community. Knowledge of the distribution of a troublesome waste and the factors controlling the distribution is needed to develop a plan to separate the troublesome waste from the overall waste stream. For instance, since community X has a rather consistent distribution that is based upon location, assuming incineration is the preferred management option, then a separation program only needs to be set up in locations where the waste is sent to the landfill. Had the waste in community X been delivered to a central facility where the waste distribution between landfill and incinerator was not consistent, then a separation program would have to be implemented for the entire community or at the central facility.

Whether or not the preferred management option is different or the same than is presently carried out in the community is also important. In the example case, had the preferred management option been to send the waste to a landfill and the community already sent 90% of the troublesome waste to a landfill then no further action may be called for, however if the community only sends 20% of the troublesome waste to a landfill, then removing the troublesome waste from the overall solid waste stream may be necessary.

Question 2: Different communities will place different values on the reduction in health and environmental risk they achieve by modifying their method of managing a troublesome waste, and there is no set formula to determine the value of risk reduction. Many factors play a role when a community is deciding how much it would be willing to spend to correct a solid waste health risk. These factors include budget constraints, other health risk concerns which may have higher priority, political forces, and the general attitude of citizens in the community as to the amount of risk they feel is acceptable.

One way to think about the health and environmental risks for a particular management option is as the liability a troublesome waste poses to officials and government in the community. For other types of liability, insurance costs could be used as an indication of the costs; however, this is not the case for liability caused by exposure to pollution. The insurance industry in general has attempted to exclude coverage "to bodily injury or property damage arising out of the discharge, dispersal, release or escape of smoke, vapors, soot, fumes, acids, alkalides, toxic chemicals, liquids or gases, waste materials, or irritants contaminants or pollutants into or upon land the atmosphere or any water course or body of water" (6). Despite the exclusion, courts have often ruled that the insurance does cover liability expenses in cases of pollution and as a result general liability coverage for environmental contamination is difficult to find and limited in protection (6,11). Nevertheless, a community may be able to estimate what its potential liability would be. Liability costs include compensation costs which are payments made out to individuals who suffered as a result of exposure to a toxic substance, abatement costs which are the costs to cleanup a contaminated site and administrative costs which include governmental administrative expenses as well as costs of acquiring information to handle the problem in the first place (11).

As well as costs to the community government, there may be social and economic costs to the community in general. Poor health in the community can disrupt many normal everyday activities including one's ability to work. In addition, personal suffering of individuals and families can occur. How the community values avoiding these social costs will determine how much should be spent in changing management options for a troublesome waste.

Question 3: Separation costs depend on many factors such as the desired separation percentage, the method of separation implemented, and what proportion of the community's waste already is managed using the option(s) of choice. Generally the more money that a particular community spends the greater separation they can achieve. However, to achieve the same desired distribution, different methods may be implemented by different communities depending upon local characteristics. Rural areas are likely to rely more upon citizen participation in the separation process. in many rural areas citizens are responsible for delivering their normal waste to a central pickup site (44).

Separation can be carried out at a central facility where all the solid waste is delivered or at the source by community residents. Source separation programs depend on the type of material collected, the frequency of collection, whether materials are collected at curbside or delivered to a central collection area and whether separation is voluntary or mandatory (34,44).

In order to calculate the cost of a separation program the following information is needed: 1) capital costs; 2) operating expenses; and 3) resident expenses. Capital costs include such items as new collection vehicles, storage bins, modifications made to present collection vehicles, mechanized separating equipment and planning costs. Operating expenses include labor costs, fuel costs, maintenance and repair costs, and administrative costs. Resident time and space expenses include time spent separating troublesome waste, space required to store waste and expenses in delivering waste to collection site if carried out by residents.

Below is a list of potential separation methods and a general indication of efficiency and costs.

Method 1: Use unskilled labor to separate troublesome waste at a facility after it has already been collected with regular solid waste.

EFFICIENCY--Low to High
CAPITAL COST--Low
OPERATING COST--Moderate to High
RESIDENT COST--None

COMMENTS: This approach may be appropriate when the troublesome waste is easily separated by hand when mixed with other solid waste, if only a minority of the troublesome waste needs to be separated out because the majority is already being managed using the option(s) of choice (for instance only the waste sent to the landfill needs to be separated), or if unskilled labor is readily available. It has the advantages that very little if any new equipment needs to be purchased and no changes to regular solid waste collection are needed. Disadvantages are that it is very labor intensive, and there are potential health and safety hazards for the workers. These health risks will have to be evaluated and incorporated into the overall health risk consideration.

Method 2: Mechanized separation at central facility after troublesome waste has been collected with regular solid waste.

EFFICIENCY--Moderate to High
CAPITAL COST--Moderate to High
OPERATING COST--Moderate
RESIDENT COST--None

COMMENTS: Some wastes such as paper, aluminum, and iron which can be recycled have been separated using mechanized equipment (32,44), and these procedure may be applicable to certain troublesome wastes. However applications are limited, and capital expense and economic risks are likely to be high. This approach does have the advantage that the regular solid waste collection system does not need to be changed but should only be used when the technology is proven and there are no simpler alternatives.

Method 3: Pickup of troublesome waste during regular collection times using a normal truck that is modified to accommodate separation of waste.

EFFICIENCY--Moderate to High
CAPITAL COST--Low to Moderate
OPERATING COST--Low to Moderate
RESIDENT COST--Low

COMMENTS: This method is primarily applicable to troublesome waste that is widely distributed throughout the community. It has the advantage that there is little disruption to the normal collection operation and the waste is separated before taken to a facility. Trucks can be modified by adding special racks or attaching a trailer which is specifically designed to handle the troublesome waste (32). A disadvantage is that extra time may be required as a result of waste separation thus increasing the time needed for a collection crew to complete its regular route. The efficiency of separation is a function of resident participation. Test have suggested that mandatory separation programs tend to have higher levels of participation (32).

Method 4: Use of separate truck to collect troublesome waste.

EFFICIENCY--High
CAPITAL COST--High
OPERATING COST--Moderate to High
RESIDENT COST--Moderate

COMMENTS: This is an effective but relatively expensive approach. Capital costs include purchasing new trucks and operating costs include hiring a crew and maintenance and repair on the trucks. However, if the troublesome waste is not part of normal everyday waste and one truck can service a large area, or this waste can be picked up by trucks the community presently uses to collect recyclable material, this may be the most appropriate and economic collection method. Residents will need to be informed what the operating procedures are for the new collection truck.

Method 5: Resident separation and delivery to central collection centers.

EFFICIENCY--Low to Moderate
CAPITAL COST--Low to Moderate
OPERATING COST--Low
RESIDENT COST--High

COMMENTS: This may be the most appropriate method when there is little capital available to implement a separation program or if the community is rural. The primary burden is on the residents who must separate the waste and deliver it to the central facility. Thus the efficiency of the separation relies totally upon resident participation and participation rates for delivering waste to a central facility have traditionally been lower than when the waste was picked up at curbside. The only capital costs are for dumpsters to collect the waste and cost of setting up facilities. Operating expenses include cost of periodically picking up the waste and transporting it to the final destination.

<u>Method 6</u>: Contract with private firm to carry out separation procedure.

EFFICIENCY--Low to High
CAPITAL COST--Low
OPERATING COST--Medium to High
RESIDENT COST--Low to Moderate

COMMENTS: This approach reduces the administrative burden on local authorities and may be particularly appropriate if the contractor has some use for the troublesome waste. It enables a community to take advantage of the expertise an outside contractor may have. The "purchase" of this expertise may have a high initial cost but in complicated situations may pay for itself by minimizing avoidable expenses. Cost and efficiency will be a function of the contract and can vary considerably, and liability responsibilities will have to be agreed to in the contract. Many of the same issues and costs outlined above will also hold for this method, but will be passed through by the contract.

<u>Question 4</u>: Once the troublesome waste has been separated, does it have a potential use that would eliminate disposal costs and possibly even have recycling value? If it does, this value serves to reduce the overall cost of separating the waste. In some cases private contractors may be willing to accept the waste for a smaller cost than any disposal option. The availability of consistent secondary markets needs to be evaluated before recycling can become a viable

option.

Question 5: Once a waste has been separated the cost of managing it using the preferred management option based on health risks as opposed to the current option needs to be calculated. In this step, the focus is on solid waste disposal costs in general rather than upon the specific troublesome waste. Once a cost difference per ton between the new and old options is calculated, the marginal cost (or savings) of changing the option for the troublesome waste can be determined based upon the amount of troublesome waste.

The cost for a management option can be broken down into collection (including hauling) and disposal (recycling will be considered a method of "disposal"). Historically collection of the waste has been the primary cost averaging 60-80% of entire costs (13,29,31) However, as waste facility sites become increasingly expensive to build, the disposal costs have begun to increase (13,31). For instance, in Charlotte N.C., the cost of curbside collection of normal MSW is approximately $35 per ton while the cost of incinerating the waste is $23 per ton (16). Depending upon the management option different components of the cost will be different. For instance, again in Charlotte N.C., the collection cost for recycled waste is approximately $70/ton and the disposal cost is $7/ton (16). Care should be taken in evaluating costs between different management options because these costs can be a function of one another. For instance if a community normally incinerates 25% of its waste but due to a breakdown has to send this waste to a landfill, the cost on a per ton basis at the landfill may drastically change. All of the costs work together. Nevertheless, when considering what the differences in cost of management options are for a troublesome waste, unless the troublesome waste is a significant proportion of the total waste stream then it can be assumed that the costs for each management option will not be altered by changing the current management option for the troublesome waste.

Each community has a unique cost structure. Major sources of cost variation between communities are wage rates, method of collection, disposal options, land costs, and the size of the community (29,44,47). Additionally, some communities own and operate the management facility as well as the collection service whereas some facilities are privately owned and operated. In rural areas, waste is often collected by private operators or hauled by residents to a central location. When determining the disposal costs at a community owned and operated facility, capital costs and operating and maintenance costs must be known, whereas in determining costs at a private facility these costs are all accounted for in one set tipping fee based on the same elements. A more detailed description of collection cost, hauling cost, operating and maintenance cost, and capitol cost is below.

Collection costs

Collection costs include capital cost to purchase trucks and operating and maintenance costs for labor, fuel, truck repair, and administrative expenses. These costs depend upon the crew size, type of collection truck, type of pickup (curbside vs back door), frequency of pickup, and distance to disposal site or transfer station (44,46). In the event that a private hauler picks up the waste, then the collection cost are normally a standard rate charged to households. Collection costs for landfills and incineration are typically equivalent.

Hauling cost

Hauling cost will occur if a transfer station is needed to deliver the waste to the final disposal site. Transfer stations are utilized to reduce transportation costs by using tractor trailers which can carry more waste than a regular garbage truck and only have one driver as opposed to an entire crew in a garbage truck. The cost is highly correlated to the distance to the final disposal site. The following parameters are needed to determine hauling cost (13):

1. Time based transportation costs
   - Tractor trailer costs
   - Driver salary
2. Mileage Cost
   - Fuel Cost
   - Oil and Tire Cost
   - Maintenance and Repair Costs
3. Transfer station capital costs and operating and

maintenance costs.

Operating and Maintenance Costs

These costs include expenses of day to day operation at a facility. Expenses include labor, utilities, equipment operation and repair, and administrative costs. Certain costs are unique to disposal facilities. Modern landfills have groundwater monitoring and cell development costs and incinerators have ash disposal costs. Incinerators are often not run by the community but rather by a firm who establishes a contract with the community where they are paid a specific rate (16). The firm is then responsible for some of the operating and maintenance costs. In general operating and maintenance cost for incinerators is greater on a per ton basis than landfills. In a small study carried out by the Office of Technology Assessment (OTA) for their report Facing America's Trash (31), incineration operating costs ranged from \$18-\$50/ton while at landfills operating costs ranged from less than \$3/ton to \$40/ton at a state of the art facility.

Capital costs

Capital costs include all the expenses of building a facility. They vary considerably from one type of facility to the next and one community to the next. The estimated cost of building a modern landfill in 1983 was 1.25 million for a 50 ton per day facility and 5.62 million for a 500 ton per day facility (13). Incinerator costs are difficult to generalize but tend to be approximately 4-10 times greater than a comparatively sized landfill (13). The 1986-87 Resource Recovery Yearbook reported adjusted capital costs (in 1986 dollars) ranging from $250,000 to $429 million for incinerators with an average of $58 million (31). Landfill costs are projected to increase more quickly than incinerators in the coming years as increased regulations are imposed increasing the difficulty of establishing and building approved disposal sites (13,30).

Typical capital costs include: land, site preparation, buildings, utilities, equipment, and planning expenses. Costs that are unique to landfills include a liner, leachate control system, and groundwater monitoring system. Costs unique to incinerators include: steam/power generation equipment and transmission lines for energy recovery facilities, air pollution control equipment, and start-up and acceptance testing expenses (13). Capital costs at a composting facility depend upon the level of technology utilized. The level of technology depends upon the space available for the composting operation and the speed with which it is desired to produce a compost product (34). Higher technology composting operations require more equipment to control moisture content, oxygenation, and temperature in the compost piles.

Question 6: The total cost can be calculated in dollars/ton and is based on factors discussed previously and summarized in Figure 5. For costs for services carried out by private companies the procedure is fairly straightforward because the expenses are in the form of tipping fee's and collection fee's which can readily be broken down into a cost per ton basis. For services owned and operated by the community, the capital expenses are amatorized over their lifetime into a yearly cost which can then be converted to a cost per ton value based on the tons of waste "handled" by the particular piece of equipment be it the incinerator itself or a trailer at the hauling station. Operating and maintenance costs can likewise be calculated for a one year period and converted to a dollar per ton value.

Overall costs for a management option are expressed in Equation 1. Cost on a per ton basis can thus be determined for the preferred management option and the normal management option. The cost to switch to the preferred management option is the tons of troublesome waste switched times the difference in cost per ton between the old management option and the management option of choice. Adding the cost of switching management options to the cost of separating out the troublesome waste gives the total cost of changing management options for a troublesome waste (Equation 2).

<u>Equations</u>

1. MC(X) = OM + Cap + Coll + Haul
   MC(X) - Management Costs for option X ($/Ton of MSW)
   OM - Operating and maintenance costs ($/Ton of MSW)
   Cap - Capital costs ($/Ton of MSW)

Separation Costs	+	New Management Option Costs − Old Management Option Costs (on a per ton basis)	=	TOTAL COSTS

Capital Cost

Collection vehicles
Storage Bins
Mechanized equipment
Planning expenses

Operating Costs

Labor
Fuel
Maintenance

Resident Costs

Time
Space

Change in Collection and Hauling Costs

Difference in distances to management sites

Change in Capital Costs

Land
Buildings
Equipment
Site preparation
Planning expenses

Change in Operating Costs

Labor
Utilities
Equipment maintenance and repair
Administrative costs

Figure 5. Summary of costs involved when changing management options for a troublesome MSW. All costs can be calculated in dollars per ton of waste.

Coll - Collection costs ($/Ton of MSW)
Haul - Hauling cost ($/Ton of MSW)

2. TOTAL COST =(MC)*(T) + SC
TT - Mass of troublesome waste (Tons)
SC - Costs of separating troublesome waste

The total costs are a function of the desired separation of the troublesome waste. With increasing levels of separation and thus greater reduction of health risk, there are higher costs. A separation efficiency/risk reduction versus cost graph can be generated for each of the separation alternatives. This is demonstrated in a hypothetical example in Figure 6. Utilizing the generated graph the community can determine what is the most cost effective method, based on what they are willing to spend, for reducing the risks posed by the troublesome waste.

In the event that costs are prohibitively high and risks are also high, the community may choose to ban the troublesome waste altogether. There are costs involved when a waste is banned. The value of the service provided by the product which eventually becomes the troublesome waste and the availability of alternatives to the troublesome waste must be determined. In some instances, the use of alternatives may also present risks which must be evaluated. Banning of a troublesome waste ordinarily should only be considered when there appears to be no alternative and the risk presented by the waste is unacceptable.

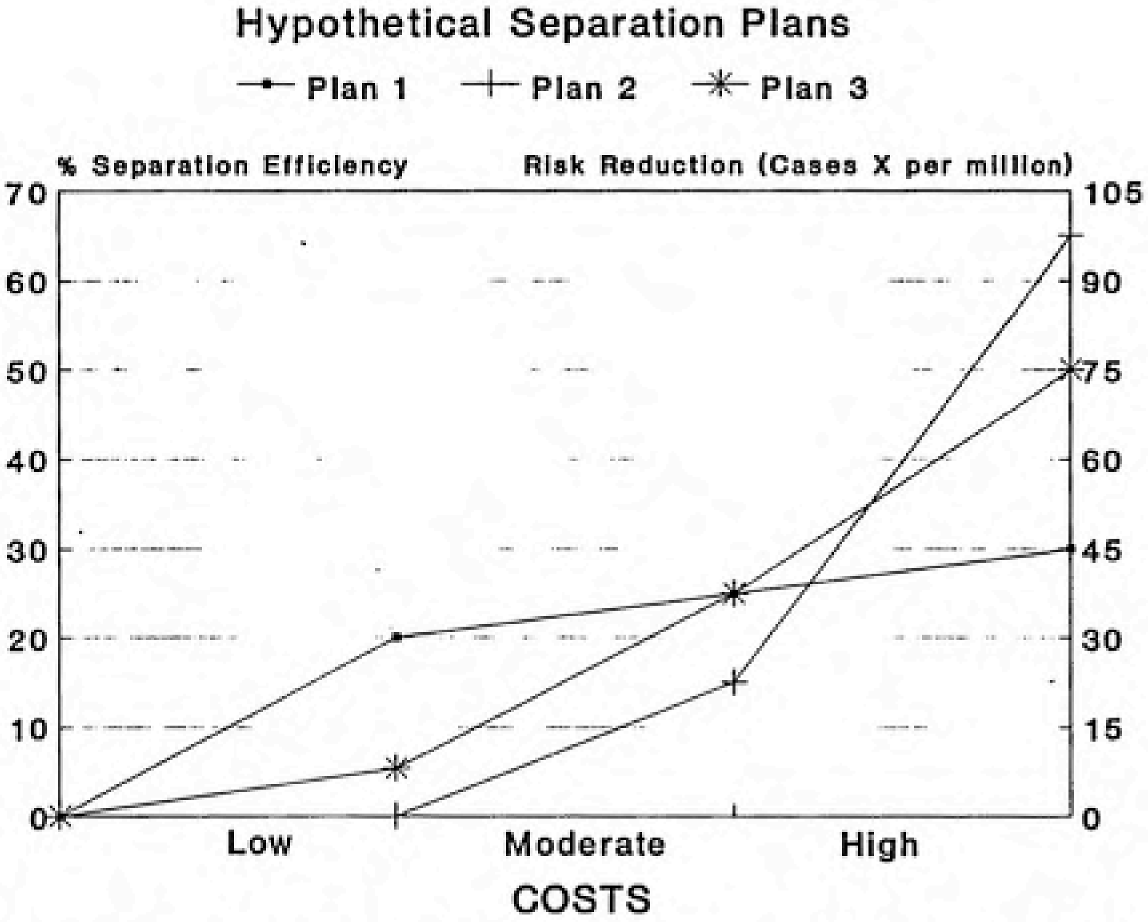


Figure 6. Separation efficiency and risk reduction versus cost for three hypothetical separation plans. The risk reduction is evaluated at the 85% point on the cancer case cumulative confidence curve in Figure 4. This point correlates to a health risk of less than 150 cancer cases per million lifetimes. Thus, with 50% separation efficiency, one can be 85% confident the risk reduction is less than 75 cases per million lifetimes.

Discussion/Case Study

Brominated Fire Retardants in Wilmington, NC MSW

Wilmington, NC is located in a small county - New Hanover -on the southern coastal area of the state. The county's population in 1980 was approximately 100,000 and was centered primarily in the southeast section of the county (43). New Hanover county has both a double lined landfill and a massburn steam recovery incinerator. The landfill and incinerator are both located in the northwest area of New Hanover close to both Pender and Brunswick counties which are more rural than New Hanover (see Figure 7). Presently the incinerator is being expanded to increase its capacity from 200 tons/day to 450 tons/day, and when the expansion is complete in 1991 the county intends to incinerate all MSW except unburnables such as concrete (4). Unburnables will be sent to the landfill along with incinerator ash (4). The landfill has a leachate removal system and the leachate is treated in a lagoon and pumped into the Cape Fear river (36).

Brominated organic compounds are among the most widely used and effective flame retardants. Hydrogen bromide, which is one chemical formed when a brominated organic compound burns, is one of the most effective agents to react with hydroxy radicals and similar species in flames, which are responsible for the propagation of fires (18). Brominated fire retardants (BFR's) are a class of chemicals

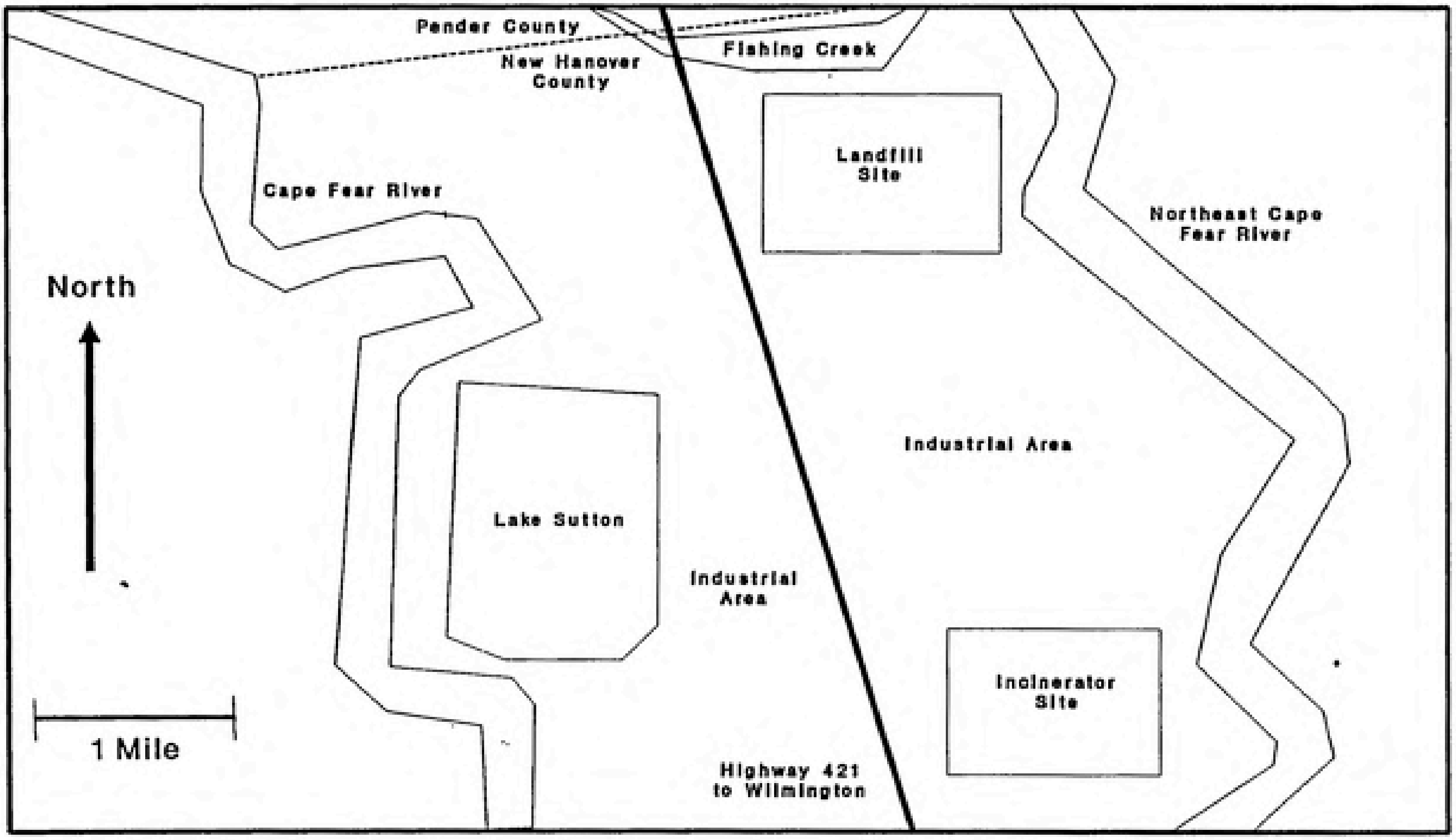


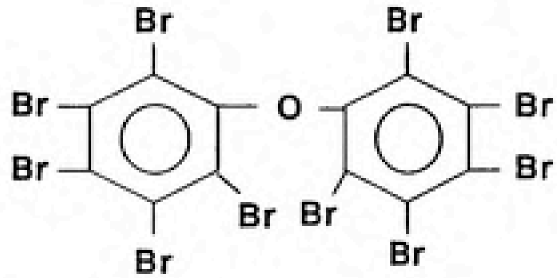
Figure 7. Approximate location of New Hanover landfill and incinerator.

added to many different products. BFR's such as biphenyl and diphenyl ethers are routinely added in 4-20% levels into plastics used in textiles, carpets, furniture and construction materials (8,37). Pyrolysis of these flame retardants is known to produce polybrominated dioxins and furans (PBDD's and PBDF's) and these compounds have been found in incinerator ash (37,18,8).

It is ironic that complex chemical additives intended to retard burning form a potentially extremely toxic substance when they are incinerated at high temperatures. Their chemical structure lends itself to a complex array of reactions that result in the formation of PBDD/F's. PBDD/F's can contain from 1 to 8 bromines as well as a combination of chlorines and bromines. The structure of a typical BFR and the resulting possible PBDD/F's is demonstrated in Figure 8. Chlorinated dioxins and furans are known to be hazardous and pose potentially significant health threats and it is thought that PBDD/F's have equal or possibly greater toxicity (14,38).

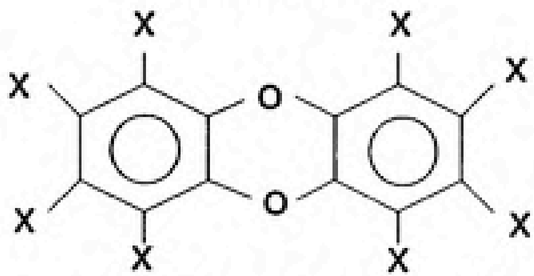
The question to be answered is what is the best method available for New Hanover county to manage substances containing BFR materials. The initial focus will be on determining the health effects of sending the materials to the landfill or the incinerator. Because BFR's are added to plastics that make up a wide variety of products, they do not immediately make easy candidates for recycling or banning. The options of recycling and banning will be

Typical Brominated Flame Retardant
Decabromo-diphenyl ether

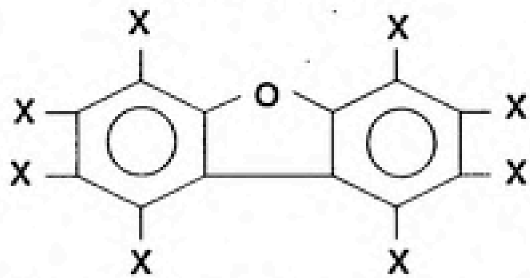


Brominated Dioxins/Furans (Contain at least one bromine)

Generic Dioxin



Generic Furan



Each site X may contain a bromine, chlorine, or hydrogen

Highly studied chlorinated dioxin
2,3,7,8 Tetrachloro-dibenzo-p-dioxin
(2,3,7,8 TCDD)

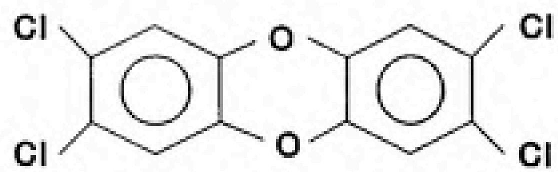


Figure 8. Brominated flame retardants and formation of dioxins and furans as a result of incineration.

addressed in more detail if merited by health and cost considerations.

EXPOSURE EVALUATION

Table 2 is a breakdown of exposures as a result of sending BFR's to a landfill or an incinerator.

Table 2: Exposures from Landfilling and Incinerating BFR's

Management Option	Exposure
1. Landfill BFR's	-BFR's leaching into groundwater. -BFR's pumped into surface water
2. Incinerate BFR's	-Air contamination of PBDD/F's -Biota contamination of PBDD/F's -Surface water contamination of PBDD/F's -PBDD/F's from incinerator ash leaching into groundwater

BFR's and BPDD/F's in groundwater

In order to determine what the extent and spread of contamination from the landfill into the groundwater would be, the groundwater model MYGRT 2.0, developed by the Electric Power Research Institute, is used. This model is adequate to provide information for a typical community to make a decision based on possible groundwater contamination. It is a two dimensional analytic model allowing for a planar analysis of the groundwater plume. The model allows a decision maker to evaluate several different conditions for groundwater transport, but at the same time is relatively simple to apply. The additional reduction of uncertainty provided by a three dimensional numerical model may be useful when potential costs of a decision are high, but for a ordinarily a two dimensional model is satisfactory. Numerical models require much more detailed data that is rarely obtained when siting landfills.

MYGRT 2.0 is a based on the advection-dispersion-retardation-decay equation (general transport equation). Its assumptions as summarized in the MYGRT 2.0 user manual (9) are listed below:

1. Parameters input to the model such as the groundwater seepage velocity remain constant throughout the aquifer.
2. Sorption is treated as linear, equilibrium partitioning between aqueous and solid phases.
3. Interactions between chemical species are not considered.
4. First order kinetics adequately simulate solute transformation or decay, and the decay rate is the same for solutes present in either solid or liquid phases.

In carrying out the groundwater exposure analysis, it was assumed that the liners remained essentially intact with respect to BFR's and PBDD/F's for 50 years (33) during which time the leachate was all pumped to the lagoon. After liner failure, it was assumed that leachate escaped into the groundwater. Groundwater contamination was determined for a period of 50 years after the failure.

Because of their chemical similarity the groundwater parameter values for BFR's and PBDD/F's were assumed to be equivalent (refer to Figure 8). Both BFR's and PBDD/F's tend to adhere to solid particles and have extremely low water solubility. The supply of BFR's and PBDD/F's in the soil is primarily from discarded BFR's and incinerator ash respectively. Concentrations of these substances in the soil are likely to be much greater than their water solubility and therefore it was assumed that the soil provided a continuous supply of BFR's and PBDD/F's to the groundwater for the time span considered. Thus, a steady state leachate concentration was reached that was maintained for a considerable length of time due to the relatively high supply of BFR's and PBDD/F's that had accumulated in the soil.

Concentration values were calculated out to a range of 500 meters, and are graphically displayed in Figure 10 for 167 meters and 500 meters. In carrying out sensitivity analysis on MYGRT 2.0 using data from New Hanover County (36), beyond 500 m the concentrations of both PBDD/F's and BFR's approached zero for all scenarios. Also, the direction of groundwater flow is towards surface waters located approximately 500 m or less from the landfill edge, and a considerable portion of the groundwater aquifer deposits into these surface waters and is highly diluted (36). For these reasons, it was assumed that beyond 500 m from the landfill site all PBDD/F and BFR concentrations in the groundwater were zero.

The concentrations at 167 m and 500 m predicted by the model are directly downflow of the contamination source. These calculated values were assumed to be indicative of groundwater concentrations at their respective distances at any point in the groundwater plume which generally flows in a northeast direction from the landfill (36). This assumption is valid because of the source of contamination is from a large area (the landfill) rather than a point. Because of the time scale involved (final concentrations evaluated at 100 years from the present time), it is difficult to estimate the population that will be effected by the groundwater plume. However, in two dimensional land area, the groundwater plume is approximately 2000 times smaller than the air plume from incineration, and therefore it was assumed that the groundwater contamination will effect one two-thousandth the population as the air plume.

By performing sensitivity analysis with the model, the following parameters were determined to be the most critical in calculating groundwater contamination with BFR's and BPDD/F's: seepage velocity, decay rate, concentration in leachate, and adsorption coefficient (retardation). These parameters were given values and assigned probabilities as summarized in Figure 9.

The seepage velocity values were obtained from a groundwater boring studies carried out by Soil & Material Engineers Inc. (36) before the landfill was constructed. Parameters such as hydraulic conductivity, soil type, porosity, aquifer depth, and horizontal gradient were determined at various boring sites on the proposed landfill

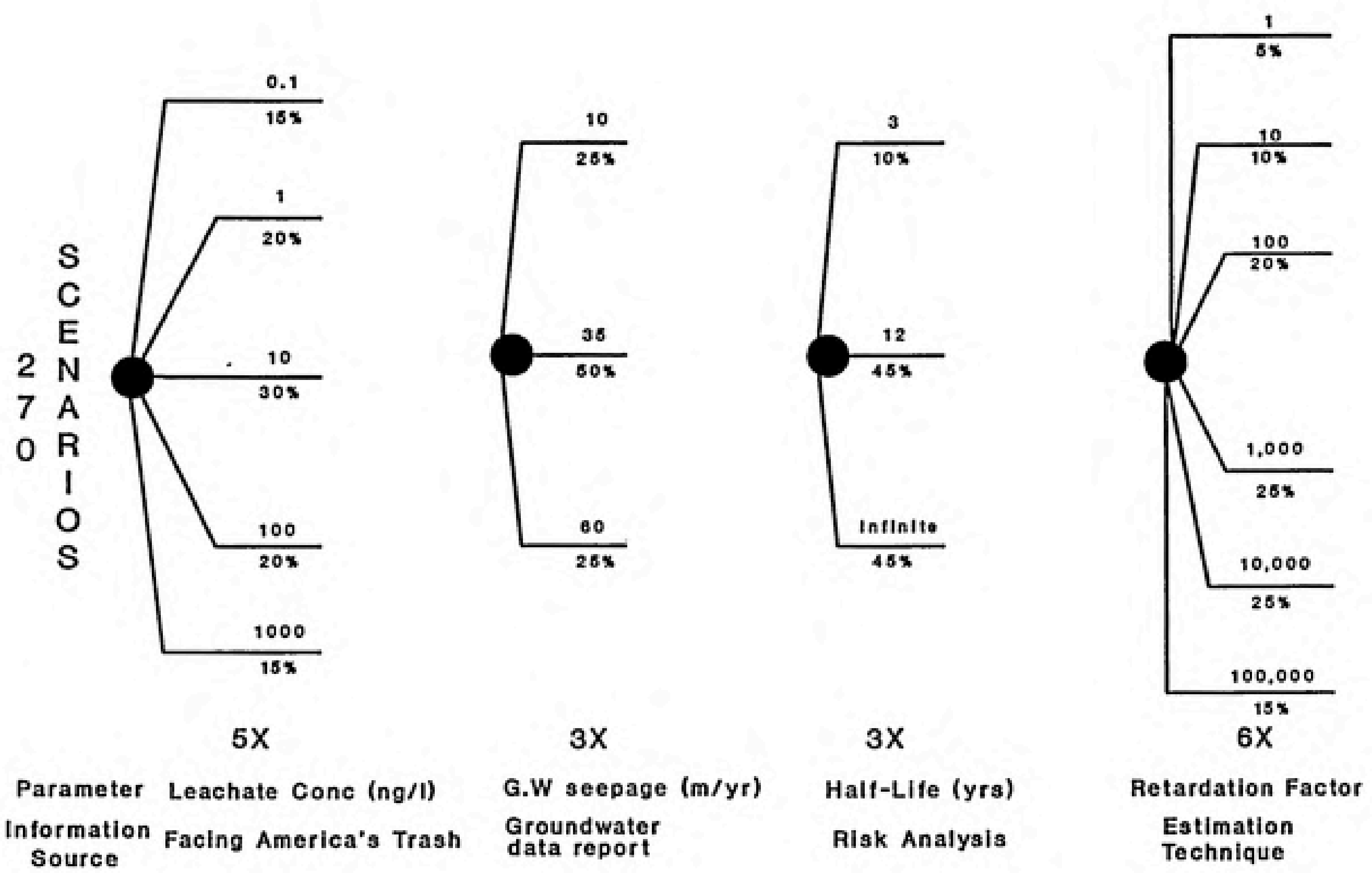


Figure 9. Uncertainity trees for parameters used in groundwater modelling. Discrete parameter values with assigned probabilities are used to predict BPDD/F or BFR contamination in groundwater surrounding New Hanover landfill.

site and used to estimate a range of seepage velocity. An approximate range of 10 m/s to 60 m/s was estimated by Soil and Material Engineers Inc. These values represent the extremes of the range and were thus each assigned a probability of 25% while the midpoint, 35 m/s, was assigned a probability of 50%. Three values for seepage velocity were considered adequate to express the range of possible values.

Data for the decay rate for BFR's and PBDD/F's in soil is not known so it was assumed that data for 2,3,7,8 TCDD (TCDD) is indicative of soil decay for PBDD/F's. Initially the rate of decay for TCDD was thought to be 3 years (29); however long term observations at sites containing high TCDD concentrations indicate a half-life of approximately 12 years (14). Nevertheless, data on TCDD half-life is limited and to take into account that BFR's and BPDD/F's may have a slower decay rate than TCDD the possibility of no decay was entered into the model. Because recent data contradict, a half-life of 3 years was assigned a probability of only 10%. A likelihood of 45% was assigned to a half-life of 12 years because this value has been reported in separate studies (14). A likelihood of 45% was also assigned to no half-life to adequately and conservatively represent the remaining uncertainties. The likelihood assignments given to the potential decay rates are an example of a methodological approach for representing conflicting information.

Leachate concentration values were estimated using published data for leachate concentrations of dioxins and furans (31) and by calculating what the maximum solubility of BFR's and PBDD/F's are in water using estimation techniques (21). The published data are limited and estimation techniques are subject to error. Potential factors that increase uncertainty in determining the concentration are the organic content of the leachate (which will tend to increase the solubility (21)) and the possibility of the soils actually moving fluidly (36), and thus BFR's and PBDD/F's adhered to soil particles being carried by the leachate. The possibility for higher leachate concentrations than have been measured or that solubility estimation techniques suggest was considered to account for these uncertainties. Refer to Figure 9 for the range of possible values and their assigned likelihoods. It should be noted that 5 different values were used to represent the considerable uncertainty in leachate concentration values.

The retardation value is an expression of the rate at which a chemical adsorbs to solid particles and thus removed from the groundwater flow. Retardation values can be calculated using octanol water coefficient values which can be estimated using a substitution technique based on a known octanol water coefficient value for a similar chemical (21). Using a Kow value for TCDD, bromines were substituted for chlorines and the appropriate adjustment to the Kow value was determined to estimate a Kow value for PBDD/F's. The same procedure was followed using diethyl ether and substituting bromines for the hydrogen groups. The calculated Kow values for both PBDD/F's and BFR's were similar with the log Kow of both having a range of 6.5-7.5. The Kow values were used to give an estimation of the retardation value.

The percent of organics in the leachate is a primary factor that may effect the retardation value. To take into account the possible effect of high organics in the groundwater a 5% probability of a retardation of 1 was assigned. A retardation of 1 is many orders of magnitude less than predicted by estimation techniques and results in considerably higher contamination values downflow of the landfill. It was assumed that beyond 500 m downflow organic concentration in the groundwater was reduced so that the retardation value for PBDD/F's and BFR's increased to a value closer to that predicted by the estimation technique. Refer to Figure 9 for the values assigned to retardation and their associated likelihoods.

Using these assigned values as summarized in Figure 9, there were 270 possible scenarios and each was evaluated using the MYGRT model. To convert from groundwater to human consumption, an intake value of 2 liters per day of direct ingestion was assumed. Through the use of LOTUS 123 the data values were arranged from the lowest to highest values with their corresponding likelihood. This is graphically represented in Figure 10 in a cumulative probability curve for contamination at 167 m and 500 m.

<u>Air exposure to PBDD/F's</u>

The Industrial Source Complex Long-Term (ISCLT) model was applied to estimate the average PBDD/F concentrations within a 25 km radius of the incinerator. The model is a steady-state Gaussian plume model requiring several input parameters related to the source as well as weather and wind data at the site. It calculates an average annual concentration at several distances and directions from a source. In determining the air exposure of PBDD/F's, it was assumed based on studies presently being carried out by Rich Kamens at the University of North Carolina at Chapel Hill, that PBDD/F's do not decay in the air.

The primary uncertainty in the air modeling was the stack exit concentration and the capacity at which the plant would be operating, which has an effect on the physical shape of the plume. It was assumed that PCDD/F's concentrations are an indication of what PBDD/F's concentrations will be. Observed values for PCDD/F's concentrations have been recorded in several sources (15,29,31). These values were used to determine a probability range for PBDD/F's. Measurements of PCDD/F's

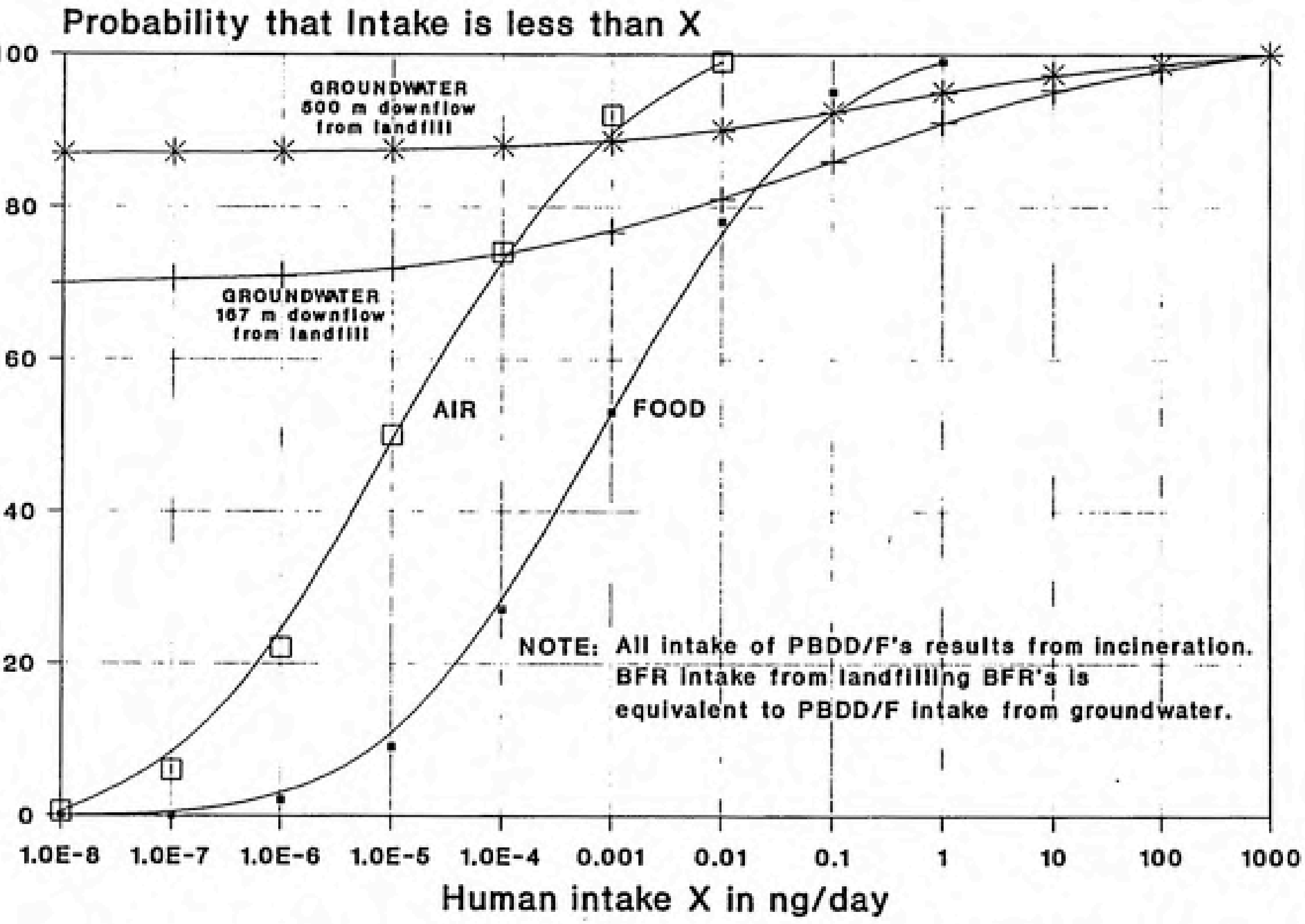


Figure 10. Cumulative probability curves of human intake of PBDD/F's from groundwater, air and food.

that are published often are obtained from incinerators that are new and running at optimal conditions (7). These values may be an underestimate of typical values and this was taken into consideration when developing a range of values and their likelihoods. Information on the New Hanover incinerator operating performance (4) was used to assign values and likelihoods for incinerator stack conditions.

Concentrations were determined in four directions from the incinerator - north, south, east and west. In Wilmington the predominant wind direction is north, followed by south, west and finally east (12). The model predicts higher concentrations in predominant wind directions. The present day population levels in each direction are assumed to be indicative of future population levels in areas surrounding the incinerator. The 1980 population level and agricultural activity percentages obtained from census data (43) for each area around the incinerator are summarized in Table 2. Agricultural activity information is needed when determining food contamination and will be discussed in the food exposure section.

Using the ISCLT model air concentrations were evaluated at distances of 1, 3, 5, 10, 25 km from the incinerator. It was assumed that population density was constant over the entire area of a given direction. There are a total of 240 scenarios for air exposure. Refer to Figure 11 for a summary of all the input parameters and their corresponding

TABLE 3: Population Levels and Agricultural Activity in a 25 km radius around The New Hanover Incinerator

DIRECTION	POPULATION LEVEL (%)	AGRICULTURAL LEVEL (%)
North	7	55
South	60	10
East	25	10
West	8	25

probabilities. As with the groundwater concentration, each case is input into LOTUS 123 to derive a cumulative confidence distribution curve (Figure 10). Using a standard breathing rate of 20 m /day the air concentrations were converted to human intake.

Surface water exposure

Sources of surface water contamination are leachate pumped from the leachate lagoon, groundwater returning to the surface water and air deposition. All of these sources are assumed to be minimal because they become highly diffused after entering surface water, which in New Hanover county tends to flow into the Atlantic Ocean. It is assumed that exposure to PBDD/F's from surface water is negligible.

Biota exposure

Dioxins have a very high octanol water coefficient and therefore have a tendency to accumulate in fatty tissues.

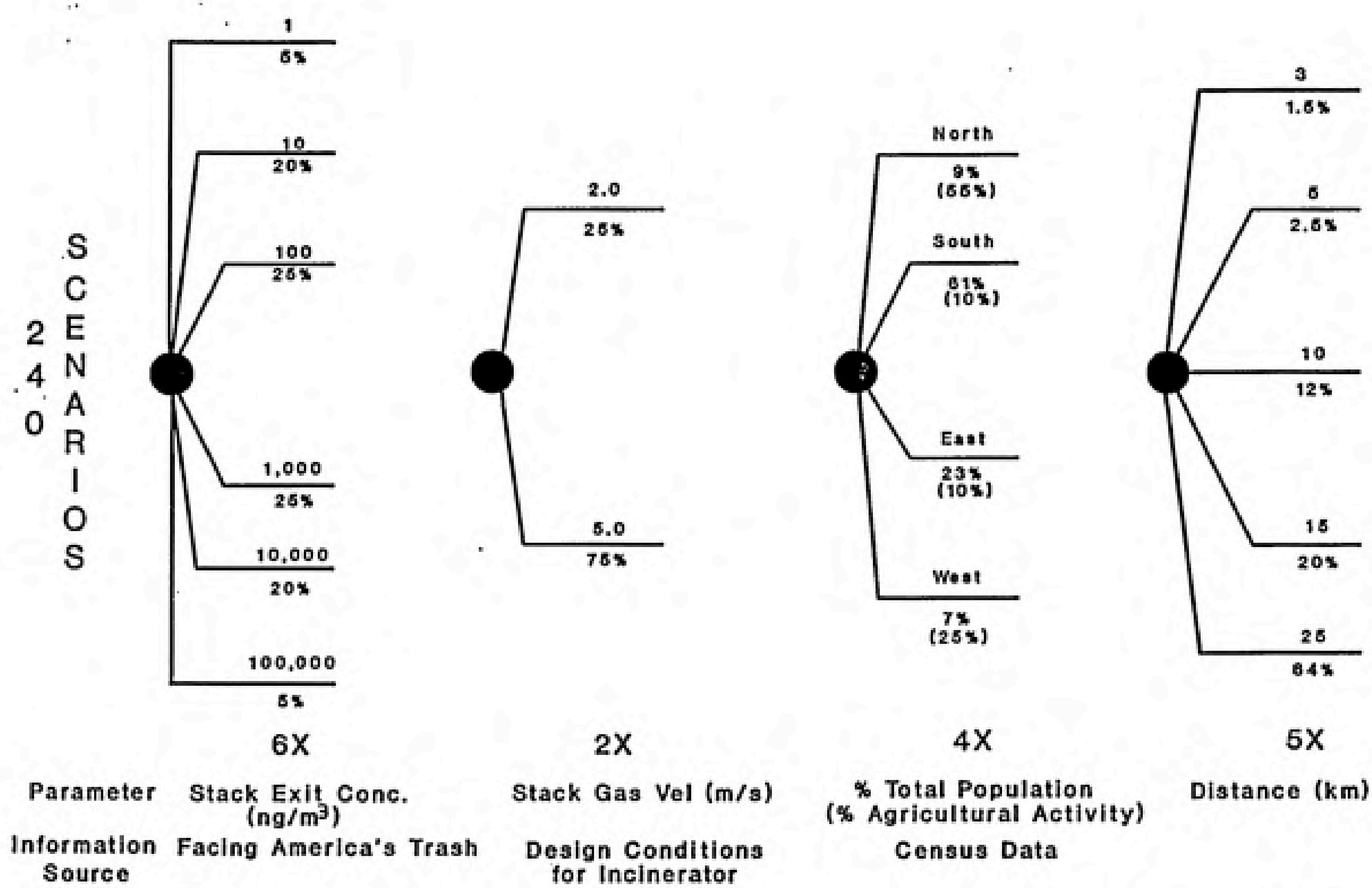


Figure 11. Uncertainity trees for parameters used in air modelling. Discrete parameter values with assigned probabilities are used to predict a range of PBDD/F air exposure to both human and cow populations in New Hanover county as a result of incinerating BFR's.

For this reason, exposure from food must be considered. In order to carry out an assessment of food exposure, two techniques as outlined in reports by Stevens and Gerbec (38) and Travis and Hattemer-Frey (41) were used. Each technique followed a slightly different procedure and therefore resulted in different values for certain food exposure parameter values. Both reports assessed total exposure to 2,3,7,8 PCDD from food starting with a single air concentration and converting that to soil and vegetation contamination, then cow intake and finally human intake. It was assumed that PBDD/F's behaves similarly to 2,3,7,8 PCDD.

A range of possible air contaminations were determined in the same manner as described in the air exposure section with one exception. The agricultural activity percentage listed in Table 2 was used as opposed to the population level. The agricultural factor is indicative of food production for an area in a given direction from the incinerator. The areas north and west of the incinerator are much more rural than those to the east and south and thus the effect of contamination in these directions was given greater weight to account for the higher levels of food production occurring there. As in the air exposure evaluation, 240 scenarios were evaluated. The air contamination predicted in these scenarios were sorted from least to highest. Eight representative values to input into the food intake calculation were obtained by averaging the sorted air contamination levels in groups of 30. The probability for each value was equal to the cumulative probability of the 30 contamination levels. Refer to Figure 12 for a listing of the eight representative values and their corresponding likelihoods.

Soil concentration was determined by multiplying each air concentration by a constant. The constant is based on the assumption of continuous and constant deposition onto the soil for a 70-year period of time and a PBDD/F half life of 12 years in the soil (38). Different types of foliage have different levels of deposition dependent upon their leaf shape and surface area. The grasses and hay consumed by cows have higher levels of deposition than vegetation consumed by humans (38,41). It was assumed that vegetation does not uptake any of the PBDF/F's from the soil.

The standard cow and human diets listed in Table 3 were used.

TABLE 4: Standard Cow and Human Daily Diets

Cow Diet Substance--Ingestion	Human Diet Substance--Ingestion
Soil--130 g	Milk/Milk Products--600 ml
Grasses--6 kg	Beef--140 g
Corn Silage--15 kg	Leafy vegetables--100 g
Grains--5 kg	(Other foods with no contamination)

In the cow diet, the grains were assumed to contain no PBDD/F's (38,41). It was assumed that all the PCDD/F's

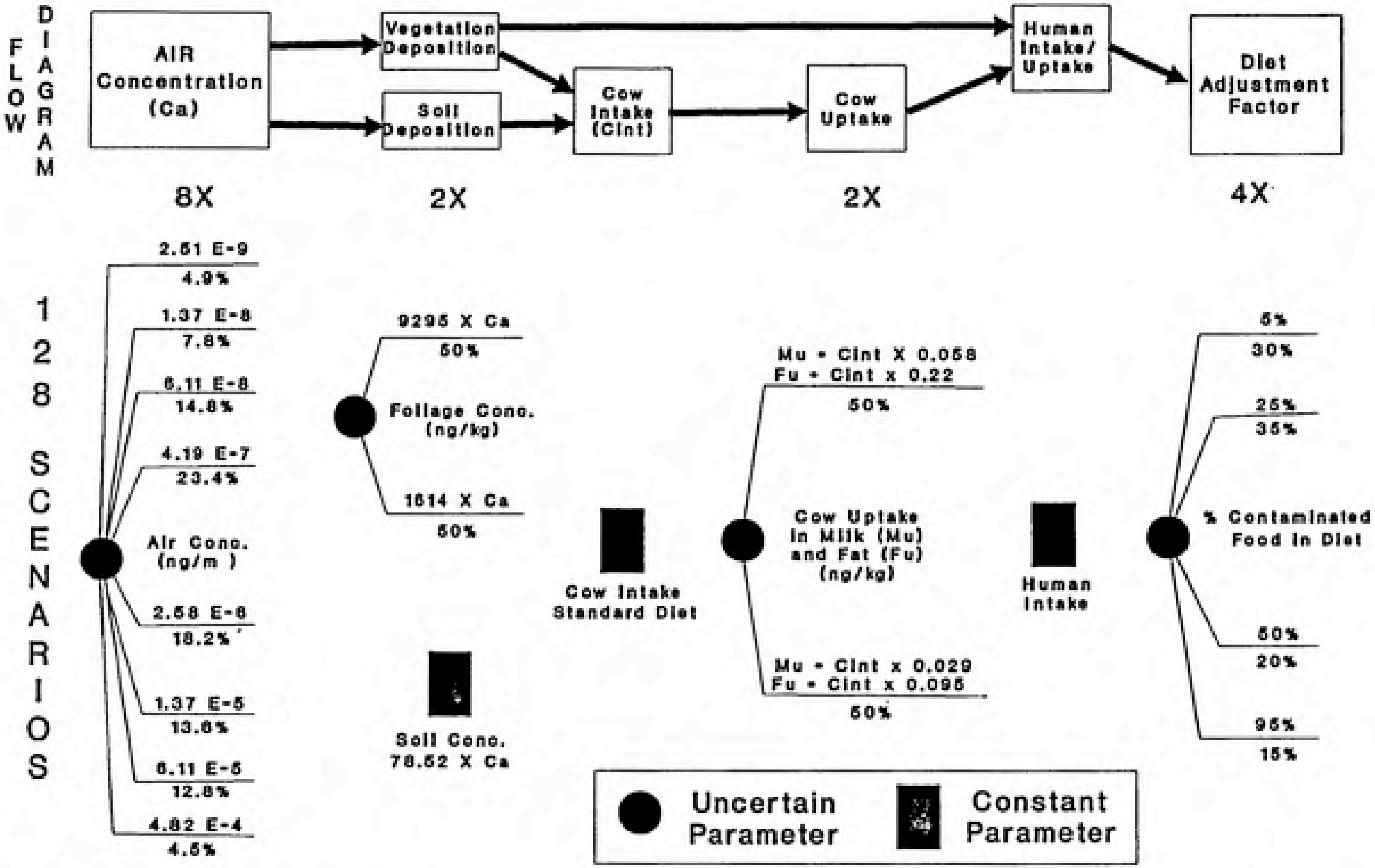


Figure 12. Uncertainity trees for parameters used to predict human intake of BPDD/F's from food contaminated as a result of incinerating BFR's.

ingested by cows migrates to their milk and fat. Refer to Figure 12 for the different possible uptake values in milk and fat.

The sources of PCDD/F's in the human diet are assumed to be leafy vegetables, cow meat, and cow milk. In the study by Stevens and Gerbec, accumulation of TCDD in chicken, eggs and pork is much lower than that of beef in milk. Accumulation does occur in lamb (38), and certain types fish (41) but these foodstuffs were assumed not to be large components of a typical human diet. It should be noted that the majority of fish likely to be consumed in New Hanover County would be salt water fish and these will not have as high levels of contamination as fresh water fish. Thus, it was assumed that fish consumption does not significantly contribute to overall PBDD/F intake. For humans, because of minimal data, it was assumed that uptake was 100%. This is not an extremely conservative assumption however, because animal studies have measured an uptake of 60% for 2,3,7,8 TCDD (38). The final step in food exposure analysis was to estimate what percentage of the human diet consists of foodstuffs contaminated by PBDD/F's as a result of the New Hanover incinerator. In Figure 12, the estimated percentage contaminated diet values and their likelihood are given.

There were a total of 128 scenarios for food exposure and the probability curve for intake from these scenarios is graphed in Figure 10.

HEALTH IMPACTS

From the range of exposure calculated an assessment can be made of the health risk of landfilling and incineration of BFR's. Conservative assumptions are made resulting in a health risk evaluation which is likely to be a worst case situation. When BFR's are landfilled, the only source of exposure to a potentially hazardous substance is from BFR's in groundwater. When BFR's are incinerated exposure to PBDD/F's in groundwater, air and food can occur. The range of intakes from these exposures is graphed in Figure 10. Through the use of health risk guideline values the potential risk these intakes pose was assessed.

Due to lack of information otherwise, it was assumed that all PBDD/F's have equivalent toxic effects as 2,3,7,8 TCDD which has a VSD value calculated by the EPA of 0.1 pg/kg/day (14). This is a significant assumption and more information on the health effects of brominated dioxins and furans will have a high value. The assumption is significant because 2,3,7,8 TCDD is the most toxic of all 75 known chlorinated dioxins and 135 known chlorinated furans (22). The EPA has devised a method for assigning a 2,3,7,8 TCDD toxic equivalency factor to dioxin and furan isomers (22). Such a method could be applied to PBDD/F's if a typical isomer distribution leaving the stack was generated.

This would result in a lower level of risk. In this initial case study, a toxic equivalency method is not used. This is done because if based on the assumption that all PBDD/F's have a toxicity equivalent to 2,3,7,8 TCDD there is no significant risk, than no further study is needed. However, if there does appear to be significant risk than the use of the equivalency factor method may be justified.

There is limited information on the toxicity of BFR's. Decabromodiphenyl ether has shown low acute toxicity in several animal studies involving different exposure routes (45). Some studies have shown liver toxicity as a result of chronic exposures. The lowest level of exposure at which liver toxicity was observed was 80 mg/kg/day (45). Carcinogenesis bioassays indicate that the liver is also the major target organ for carcinogenicity; however, the majority of tumors were benign. These studies do suggest a health risk from BFR's. However based upon the lowest exposure levels from decabromodiphenyl ether at which liver toxicity was observed, the risk from BFR's is several orders of magnitude less than that from the PBDD/F's for equal exposures. The groundwater modelling predicts equal levels of exposure to either BFR's if the BFR's are landfilled or PBDD/F's if the BFR's are incinerated (see figure 10). Since the exposure levels are for very low concentrations, it was concluded that the health risk from BFR intake was negligible.

The intake range for each exposure route to PBDD/F was divided by the VSD value of 0.1 pg/kg/day to determine the cancer risk for each exposure. The models predict exposure to PBDD/F's only as a result of incinerating BFR's. In Figure 13 the increased likelihood of cancer from exposure to PBDD/F's in groundwater, food and air as a result of incinerating BFR's is graphed. In this graph, the wide range and potentially high risk of groundwater exposure is demonstrated. At 167 m downflow from the landfill, there is an 18% chance of a 1 in a million lifetime cancer risk, and a 2% of a 1 in a 100 lifetime cancer risk from groundwater. At 500 m downflow, there is a 9% chance for a 1 in a million lifetime cancer risk and a 1% chance for a 1 in a hundred lifetime cancer risk. Risk from air exposure is low with only a 4% chance of exceeding a risk of 1 in a million and a maximum possible risk of approximately 10 in a million. On the other hand, there is a 26% chance of greater than 1 in a million lifetime cancer cases from food intake. The maximum possible food intake risk is approximately 1 in a thousand chance of cancer over a lifetime.

In Figure 14, the population exposed from groundwater air and food contamination is taken into consideration. When population exposed is considered, the overall risk from groundwater is much smaller. With a population of 1 million there is only a 2% chance of 1 additional cancer case over a lifetime from groundwater intake. This graph suggest that

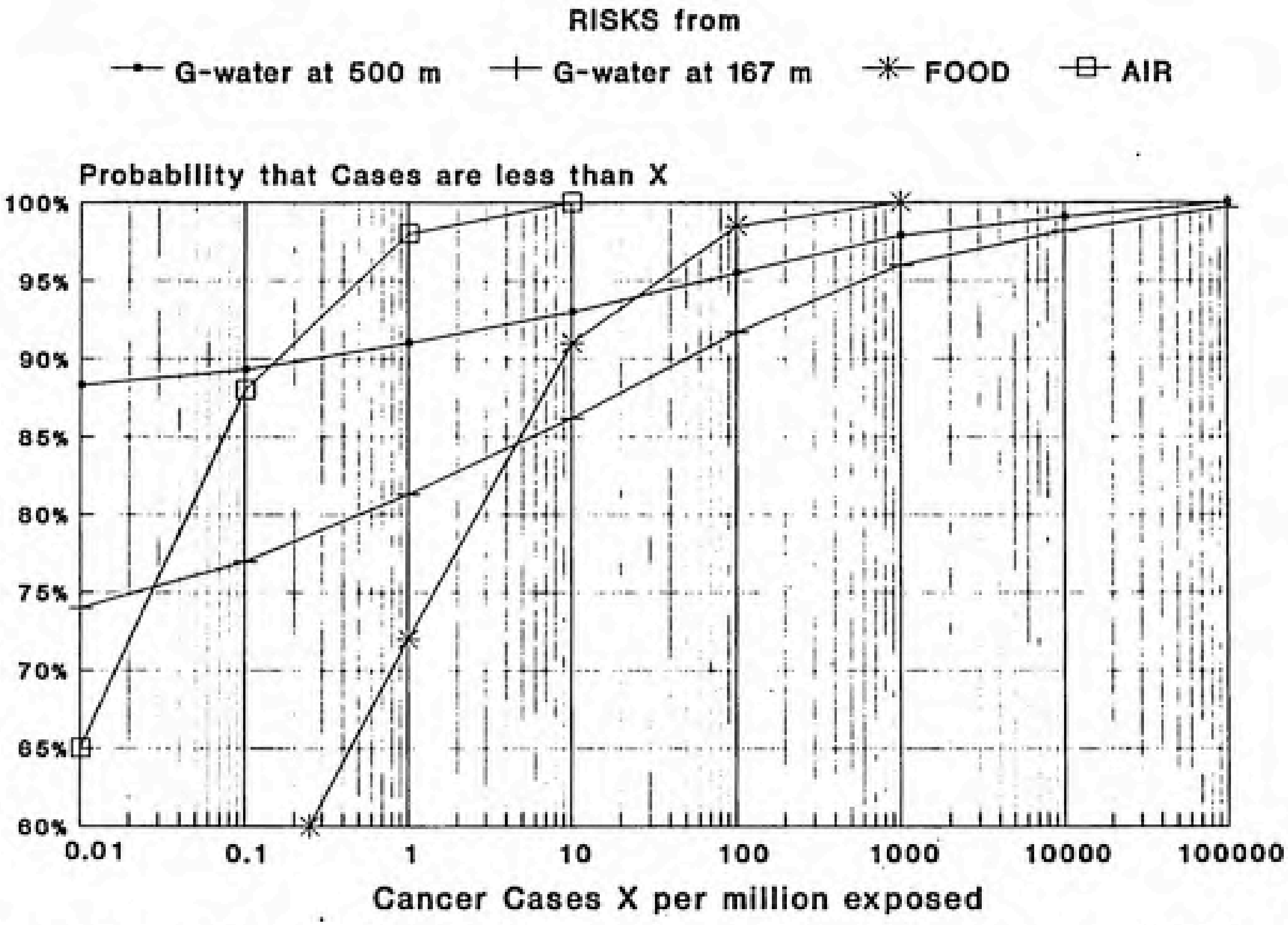


Figure 13. Cumulative probability curves for cancer risk to individuals exposed to PBDD/F's from groundwater, air and food. The health standard used is the US EPA's VSD value for 2,3,7,8 TCDD which is 0.1 pg/kg/day.

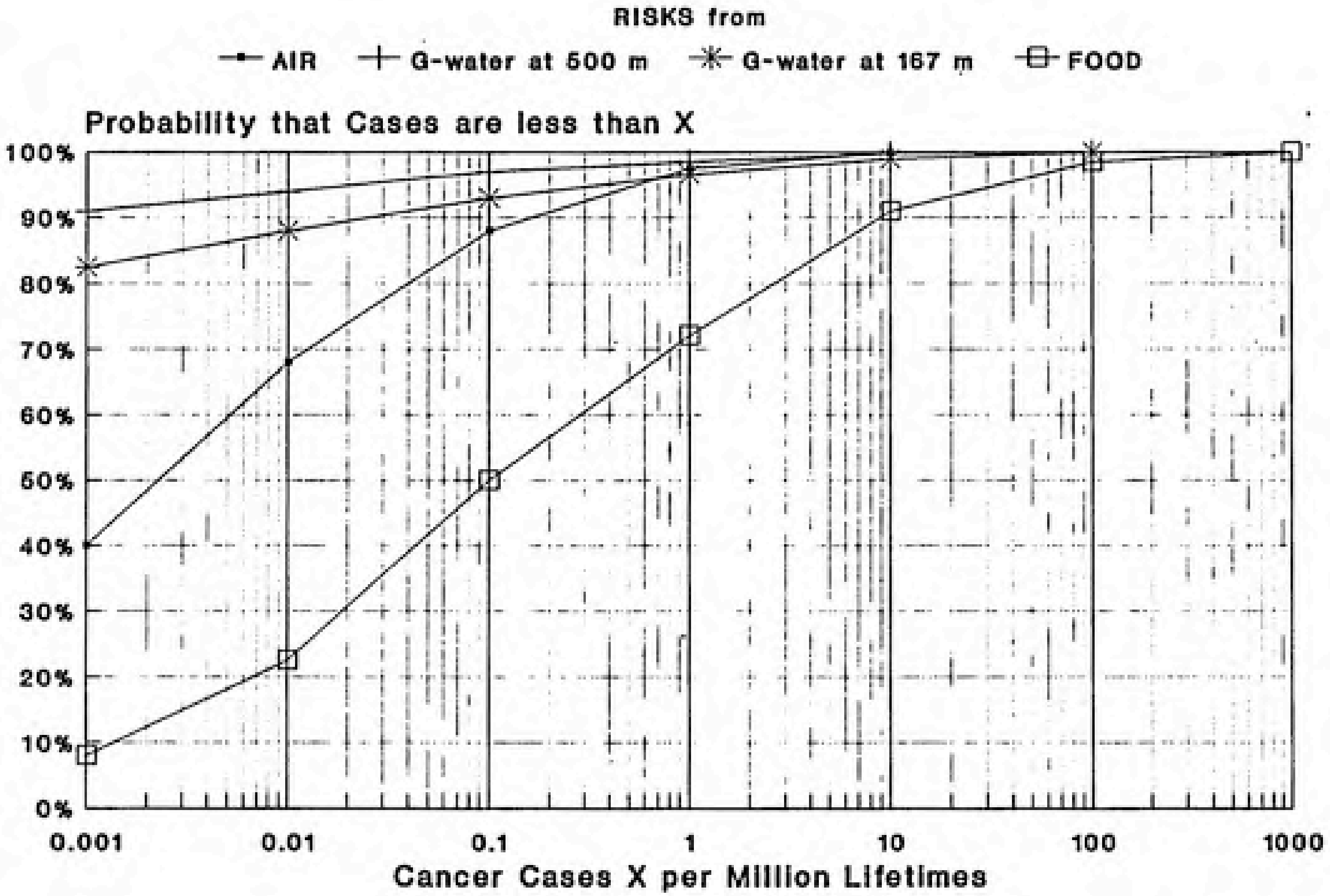


Figure 14. Cumulative probability curves of cancer cases from each exposure route. Groundwater is estimated to expose 1/2000th of the population exposed to air and food.

the primary risk from incineration BFR's comes from food exposure to PBDD/F's. From Figure 13, it can be concluded that if individuals are living within 500 m of the landfill and consuming the groundwater there they have a potentially significant risk of cancer during their lifetime; however, this risk can be prevented by obtaining drinking water from a source other than groundwater near the landfill.

The following conclusions can be made in regards to managing BFR's in New Hanover county's MSW.

1. Landfilling of BFR's results in potential intake of very small quantities of BFR's in groundwater, and based on present toxicity data poses no apparent health risk.

2. Incineration of BFR's results in exposure to PBDD/F's from groundwater, air and food.

3. Intake of groundwater at 167 m and 500 m has 18% and 9% chance respectively in resulting in a cancer risk of greater than one cancer case per million exposed over a lifetime, and has low probabilities (less than 5% at both distances) of risk of over one cancer case per one thousand exposed.

4. Risk from food intake is on average 10 to 100 times greater than that from air intake. Food intake risk has a 22% of being greater than one in a million, a 11% chance of being greater than 10 in a million and a 2% chance of being greater than 100 in a million cancer cases per lifetime of those exposed.

5. The overall population exposed to contaminated groundwater is estimated as one two-thousandth that of air and food resulting in an overall relative risk less than that of food exposure. Exposure from groundwater also can be prevented by consuming water from other sources.

6. In general, intake of PBDD/F's from groundwater and air as a result of incinerating BFR's poses a small health risk, but intake from food poses a potentially significant (a 22% chance of increased cancer risk of greater than 1 in a million) health risk.

7. Because of the potentially significant risk from food intake of PBDD/F's it is recommended that material containing BFR's are sent to the landfill.

Because there appears to be a legitimate health risk from incinerating BFR's, comments regarding recycling and banning are called for. Presently the option of returning materials containing BFR's to their original manufacturers where they can be recycled does not exist. Thus once a material containing BFR's is produced it will eventually become part of the solid waste stream that needs to be disposed. However, many of the materials containing BFR's (rugs, furniture etc.) are such that they can be reused and their entry into the solid waste system delayed. This will in effect reduce the demand for new products containing BFR's and thus reduce the rate at which these products will enter into the market. Thus, at present the recycling of BFR's should focus on reusing those materials for which there may be a demand. Banning BFR's in a community is presently not advisable for the following reasons:

1. BFR's are used in a wide array of products.
2. The health risks for sending BFR's to a landfill is small.
3. The costs of finding substitutes may be great, the substitutes may not retard fires as effectively, and may pose health risks of their own.

It is recommended when possible to reuse materials containing BFR's. If it is not feasible to reuse these materials, they should be routed to the landfill.

This recommendation supports the premise put forth earlier in the paper that a community should have several options for managing MSW. It presents an actual case where sending a material to a landfill is preferable to incinerating it. This is contrary to the present thinking of many on solid waste management who feel that landfilling should be eliminated as much as possible.

COST CONSIDERATIONS

In order to make a decision on whether or not to implement the preferred management option, the cost of implementing the option and the reduction in health risk it offered were evaluated. The total amount of solid waste produced in New Hanover County is approximately 165,000 tons per year (4,5). Assuming 10% of this waste is plastic and 1% of the plastic contains BFR's approximately 165 tons of substances containing BFR's are disposed yearly. The cost of incinerating waste is slightly higher than that for sending waste to the landfill in New Hanover County (4). It is estimated that there will be a saving of $10 per ton of waste. Because there are only approximately 165 tons of waste containing BFR's this savings is not significant.

As previously mentioned, the goal for New Hanover County is to send all solid waste except unburnables to its incinerator. Since the preferred management option determined when considering health impacts is to send BFR's that are not recycled to the landfill, this will necessitate somehow separating out materials containing BFR's before they are incinerated. One factor that has a significant impact on choosing a separation scheme for New Hanover county is that presently in unincorporated areas (which includes most of the county except the city proper and some of the beach communities) there are many private operators whom offer waste collection services (4,5). Efforts to coordinate waste collection so that specific operators have an assigned area in which to collect have failed (5). In this way, there is little control or regulation over routes and collection practices for much of the MSW in the county.

Taking the local characteristics into consideration the following separation schemes are suggested for evaluation:

Plan 1: Separate out suspected waste containing BFR's at the incinerator and reroute to the landfill.

COMMENTS
- Little burden on residents and business
- There is room to do this at the incinerator and plans have been made for separating recyclable MSW in this way (5).
- The landfill is close by so there will be little additional transport cost
- May enable some materials to be recovered for reuse

Plan 2: Pickup BFR waste in normal truck with a trailer attached.

COMMENTS
- Little burden on residents and business other than separating out BFR waste before collection
- Will not cause considerable slow down a waste collection
- Will enable some materials to be recovered for reuse
- Plan will have to be coordinated with private collection operators

Plan 3: Use a separate truck to collect waste

COMMENTS
- Collection schedule will have to be communicated to residents

- Program can be carried out without relying on private collection operators
- Costs are likely to be high

Plan 4: Delivery of waste to central collection facility by residents

COMMENTS
- High individual resident and business burden
- Low initial cost

Plan 5: A combination of Plans 1 and 4.

The following plans are plotted on a separation efficiency versus cost graph (Figure 15). This graph is very general in nature and intended only to suggest a possible separation scheme. It demonstrates which plans give the best return (separation efficiency) for the money invested. More detailed analysis is necessary before actually implementing a separation plan and very specific community information is needed for such an analysis.

Based on Figure 15, the most cost effective separation scheme is Plan 5 which includes resident separation and separation of waste by workers at the incinerator. This plan results in the highest separation efficiency at almost all levels of investment. There is a wide range of costs on which such a scheme could be implemented. A very inexpensive implementation of the separation scheme would include the following. Several public sites throughout the county could be designated as drop-off centers where bins or dumpsters are placed for residents to drop-off waste containing BFR's. All residents in the community could be mailed information pertaining to what materials contain

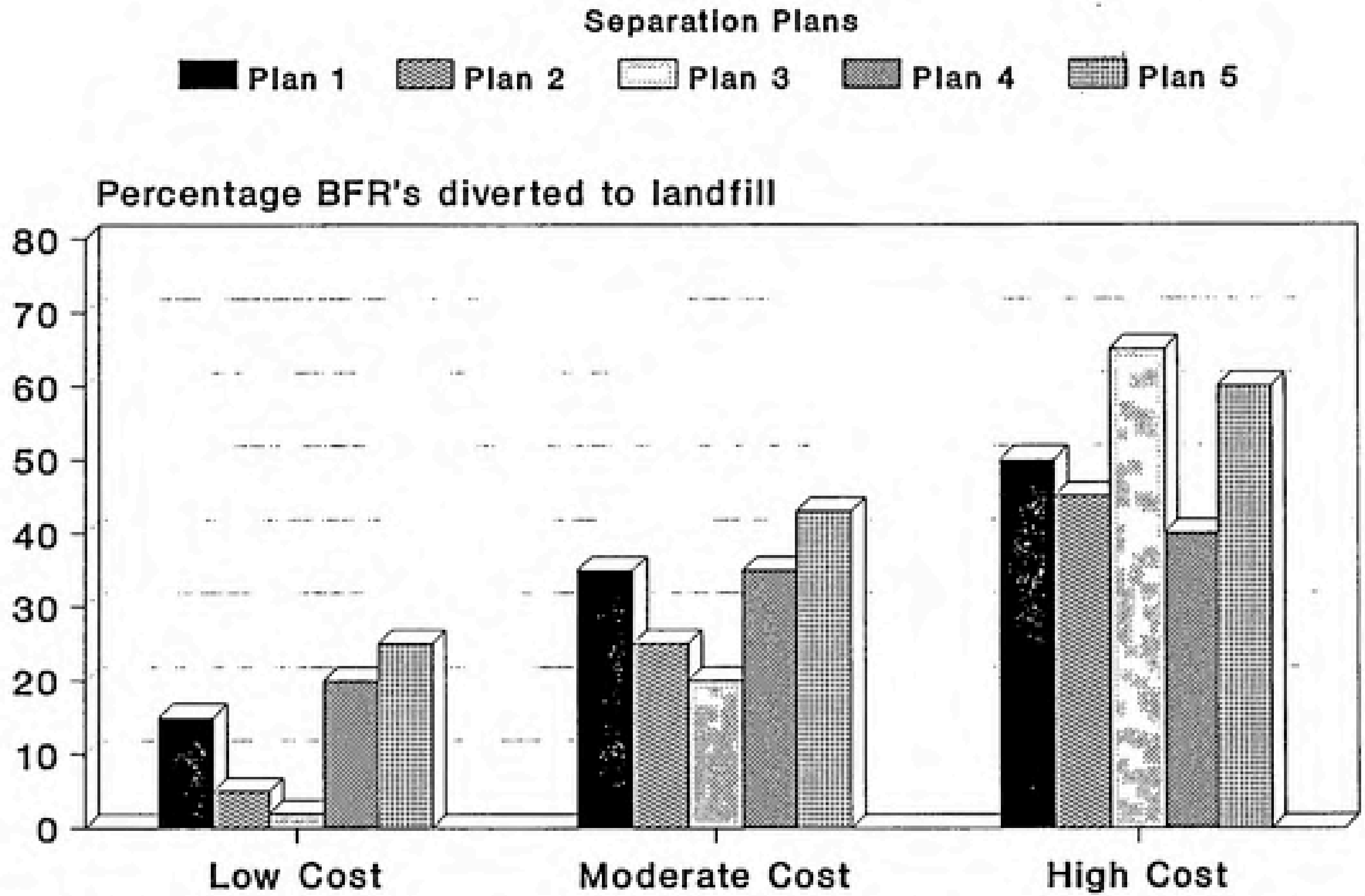


Figure 15. Separation efficiency in diverting waste containing BFR's to the landfill for several separation plans. General values for separation efficiency are predicted at low, moderate and high costs.

BFR's and where they should deliver these materials. At the incinerator truck operators could be required to separate out waste containing the BFR's when they drop it off. A very aggressive and more expensive separation plan could include additional advertising with local media, mandatory separation laws, and hiring labor at the incinerator to separate the waste.

The level of implementation depends on how much the community is willing to spend. This is based on what reduction in health risks are obtained for the cost. Figure 16, is a general graph of health risk reduction for the cost when implementing the Plan 5 separation scheme and is the key to a community decision when determining whether or not and at what level to implement the preferred management option for a troublesome solid waste. The reduction in health risks value is calculated by multiplying the separation efficiency achieved for the cost by the health risk at a specific probability from the health risks curve. For example, according to Figure 15 Plan 5 has a 25% separation efficiency at a low cost and according to Figure 14 there is 95% probability that the health risk is less than 40 lifetime cancer cases. Thus, the health risk reduction when implementing Plan 5 at a low cost is 95% likely to be less than 0.25 times 40 or 10 lifetime cancer cases. Several points from the health risk curve are used to demonstrate the range of likely health risk reduction for

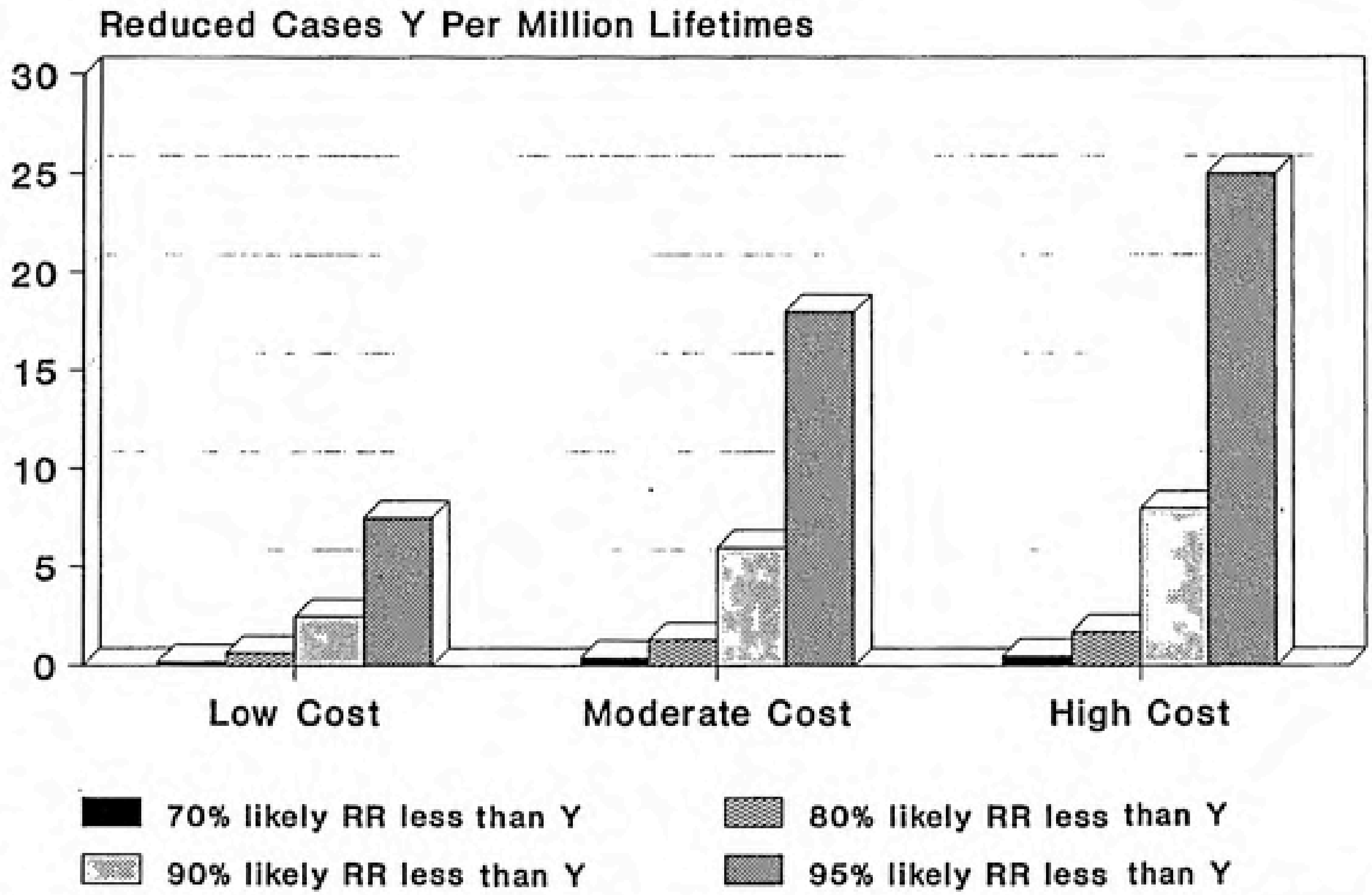


Figure 16. Risk reduction (RR) in cancer cases as a result of implementing separation Plan 5. Separation efficiency values for Plan 5 are multiplied by probable cancer cases predicted at the 70%, 80%, 90% and 95% points from Figure 14 to give the RR.

low, moderate and high cost.

Conclusions

For the case study evaluated, managing solid waste containing BFR's in North Carolina's New Hanover county, the initial results suggest there is a potential health risk from food exposure to BPDD/F's as a result of incinerating of BFR's. However, it is recommended that further study is carried out particularly in regards to the toxicity of BPDD/F's before implementing any expensive separation scheme. The method of assigning toxicity equivalency factors mentioned previously would be a good starting point for a more detailed analysis of toxicity. If the community so desired an low cost separation scheme such as Plan 5 could be implemented.

One of the difficulties in developing a framework for managing troublesome MSW is the complexity involved in evaluating the exposure, health risk and costs. Strategic planning and decision making in the face of uncertainty have always presented a serious challenge to decision makers. The present scale of uncertainty in making many decisions is unprecedented (26). Ineffective methods of dealing with uncertainty can lead to serious mistakes with costly consequences.

This paper attempts to provide a decision framework that explicitly spells out what assumptions are made for uncertain parameters and effectively deal with uncertainty. The framework presented primarily focusses upon landfilling and incineration, but lends itself to consideration of all MSW management options and in this way can be utilized by communities who have a comprehensive solid waste management approach. The framework can be utilized by several parties involved in the decision making process. It is hoped that an established method for incorporating subjective elements into a decision will enable those with differing opinions to communicate in a manner that leads to progress in reaching a decision, and that the public will be able to scrutinize decisions reached as well as participate in the decision making process.

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