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The Disposal of Canada's Nuclear Fuel Waste: The Biosphere Model, BIOTRAC, for Postclosure Assessment

Le stockage permanent des déchets de combustible nucléaire du Canada : Le modèle de biosphère, BIOTRAC, pour l'évaluation de post-fermeture

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AECL RESEARCH

THE DISPOSAL OF CANADA'S NUCLEAR FUEL WASTE: THE BIOSPHERE MODEL, BIOTRAC, FOR POSTCLOSURE ASSESSMENT

by

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Whiteshell Laboratories
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LE STOCKAGE PERMANENT DES DÉCHETS DE COMBUSTIBLE NUCLÉAIRE DU CANADA : LE MODÈLE DE BIOSPHÈRE, BIOTRAC, POUR L'ÉVALUATION DE POST-FERMETURE

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RÉSUMÉ

Le concept de gestion des déchets de combustible nucléaire du Canada demande le stockage permanent de ces déchets dans une enceinte excavée à grande profondeur dans la roche plutonique du Bouclier canadien. EACL documentera la possibilité de réalisation technique du concept et son impact sur l'environnement et la santé de l'être humain dans une Étude d'impact sur l'environnement (EIE). Le présent rapport est l'un des neuf documents principaux d'EIE cités dans la bibliographie. Dans le présent rapport, on décrit le modèle BIOTRAC dont on se sert pour suivre le mouvement des nucléides à partir de la géosphère et à travers la biosphère et pour calculer les concentrations dans l'environnement et les doses radiologiques à l'être humain et autre biote, lesquelles sont dépendantes du temps, pour la phase de post-fermeture. Ces concentrations et doses sont cruciales pour l'évaluation de l'acceptabilité du concept quant à la sûreté et à l'environnement en fonction de la toxicité chimique et radiologique.

On a réalisé BIOTRAC en particulier pour évaluer les effets d'une enceinte de stockage permanent de combustible usé. C'est un modèle complet, générique dont les valeurs paramétriques sont distribuées ou probabilistes pour tenir compte de la variabilité et de l'incertitude spatiales et temporelles. Il se compose de quatre sous-modèles autonomes mais étroitement liés représentants les eaux superficielles, les sols, l'atmosphère et la chaîne alimentaire. En outre, il comporte un modèle unique de prédiction des doses radiologiques au biote non humain représenté par des organismes cibles génériques. On développe en détail l'expression mathématique de chaque sous-modèle et on interprète les résultats du sous-modèle en comparant ceux-ci avec les résultats physiques réels et on évalue et examine complètement toutes les hypothèses. On montre comment on développe les valeurs et distribution paramétriques adoptées pour chaque sous-modèle à partir des résultats obtenus. On présente en détail l'interface entre les sous-modèles et entre BIOTRAC et le modèle de géosphère.

On explique les fluctuations d'état physique de la biosphère au moyen des distributions paramétriques. On examine quantitativement et au moyen d'arguments les modifications majeures de l'environnement, telles que celles dues à la glaciation continentale, et les arguments prouvent que les doses radiologiques à l'être humain n'augmenteront pas soudainement ou intensément au-delà d'une période de 10 000 ans.

On examine la fiabilité de BIOTRAC quant à la validation expérimentale, à l'évaluation du modèle et des résultats, à l'examen critique par les confrères, à l'intercomparaison des modèles, aux hypothèses prudentes, aux procédures d'assurance de la qualité et aux analogues naturels. On se sert d'une analyse de sensibilité du modèle pour identifier les nucléides, les voies et les paramètres qui sont importants dans la détermination des doses radiologiques et pour montrer que le modèle fonctionne par intuition comme il se doit. On identifie les modifications nécessaires pour appliquer le modèle à un site particulier au cours de la sélection de sites éventuels pour une installation de stockage permanent et on montre que ces modifications sont réalisables.

Les renseignements présentés dans le présent rapport indiquent que BIOTRAC décrit de manière convenable et satisfaisante le comportement des nucléides dans la biosphère et ne sous-estimera pas les conséquences lorsqu'on s'en servira pour évaluer le concept de stockage permanent des déchets de combustible nucléaire du Canada.

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ABSTRACT

The nuclear fuel waste management concept of Canada calls for disposal of the waste in a vault mined deep in plutonic rock of the Canadian Shield. The technical feasibility of this concept, and its impact on the environment and human health, will be documented in an Environmental Impact Statement (EIS) by AECL. The present report is one of nine EIS primary references. The report describes the BIOTRAC model, which is used to trace nuclide movement from the geosphere through the biosphere and to calculate time-dependent environmental concentrations and radiological doses to humans and other biota for the postclosure phase. These concentrations and doses are crucial for evaluating the safety and environmental acceptability of the concept in terms of chemical and radiological toxicity.

BIOTRAC was developed specifically to assess the impacts of a used-fuel disposal vault. It is a comprehensive, generic model with distributed or probabilistic parameter values to account for spatial and temporal variability and uncertainty. It is composed of four separate but closely linked submodels representing surface waters, soils, the atmosphere and the food chain. It also includes a unique model for predicting radiological doses to non-human biota, represented by generic target organisms. The mathematical formulation of each submodel is derived in detail and interpreted physically, and all the assumptions are fully evaluated and discussed. It is shown how the parameter values and distributions adopted for each submodel are derived from the available data. The interfaces between the submodels, and between BIOTRAC and the geosphere model, are presented in detail.

Fluctuations in the physical state of the biosphere are accounted for through the parameter distributions. Major environmental changes, such as those caused by continental glaciation, are addressed quantitatively and through reasoned arguments, which indicate that radiological doses to humans will not increase suddenly or dramatically beyond 10 000 a.

The reliability of BIOTRAC is discussed in terms of experimental validation, model and data evaluation, peer review, model intercomparisons, conservative assumptions, quality assurance procedures and natural analogs. A sensitivity analysis of the model is used to identify the nuclides, pathways and parameters that are important in determining radiological doses, and to show that the model is performing as one would expect on an intuitive basis. The changes required to apply the model to a specific site during siting of a disposal facility are identified and shown to be achievable.

The information presented in this report indicates that BIOTRAC provides a suitable and satisfactory description of nuclide behaviour in the biosphere, and will not underestimate consequences when used to assess the concept for the disposal of Canada's nuclear fuel waste.

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PREFACE

In 1992, 15% of the electricity generated in Canada was produced using CANDU nuclear reactors. A by-product of the nuclear power is used CANDU fuel, which consists of ceramic uranium dioxide pellets and metal structural components. Used fuel is highly radioactive. The used fuel from Canada's power reactors is currently stored in water-filled pools or dry storage concrete containers. Humans and other living organisms are protected by isolating the used fuel from the natural environment and by surrounding it with shielding material. Current storage practices have an excellent safety record.

At present, used CANDU fuel is not reprocessed. It could, however, be reprocessed to extract useful material for recycling, and the highly radioactive material that remained could be incorporated into a solid. The term "nuclear fuel waste," as used by AECL, refers to either

- the used fuel, if it is not reprocessed, or
- a solid incorporating the highly radioactive waste from reprocessing.

Current storage practices, while safe, require continuing institutional controls such as security measures, monitoring, and maintenance. Thus storage is an effective interim measure for protection of human health and the natural environment but not a permanent solution. A permanent solution is disposal, a method "in which there is no intention of retrieval and which, ideally, uses techniques and designs that do not rely for their success on long-term institutional control beyond a reasonable period of time" (AECB 1987).

In 1978, the governments of Canada and Ontario established the Nuclear Fuel Waste Management Program "... to assure the safe and permanent disposal" of nuclear fuel waste. AECL was made responsible for research and development on "... disposal in a deep underground repository in intrusive igneous rock" (Joint Statement 1978). Ontario Hydro was made responsible for studies on interim storage and transportation of used fuel and has contributed to the research and development on disposal. Over the years a number of other organizations have also contributed to the Program, including Energy, Mines and Resources Canada; Environment Canada; universities; and companies in the private sector.

The disposal concept is to place the waste in long-lived containers; emplace the containers, enveloped by sealing materials, in a disposal vault excavated at a nominal depth of 500 to 1000 m in intrusive igneous (plutonic) rock of the Canadian Shield; and (eventually) seal all excavated openings and exploration boreholes to form a passively safe system. Thus there would be multiple barriers to protect humans and the natural environment from contaminants in the waste: the container, the very low-solubility waste form, the vault seals, and the geosphere. The disposal technology includes options for the design of the engineered components, including the disposal container, disposal vault, and vault seals, so that it is adaptable to a wide range of regulatory standards, physical conditions, and social requirements. Potentially suitable bodies of plutonic rock occur in a large number of locations across the Canadian Shield.

In developing and assessing this disposal concept, AECL has consulted broadly with members of Canadian society to help ensure that the concept and the way in which it would be implemented are technically sound and represent a generally acceptable disposal strategy. Many groups in Canada have had opportunities to comment on the disposal concept and on the waste management program. These include government departments and agencies, scientists, engineers, sociologists, ethicists, and other members of the public. The Technical Advisory Committee to AECL on the Nuclear Fuel Waste Management Program, whose members are nominated by Canadian scientific and engineering societies, has been a major source of technical advice.

In 1981, the governments of Canada and Ontario announced that "... no disposal site selection will be undertaken until after the concept has been accepted. This decision also means that the responsibility for disposal site selection and subsequent operation need not be allocated until after concept acceptance" (Joint Statement 1981).

The acceptability of the disposal concept is now being reviewed by a federal Environmental Assessment Panel, which is also responsible for examining a broad range of issues related to nuclear fuel waste management (Minister of the Environment, Canada 1989). After consulting the public, the Panel issued guidelines to identify the information that should be provided by AECL, the proponent of the disposal concept (Federal Environmental Assessment Review Panel 1992).

AECL is preparing an Environmental Impact Statement to provide information requested by the Panel and to present AECL's case for the acceptability of the disposal concept. A Summary will be issued separately. This report is one of nine primary references that summarize major aspects of the disposal concept and supplement the information in the Environmental Impact Statement. A guide to the contents of the EIS, the Summary, and the primary references follows this Preface.

In accordance with the 1981 Joint Statement of the governments of Canada and Ontario, no site for disposal of nuclear fuel waste is proposed at this time. Thus in developing and assessing the disposal concept, AECL could not design a facility for a proposed site and assess the environmental effects to determine the suitability of the design and the site, as would normally be done for an Environmental Impact Statement. Instead, AECL and Ontario Hydro have specified illustrative "reference" disposal systems and assessed those.

A "reference" disposal system illustrates what a disposal system, including the geosphere and biosphere, might be like. Although it is hypothetical, it is based on information derived from extensive laboratory and field research. Many of the assumptions made are conservative, that is, they would tend to overestimate adverse effects. The technology specified is either available or judged to be readily achievable. A reference disposal system includes one possible choice among the options for such things as the waste form, the disposal container, the vault layout, the vault seals, and the system for transporting nuclear fuel waste to a disposal facility.

The components and designs chosen are not presented as ones that are being recommended but rather as ones that illustrate a technically feasible way of implementing the disposal concept.

After the Panel has received the requested information, it will hold public hearings. It will also consider the findings of the Scientific Review Group, which it established to provide a scientific evaluation of the disposal concept. According to the Panel's terms of reference "As a result of this review the Panel will make recommendations to assist the governments of Canada and Ontario in reaching decisions on the acceptability of the disposal concept and on the steps that must be taken to ensure the safe long-term management of nuclear fuel wastes in Canada" (Minister of the Environment, Canada 1989).

Acceptance of the disposal concept at this time would not imply approval of any particular site or facility. If the disposal concept is accepted and implemented, a disposal site would be sought, a disposal facility would be designed specifically for the site that was proposed, and the potential environmental effects of the facility at the proposed site would be assessed. Approvals would be sought in incremental stages, so concept implementation would entail a series of decisions to proceed. Decision-making would be shared by a variety of participants, including the public. In all such decisions, however, safety would be the paramount consideration.

The EIS, Summary, and Primary References

Environmental Impact Statement on the Concept for Disposal of Canada's Nuclear Fuel Waste (AECL 1994a)

Summary of the Environmental Impact Statement on the Concept for Disposal of Canada's Nuclear Fuel Waste (AECL 1994b)

The Disposal of Canada's Nuclear Fuel Waste: Public Involvement and Social Aspects (Greber et al. 1994)

The Disposal of Canada's Nuclear Fuel Waste: Site Screening and Site Evaluation Technology (Davison et al. 1994a)

The Disposal of Canada's Nuclear Fuel Waste: Engineered Barriers Alternatives (Johnson L.H. et al. 1994a)

The Disposal of Canada's Nuclear Fuel Waste: Engineering for a Disposal Facility (Simmons and Baumgartner 1994)

The Disposal of Canada's Nuclear Fuel Waste: Preclosure Assessment of a Conceptual System (Grondin et al. 1994)

The Disposal of Canada's Nuclear Fuel Waste: Postclosure Assessment of a Reference System (Goodwin et al. 1994)

The Disposal of Canada's Nuclear Fuel Waste: The Vault Model for Postclosure Assessment (Johnson L.H. et al. 1994b)

The Disposal of Canada's Nuclear Fuel Waste: The Geosphere Model for Postclosure Assessment (Davison et al. 1994b)

The Disposal of Canada's Nuclear Fuel Waste: The Biosphere Model, BIOTRAC, for Postclosure Assessment (Davis et al. 1993)

GUIDE TO THE CONTENTS OF THE ENVIRONMENTAL IMPACT STATEMENT, THE SUMMARY, AND THE PRIMARY REFERENCES

ENVIRONMENTAL IMPACT STATEMENT AND SUMMARY

Environmental Impact Statement on the Concept for Disposal of Canada's Nuclear Fuel Waste (AECL 1994a)

- provides an overview of AECL's case for the acceptability of the disposal concept

- provides information about the following topics:
  - the characteristics of nuclear fuel waste
  - storage and the rationale for disposal
  - major issues in nuclear fuel waste management
  - the disposal concept and implementation activities
  - alternatives to the disposal concept
  - methods and results of the environmental assessments
  - principles and potential measures for managing environmental effects
  - AECL's overall evaluation of the disposal concept

Summary of the Environmental Impact Statement on the Concept for Disposal of Canada's Nuclear Fuel Waste (AECL 1994b)

- summarizes the contents of the Environmental Impact Statement

PRIMARY REFERENCES

The Disposal of Canada's Nuclear Fuel Waste: Public Involvement and Social Aspects (Greber et al. 1994)

- describes the activities undertaken to provide information to the public about the Nuclear Fuel Waste Management Program and to obtain public input into the development of the disposal concept

- presents the issues raised by the public and how the issues have been addressed during the development of the disposal concept or how they could be addressed during the implementation of the disposal concept

- discusses social aspects of public perspectives on risk, ethical issues associated with nuclear fuel waste management, and principles for the development of a publicly acceptable site selection process

The Disposal of Canada's Nuclear Fuel Waste: Site Screening and Site Evaluation Technology (Davison et al. 1994a)

- discusses geoscience, environmental, and engineering factors that would need to be considered during siting

- describes the methodology for characterization, that is, for obtaining the data about regions, areas, and sites that would be needed for facility design, monitoring, and environmental assessment

<u>The Disposal of Canada's Nuclear Fuel Waste: Engineered Barriers Alternatives (Johnson L.H. et al. 1994a)</u>

- describes the characteristics of nuclear fuel waste
- describes the materials that were evaluated for use in engineered barriers, such as containers and vault seals
- describes potential designs for containers and vault seals
- describes procedures and processes that could be used in the production of containers and the emplacement of vault-sealing materials

<u>The Disposal of Canada's Nuclear Fuel Waste: Engineering for a Disposal Facility (Simmons and Baumgartner 1994)</u>

- discusses alternative vault designs and general considerations for engineering a nuclear fuel waste disposal facility
- describes a disposal facility design that was used to assess the technical feasibility, costs, and potential effects of disposal (Different disposal facility designs are possible and might be favoured during concept implementation.)
- presents cost and labour estimates for implementing the design

<u>The Disposal of Canada's Nuclear Fuel Waste: Preclosure Assessment of a Conceptual System (Grondin et al. 1994)</u>

- describes a methodology for estimating effects on human health, the natural environment, and the socio-economic environment that could be associated with siting, constructing, operating (includes transporting used fuel), decommissioning, and closing a disposal facility
- describes an application of this assessment methodology to a reference disposal system (We use the term "reference" to designate the disposal systems, including the facility designs, specified for the assessment studies. Different disposal facility designs are possible and might be favoured during concept implementation.)
- discusses technical and social factors that would need to be considered during siting
- discusses possible measures and approaches for managing environmental effects

The Disposal of Canada's Nuclear Fuel Waste: Postclosure Assessment of a Reference System (Goodwin et al. 1994)

- describes a methodology for
  - estimating the long-term effects of a disposal facility on human health and the natural environment,
  - determining how sensitive the estimated effects are to variations in site characteristics, design parameters, and other factors, and
  - evaluating design constraints

- describes an application of this assessment methodology to a reference disposal system (We use the term "reference" to designate the disposal systems, including the facility designs, specified for the assessment studies. Different disposal facility designs are possible and might be favoured during concept implementation.)

The Disposal of Canada's Nuclear Fuel Waste: The Vault Model for Postclosure Assessment (Johnson L.H. et al. 1994b)

- describes the assumptions, data, and model used in the postclosure assessment to analyze processes within and near the buried containers of waste

- discusses the reliability of the data and model

The Disposal of Canada's Nuclear Fuel Waste: The Geosphere Model for Postclosure Assessment (Davison et al. 1994b)

- describes the assumptions, data, and models used in the postclosure assessment to analyze processes within the rock in which a disposal vault is excavated

- discusses the reliability of the data and model

The Disposal of Canada's Nuclear Fuel Waste: The Biosphere Model, BIOTRAC, for Postclosure Assessment (this volume)

- describes the assumptions, data, and model used in the postclosure assessment to analyze processes in the near-surface and surface environment

- discusses the reliability of the data and model

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EXECUTIVE SUMMARY

ES.1 INTRODUCTION

The Canadian Nuclear Fuel Waste Management Program (NFWMP) was established jointly by the governments of Canada and Ontario in 1978 to develop a method to ensure the safe disposal of fuel waste from Canada's nuclear reactors (Joint Statement 1978). In the disposal concept under consideration, the waste would be placed in corrosion-resistant metal containers, surrounded by clay-based buffer material, in a vault excavated at a depth of 500 to 1000 m in plutonic rock of the Canadian Shield. The engineered and natural barriers of this disposal system would isolate the waste from the biosphere, with its humans and other biota, for many years. Nevertheless, it is possible that the containers could eventually corrode, and that groundwater would transport waste material to the earth's surface (Figure ES-1). Because this would occur far in the future, the environmental and health impacts from radiological and chemical toxicity must be predicted and assessed using mathematical models.

This report is one of nine primary references that support the Environmental Impact Statement (EIS) (AECL 1994a), which is subject to scientific and public review to help a federal Environmental Assessment Panel (Minister of the Environment, Canada 1989) determine the acceptability of the disposal concept. The Panel has issued detailed guidelines to AECL for preparing the EIS (Federal Environmental Assessment Review Panel 1992). The process of choosing a location for the vault will not be started until the concept has been fully reviewed and approved (Joint Statement 1981).

In this primary reference, we describe the BIOTRAC (BIOsphere TRansport And Consequence) model used to simulate the transport of nuclides through the biosphere in the postclosure phase of the disposal concept when the vault is full, and has been decommissioned and closed. Under the direction of the executive code SYVAC3 (SYstems Variability Analysis Code - Generation 3), BIOTRAC is coupled with models of transport in the vault and geosphere to provide probabilistic estimates of nuclide concentrations in the environment, and of radiological doses to humans and other organisms.

BIOTRAC was developed specifically to evaluate the postclosure environmental and health impacts of the concept for the disposal of Canada's nuclear fuel waste. The model is applicable for up to 10 000 a into the future, the period of quantitative assessment of the concept specified by the Atomic Energy Control Board (AECB 1987). This period will likely be free from continental glaciation.

BIOTRAC and its numerous parameter values are based on a vast amount of literature data. Since 1978, the model has also been supported by an extensive research program involving a variety of field and laboratory studies to fill in gaps in knowledge on the transport of nuclides in the biosphere and their effects on biota (Iverson et al. 1982, Zach 1985a, Zach et al. 1987). Most of this research was focussed on the Canadian Shield, and has involved interactions with researchers from nuclear waste disposal programs in several other countries.

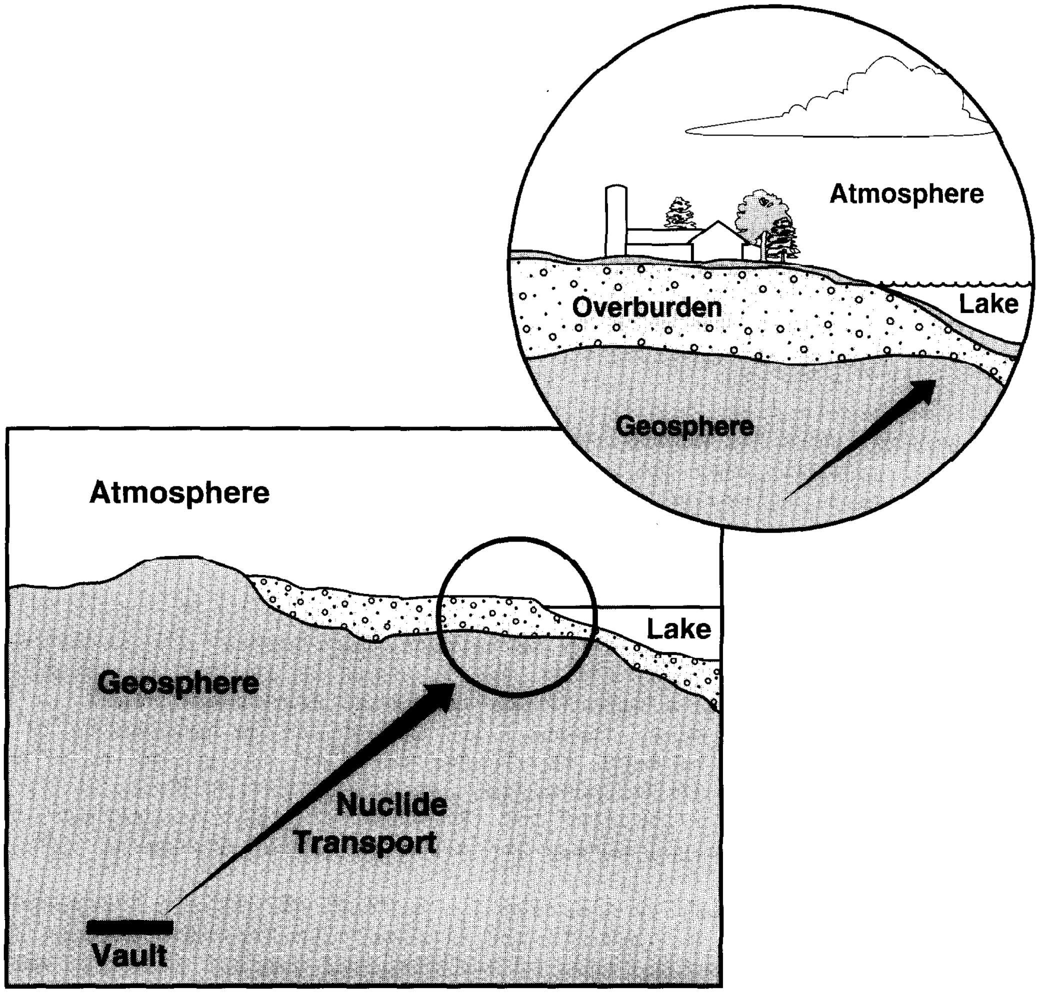

FIGURE ES-1: Schematic Representation of Groundwater Transport of Nuclides from the Vault, 500 to 1000 m Underground, to the Biosphere (Enlarged Insert)

The important processes and pathways associated with transport from an underground source over very long times were identified through a rigorous scenario analysis procedure that involved literature reviews, brainstorming sessions and the use of expert opinion. These processes and pathways are modelled probabilistically using a systems variability analysis approach (Goodwin et al., in preparation). The values of many model parameters are represented by probability density functions (PDFs), which allow for uncertainty in model structure, variability and uncertainty in parameter values, and natural variability in spatial and temporal aspects of the biosphere. Moreover, in the absence of a specific site for the vault, the distributed parameter values allow the assessment to be conducted generically as far as the biosphere is concerned. The PDFs chosen for the biosphere parameters encompass the full range of values that might be encountered on the Ontario portion of the Canadian Shield. These values are also representative of much of the Shield in general. At the beginning of each computer simulation or run, SYVAC3 selects a possible state of the system by randomly sampling a value for each parameter from its specified PDF.

This set of values is used to calculate nuclide concentrations and doses for that state. The procedure is repeated, typically several thousand times, to provide the full range of possible consequences and their frequency of occurrence, from which the uncertainty in the results can be estimated. To ensure that computer requirements do not become impractical, the various transport processes are modelled as simply and efficiently as possible, consistent with the accuracy and the detail needed in the results. In areas where knowledge is limited, and where realistic models cannot be formulated or validated, we make conservative assumptions to ensure that environmental concentrations and doses are not underestimated.

Human radiation doses are calculated by BIOTRAC for individuals belonging to a group of people receiving the greatest exposure because of its location and lifestyle. We assume that this all-inclusive critical group is composed of a sequence of self-sufficient rural households living near where nuclides would discharge to the biosphere, and where nuclide dilution and dispersion are at a minimum. The lifestyle of the critical group is based on present human behaviour using conservative, yet reasonable, assumptions. For example, members are assumed to live their entire lives at the discharge zone, having access only to those parts of the biosphere that are potentially contaminated. They are assumed to be entirely self-sufficient, drawing all their resources, including food, water, air, heating fuels and building materials, from the local environment. For dose prediction purposes, the group is assumed to be represented by reference man, as defined by the International Commission on Radiological Protection (ICRP 1975). Predicted doses can be compared (Goodwin et al. 1994) with regulatory requirements established by the AECB (1987).

Radiation doses for non-human biota are predicted by BIOTRAC for a set of generic target organisms at the discharge zone, including a plant, a mammal, a bird and a fish. Doses to these organisms can be compared with the detrimental effects known to occur at various dose levels (Goodwin et al. 1994). Humans also serve as an indicator species for evaluating radiological effects on other biota. Furthermore, we show that radiological and chemical protection of the biota and their habitat in general can be evaluated by comparing the concentrations of nuclides in surface water, soil and air, predicted by BIOTRAC with various regulatory criteria and guidelines, and with existing environmental baseline concentrations and their variability (Goodwin et al. 1994). With these assessment methodologies, we aim to achieve the protection of plant and animal populations and also, indirectly, of higher ecological levels - communities and ecosystems.

ES.2 THE GEOSPHERE/BIOSPHERE INTERFACE

BIOTRAC is driven by the output of the geosphere model (Figure ES-2), which for the postclosure assessment is a site-specific model based on data obtained at the Whiteshell Research Area (WRA) (Davison et al. 1994b, Goodwin et al. 1994). For modelling purposes, a hypothetical vault is

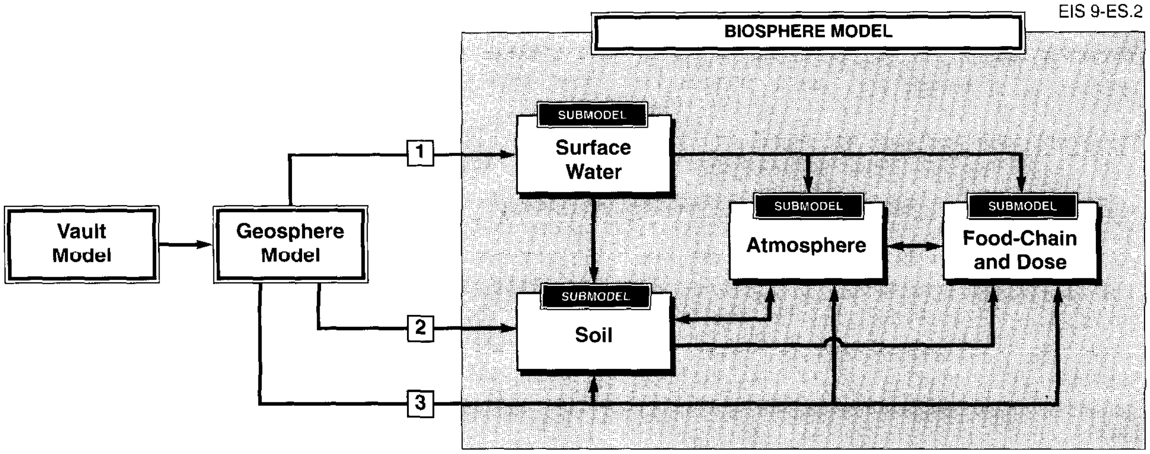

FIGURE ES-2: Schematic Representation of the Three Main Assessment Models (Vault, Geosphere and Biosphere) for the Disposal Concept Assessment, and of the Main Nuclide Transfers Among the Four Submodels of the Biosphere Model (Surface Water, Soil, Atmosphere, and Food-Chain and Dose) and Between the Geosphere and Biosphere Models. Discharges from the geosphere to the biosphere model are: (1) aquatic, (2) terrestrial and (3) well.

located at a depth of 500 m in the region of the WRA. The topography of the area and a conceptual model of the subsurface structure suggest that groundwater carrying nuclides from the vault would reach the surface at three distinct discharge zones in or near a water body known as Boggy Lake. Although the discharge would occur primarily to the lake itself, we assume that a small portion of each zone underlies a terrestrial area that is suitable for terrestrial biota and for farming by the critical group. Permanent and temporary, or seasonal, wetlands are considered through discharge to the lake and terrestrial areas respectively. In some model simulations, a final point of nuclide discharge to the biosphere is a domestic bedrock well drilled into the contaminated groundwater plume.

Because the geosphere model is site-specific, its parameters have values representative of the WRA. A few of these parameters also appear in the biosphere model. To ensure consistency between the two-models, the values of the common parameters were not set independently in BIOTRAC, but were set equal to the values assigned in the geosphere model.

The interfaces between the geosphere and the biosphere occur at the top of the compacted layer of sediment beneath the lake, at the bottom of the unsaturated soil zone, or water table, and at the well. In the compacted sediments, the nuclide load from the vault is assumed to arise entirely through sorption from upward moving groundwater. Concentrations in compacted sediments are calculated on the assumption that the flow through them is advection-dominated, and that nuclides in the flow are partitioned between the solid and liquid phases.

The biosphere model is driven by the total nuclide flow out of the geosphere, including flows to aquatic and terrestrial areas and to the well. For aquatic areas, nuclides released from the geosphere are discharged directly into the lake from compacted sediment. For terrestrial areas, the biosphere model is driven by the nuclide concentration in the pore-water of the lowest soil layer, which is calculated from a mass balance equation that takes into account advection into and out of the layer, and ingrowth through radioactive decay and decay of the radionuclide itself. Finally, well-water concentrations are calculated using a two-dimensional analytical model that is part of the geosphere model.

ES.3 THE BIOSPHERE SUBMODELS

Nuclide transport in the biosphere is modelled with four separate but closely linked submodels representing surface waters, unsaturated soils, the atmosphere and food chains (Figure ES-2).

The surface water body is assumed to be a typical Canadian Shield lake, and is modelled as a two-compartment system, one compartment representing the water column and the other compartment representing recently deposited mixed sediments that overlie the compacted sediments that are part of the geosphere model. Nuclides from the geosphere are released directly into the water column, from where they may be transferred to the mixed sediments. This system is described by coupled mass balance equations that take into account hydrological flushing, dilution, mixing, sedimentation, gaseous evasion, and radioactive decay and ingrowth. Nuclide inputs to the lake, caused by runoff and atmospheric deposition, and the resuspension of nuclides from the sediments to the water column, are treated implicitly. The model output includes time-dependent nuclide concentrations in the water column and in the mixed sediment.

The prediction of soil concentration is based on a mechanistic soil model, SCEMR1 (Soil Chemical Exchange and Migration of Radionuclides Model, Revision 1). This model can provide the detailed treatment of processes and the fine time and space resolutions necessary to simulate nuclide migration through the soil profile. SCEMR1 is a one-dimensional, time-dependent model that uses detailed meteorological data, together with the Darcy equation and the equation of continuity, to calculate water flows between four soil layers on a daily basis. Nuclides introduced into this system from groundwater below or from aerial irrigation water above may be advected downward by leaching or upward by capillary rise. Concentrations in a given soil layer are calculated from a simple mass balance equation involving the flows into and out of the layer, assuming that the nuclides are mixed instantaneously and uniformly within each layer. Nuclides are partitioned between solid and liquid phases using the soil solid/liquid partition coefficient. SCEMR1 is driven by the nuclide concentration in the pore-water of the soil layer that receives the contaminant input; these concentrations are also calculated using a mass balance approach. The output of SCEMR1 is the time-dependent nuclide concentration in the soil root zone for each of three contamination pathways - groundwater discharge, aerial irrigation and atmospheric deposition.

Because SCEMR1 is a detailed research model, it requires too much computer time to be of practical use in a long-term, probabilistic assessment. Accordingly, a more efficient model was derived for inclusion in BIOTRAC. This model is based on a statistical summary, in regression equation form, of the steady-state root-zone concentrations, designated by Css, and the times to steady state, designated by tss, predicted by SCEMR1 for a constant source term and a wide range of values of the important model parameters. The root-zone concentrations were successfully approximated as a function of time by a simple analytical expression involving Css and tss. This expression was used to write a mass balance equation for the root zone to allow for a time-dependent nuclide source term, ingrowth of daughter nuclides, and nuclide losses resulting from gaseous evasion, cropping and radioactive decay. In this way, root soil concentrations can be calculated for any contaminant source in a few seconds of computer time.

In each BIOTRAC simulation, we calculate nuclide concentrations in the soils of three distinct fields: a garden, which supplies all the plant food eaten by the critical group; a forage field, which provides the feed required by their livestock; and a woodlot, which supplies the wood needed to build and heat their home. Non-human organisms also live on these fields and depend on the fields for food and shelter. We model a fourth field with the characteristics of a peat bog for simulations involving an organic soil and when the critical group heats its home with peat.

The transport equations defining the surface water and soil submodels are solved by a response function/convolution approach that is used throughout SYVAC3 to treat time-dependent systems.

Nuclides reach the atmosphere as a result of suspension from contaminated water bodies, soils and vegetation. The atmosphere submodel treats a variety of suspension mechanisms, both natural and anthropogenic, including the suspension of particulate nuclides from terrestrial and aquatic sources, the evasion of gases from terrestrial and aquatic sources, and the release of nuclides when biomass is burned. Once in the air, the nuclides undergo dispersion and deposition back to the underlying surface. Additional processes can raise indoor air concentrations above outdoor levels. We model the diffusion of volatile nuclides from the soil into buildings and the release of nuclides from water used inside the home of the critical group.

The models we use to simulate the suspension mechanisms vary considerably in complexity, depending on our theoretical understanding of the process and on the amount and quality of the available data. In some cases, simple mass loading parameters are used to calculate air concentrations directly from the nuclide concentration in the source compartment, such as soil. This approach allows a number of suspension mechanisms to be modelled collectively, and also accounts for the effects of atmospheric dispersion. For other mechanisms, nuclide fluxes to the atmosphere can be predicted and combined with a dispersion model to calculate air concentrations. For a ground-level area source, such as a contaminated field or water body, we based our dispersion model on the trajectory simulation approach. In all cases, the models are equilibrium models, in that air concentrations are assumed to adjust instantaneously to changes in the concentration of the source compartment. Total air concentrations are calculated by summing the contributions from the individual suspension mechanisms. Indoor and outdoor concentrations are calculated separately for each nuclide.

The rate at which nuclides are deposited from the air to soil and vegetation is also predicted by the atmosphere submodel. Deposition velocities are used to model the dry deposition process and washout ratios are used to model wet deposition.

The food-chain submodel, CALDOS (CALculation of DOSe food-chain and dose model), traces nuclide movement from the physical compartments of the biosphere, i.e., surface water, soil and air, through the food chain to humans and other organisms, and calculates radiological doses from both internal and external exposure pathways. Transfer is predicted using simple multiplicative chain equations that assume the nuclide uptake by plants and animals, and doses, are directly proportional to nuclide concentrations in the source compartment. The model is therefore a steady-state, equilibrium model.

The internal exposure pathways considered in CALDOS are the ingestion of contaminated plants, terrestrial animals, water and soil by humans; the ingestion of terrestrial animals and fish that have consumed contaminated plants, water or soil; and the inhalation of air by humans. In treating these pathways, CALDOS accounts for processes such as root uptake, contamination of plant surfaces by irrigation and atmospheric deposition, losses from plant surfaces as a result of environmental processes, transfer to animals and humans, and radioactive decay and ingrowth. The external pathways treated are immersion in contaminated air and water, and exposure to contaminated soil and building materials. The total dose to a member of the critical group and other organisms is found by summing the individual doses from all nuclides and exposure pathways.

Internal doses depend on the amount of contaminated food, water and air taken into the body. For humans, CALDOS calculates these amounts in an integrated way from the total energy need, the diet, and the nutritional content of the diet. For modelling purposes, the diet is assumed to consist of five general food types: terrestrial plant foods, mammalian meats, milk and dairy products, poultry and eggs, and freshwater fish. Some of these food types are also used as representative organisms for evaluating doses to non-human organisms.

A few nuclides exhibit special properties that require alternative approaches to transport modelling and dose calculation. A specific-activity model is used to predict internal doses to humans from tritium (^{3}H), which is very mobile in the environment. A limited specific-activity model is also used for ^{129}I because internal iodine doses are dominated by the thyroid gland and the iodine content of the thyroid is regulated metabolically. The specific activity of ^{129}I in the thyroid and of ^{14}C in the body are not allowed to exceed the specific activity of these nuclides in the groundwater discharging from the geosphere to the biosphere. The transport and exposure pathways can be greatly simplified for the noble gases, which do not accumulate and disperse rapidly in the biosphere, but special attention has been paid to radon (^{222}Rn) inhalation. Short-lived daughter radionuclides with half-lives less than one day are assumed to be in secular equilibrium with their precursors throughout the biosphere, and are not modelled explicitly. The contribution of these nuclides to dose is accounted for through their precursors.

In addition to the four submodels, BIOTRAC also includes a model for predicting radiological doses to non-human biota. This model is similar to CALDOS, and the two models share many parameter values and PDFs. The model considers four generic target organisms for dose prediction - a fish, a plant, a mammal and a bird. These organisms broadly represent Canadian Shield biota in terms of exposure situations and parameter values.

The model focuses on the three nuclides that are potentially most important, i.e., ^{14}C, ^{129}I and ^{99}Tc, and it considers both internal and external exposure, largely in terms of whole-body exposure. Internal exposure is based on food-chain transfer, which includes food, water and soil ingestion. External exposure includes water immersion, air immersion, soil or sediment immersion, and immersion in vegetation. Depending on the exact target organism, several of these exposures are modelled simultaneously to include the diverse habits of Canadian Shield organisms, particularly animals. We also show how radiological doses can be calculated for specific species rather than for generic target organisms.

Values and PDFs for the various BIOTRAC parameters were selected only after a careful appraisal of the available data. Most of the values were drawn from the literature, but some were supplied by our own research programs. Where possible, the values used were annual averages based on data from the Canadian Shield. Where the data were numerous, a quantitative statistical analysis was used to assign a distribution type and attributes to a given parameter; otherwise the PDF was set subjectively on the basis of all the available information. Truncations of the PDF and correlations between parameters were used to avoid unreasonable values or combinations of values. For each parameter, we show how appropriate values and PDFs were derived from the data.

ES.4 THE INTEGRATED MODEL

The four BIOTRAC submodels, the geosphere/biosphere interface model and the model for non-human biota that make up BIOTRAC, although distinct, were designed to interface smoothly with each other to provide a cohesive description of nuclide transport through the biosphere as a whole. The output of one model serves as input to the next. The order in which the calculations are done is chosen to ensure that the information required at each point in BIOTRAC is available from previous calculations. A step-by-step walkthrough of a typical BIOTRAC simulation, focusing on human dose prediction, is presented to illustrate how the model works and to put the various exposure pathways into perspective.

The resources required by the critical group are calculated internally by BIOTRAC in a consistent manner, given the number of people in the household under consideration. The number of animals needed by the household is computed from the food yield of each animal and the quantity consumed by household members. Similarly, the size of the garden is calculated as the area needed to grow the terrestrial plant foods required by the group. The size of the forage field is calculated by considering the area required to grow the feed needed by the livestock. The area required to provide sufficient wood or peat to heat the household is calculated on the basis of energy needs and the energy content of the fuel. The amount of water used by the household is found by adding the water required for domestic purposes (drinking, cooking, bathing, laundry, etc.), the drinking water needed by the domestic animals, and the water used to irrigate the garden or forage field, if irrigation is practised.

Nuclide mass is conserved within each of the four submodels of BIOTRAC, but not necessarily when nuclides are transferred between compartments. The inventories of donor compartments are often not depleted when nuclides migrate to a new compartment. For example, soil inventories are usually not reduced when nuclides are suspended into the atmosphere. Although this type of assumption results in a generation of nuclide mass within the model, it allows complex processes such as runoff, recycling and atmospheric suspension and deposition to be handled very simply. Furthermore, it results in conservative predictions of nuclide masses for both donor and receptor compartments. In all cases where source inventories are not depleted, the nuclide flux out of the source compartment is small compared with other loss terms. The amount of nuclide mass created is small, and does not greatly increase predicted environmental concentrations and doses.

For the concept of geological disposal of nuclear fuel waste to be practical and acceptable, it is necessary to demonstrate that a suitable site can be found and assessed (Davison et al. 1994a). Although BIOTRAC is a generic model, it can readily be modified for application to specific sites. We discuss additional processes and the different modelling approaches required for a site-specific model. We identify the parameters that would have to take on site-specific values, and show how such values can be obtained. In all cases, the required changes are relatively minor and easily achievable.

A separate model, PREAC, has been designed to assess the preclosure phase of the disposal concept, which comprises the construction, operation, decommissioning and closure of the vault (Grondin et al. 1994). Although modelling requirements in the pre- and postclosure phases are quite different, they both deal with nuclide transport and exposure in the biosphere immediately surrounding the disposal facility. We compare PREAC and BIOTRAC, and show that the differences between them do not constitute inconsistencies, given the different phases that the models address. The models are well suited to their respective purposes and provide complementary approaches to different aspects of the assessment.

We performed a sensitivity analysis of BIOTRAC to quantify its response to changes in input parameters, and to identify the nuclides and pathways that are important in determining doses to the critical group. The submodels were analyzed first, using unit inputs. These results were then used to guide the analysis of BIOTRAC as a whole, which was done with a simplified but realistic input from the geosphere. The results show that ^{129}I causes, by far, most of the dose to the critical group, with ^{14}C contributing almost all the remainder. Most of the ^{129}I dose occurs through ingestion of terrestrial plant foods contaminated by root uptake from soil irrigated with well water. The next most important exposure pathway is the ingestion of plants contaminated through atmospheric deposition. The parameters to which the total dose to man is most sensitive are, in order of importance, the source of domestic water (well or lake), the parameter describing ^{129}I evasion from the lake to the atmosphere, and the evasion rate of ^{129}I from soil to the atmosphere. All of the sensitivity analysis results agree with our intuitive expectation of biosphere behaviour and increase our confidence in the predictions of BIOTRAC.

ES.5 ENVIRONMENTAL CHANGES

BIOTRAC was developed to provide predictions over a period of about 10 000 a, during which time current interglacial climatic conditions are assumed to persist. Because the parameter values sampled at the beginning of each simulation are held constant throughout that simulation, the state of the biosphere is assumed to remain unchanged with time, instead of exhibiting its characteristic fluctuations. However, the effects of such fluctuations are incorporated implicitly through the use of distributed parameter values, assuming that nuclide concentrations depend primarily on environmental conditions at the time of interest, and not on conditions prior to that time. Our parameter distributions likely account for all the temporal changes that could occur at a specific site during interstadial conditions of the glacial cycle since they reflect today's very large spatial variability across the Canadian Shield.

Many geological processes will affect the Canadian Shield on time scales longer than 10 000 a. However, the majority of these processes need not be considered in detail because their potential to influence nuclide migration through the biosphere is small. We believe that only continental glaciation, including glacially induced faulting and succession in a glacial regime, could affect consequence predictions significantly. We assessed glaciation by using a modified version of BIOTRAC to calculate radiological doses to humans for conditions representative of a cold interstadial climate, and by qualitatively evaluating a number of glacially induced pathways. The results demonstrated that glaciation will not cause doses to rise appreciably above those predicted for current interglacial conditions. Furthermore, we conclude that the only aspect of succession that must be considered is the gradual filling in of surface water bodies. Even here, the important effects on dose occur in the final phase of succession when the bottom sediments become exposed and are used for agriculture. The use of sediment as soil has been explicitly included in BIOTRAC.

Long-term changes in human culture and technology are impossible to predict, and we have made no attempt to account for their effects in the model. The use of the critical-group concept overcomes many of the difficulties in defining appropriate exposure pathways to humans far in the future. However, we have explicitly considered human intrusion. We calculate the consequences of a bedrock well drilled into the groundwater plume from the vault. We also discuss the impacts of other intrusion scenarios, including exploratory drilling and mining.

ES.6 BIOTRAC VALIDATION

The surface water and soil submodels of BIOTRAC, and parts of the atmosphere submodel, have been validated experimentally. The predictions of the soil and surface water models agree well with observations from experiments that lasted over several years and dealt with a variety of nuclides under a range of Canadian Shield conditions. These models therefore simulate nuclide behaviour adequately, and produce realistic predictions on time scales of a few years. Similarly, the model that forms the basis of the dispersion relations in the atmosphere submodel agrees well with experimental data.

The remainder of the atmosphere submodel, the food-chain and dose submodel, and the model for non-human biota have not been validated experimentally; neither has BIOTRAC as a whole. Independent data sets for comparison with predictions of nuclide migration from an underground source do not exist at this time. It is clearly impossible to validate predictions of nuclide concentrations thousands of years in the future.

In view of this challenge, we used a variety of other methods to establish BIOTRAC's credibility. We drew heavily on the literature, on our own research programs and on international experience to define the processes, parameter values and modelling approaches best suited for assessing the long-term performance of an underground disposal vault. We have continually exposed the models, the data and our research programs to peer review through publication in the open literature, formal independent reviews, and review by the Technical Advisory Committee (TAC) to the NFWMP. In this way, we have demonstrated that BIOTRAC reflects the large accumulated body of knowledge on nuclide transport through the biosphere, and that it follows scientific practice for models designed to assess geological disposal systems.

We have used a number of other approaches to assure the validity of our models. Natural analogs of nuclide transfer in the biosphere have been used to determine parameter values and to develop approaches to model nuclide transfer over long periods of time. An informal quality assurance program based on well-established scientific principles has guided the development of BIOTRAC from the outset, and a formal quality assurance program was set up in 1990 for future work. All the major assumptions made in deriving the models are identified clearly and are reviewed critically, and their effects on model predictions are discussed. We have used many conservative assumptions in developing our assessment philosophy, and in establishing our models and parameter values to compensate for uncertainties. This ensures that nuclide concentrations and radiological doses to humans and other biota are not underestimated. We have also ensured that they are reasonable and not grossly overestimated. The systems variability analysis approach provides quantitative estimates of the combined variability and uncertainty in model predictions. Finally, we are reviewing and improving BIOTRAC on a continuing basis, using new results from the literature and from our own research programs.

ES.7 CONCLUSIONS

BIOTRAC was specifically developed to simulate the long-term transport of nuclides from an underground vault through the Canadian Shield biosphere. To the extent possible, we have shown that it provides a satisfactory description of nuclide behaviour in the biosphere, and that its predictions are realistic. The model was designed to ensure that radiological and chemical toxic impacts on the environment and humans will not be underestimated. BIOTRAC therefore is an appropriate tool for assessing the geological nuclear fuel waste disposal concept selected by Canada.

1. INTRODUCTION

1.1 CANADA'S NUCLEAR FUEL WASTE MANAGEMENT PROGRAM

The Canadian Nuclear Fuel Waste Management Program (NFWMP) was formally established in 1978 by an agreement between the governments of Canada and Ontario (Joint Statement 1978) to develop and assess a method to safely dispose of fuel wastes. AECL was made responsible for undertaking research and development for disposal in a deep underground vault in the plutonic rock of the Canadian Shield. Ontario Hydro, the provincial utility with the largest nuclear power program in Canada, was made responsible for studies on interim storage and transportation of fuel wastes. In Canada, fuel wastes are currently in the form of used fuel. No decision has been made on whether there will be fuel reprocessing, which would result in a different waste form.

Plutons are stable, intrusive rock masses that are common on the Canadian Shield (Davison et al. 1994a). The wastes would be immobilized and placed in corrosion-resistant metal containers in a vault excavated at a depth of 500 to 1000 m in plutonic rock (Figure 1-1). The containers would be surrounded by a clay-based buffer material. Backfilling of the vault would start during container emplacement. Once the vault has been completely filled, the remaining spaces would be backfilled, and the shafts sealed. No further maintenance would be required following decommissioning and closure. Research conducted as part of the NFWMP has developed a reference design for the containers and vault, and reference materials for the buffer, backfill and other sealing components; the research has also identified specific technologies for the construction of the facility (Johnson L.H. et al. 1994a, Simmons and Baumgartner 1994). Construction, operation, decommissioning, extended monitoring and closure of the disposal facility make up the preclosure phase of the program (Grondin et al. 1994); the postclosure phase is concerned with the behaviour and performance of the facility after closure (Goodwin et al. 1994).

The geological disposal concept for nuclear fuel wastes involves natural barriers supplemented by engineered barriers to ensure the long-term safety of humans and the environment. It is recognized that the wastes may not remain isolated indefinitely. Corrosion of the containers may lead to the dissolution of wastes by groundwater and the transport of nuclides through the buffer, backfill and geosphere to the earth's surface (Figure 1-2, Goodwin et al., in preparation). The effects of such transport could extend far into the future, and their potential impacts must be assessed quantitatively for at least 10 000 a using mathematical models, as indicated by the Atomic Energy Control Board (AECB 1985, 1987). Separate models have been developed to simulate nuclide movement through the vault (Johnson L.H. et al. 1994b), the geosphere (Davison et al. 1994b) and the biosphere (this document). These models are integrated under the control of an executive code SYVAC3 (SYstems Variability Analysis Code - Generation 3; Goodwin et al. 1994) to repeatedly simulate the behaviour of the system as a whole (Figure 1-3). The output is a distribution of nuclide concentrations in the environment and of the radiological doses to humans and other organisms. These calculations provide a quantitative assessment of the performance of the system for comparison with criteria

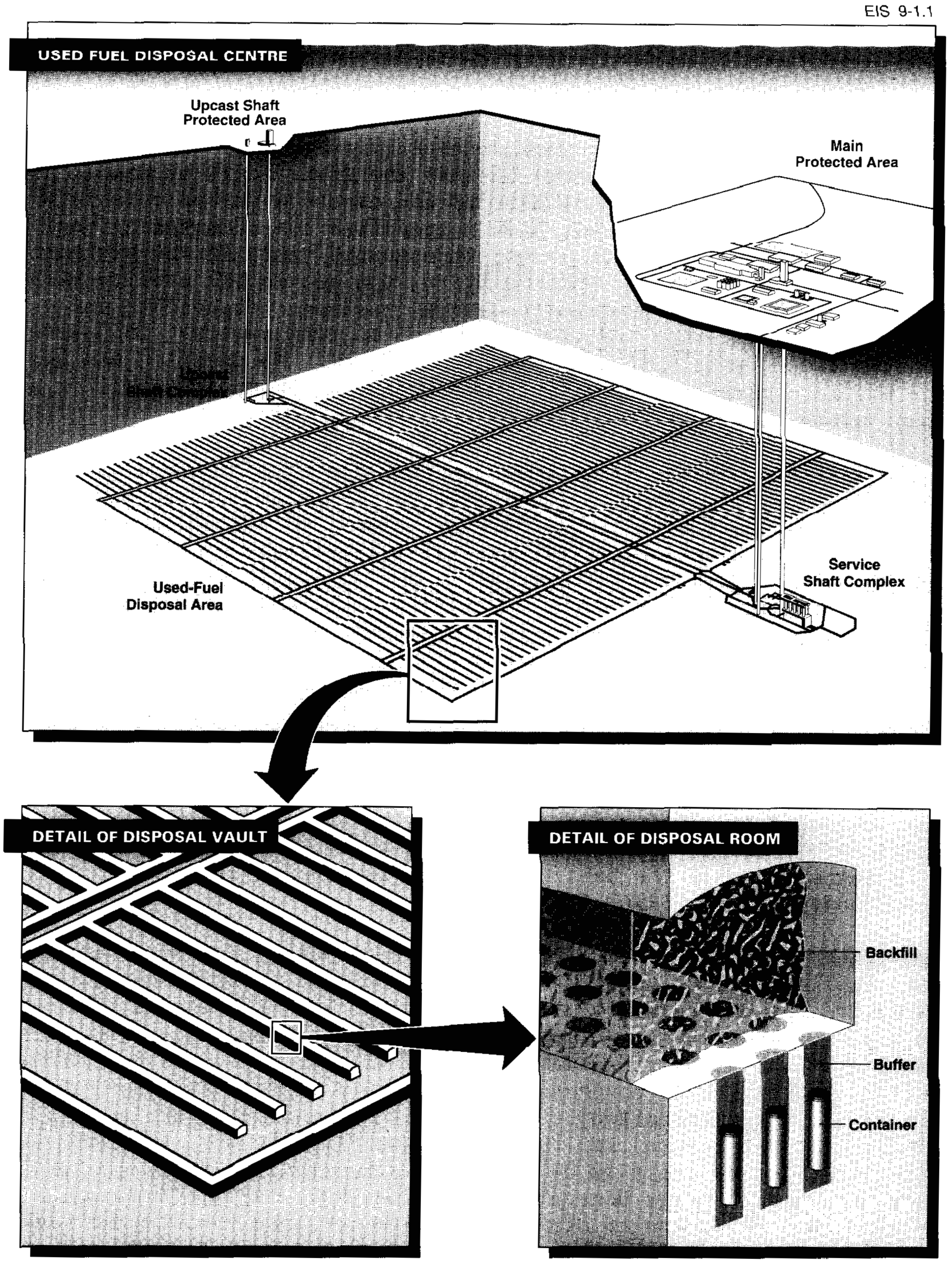

FIGURE 1-1: Schematic Representation of a Conceptual Design of a Nuclear Fuel Waste Disposal Facility Showing Surface Structures, Underground Structures and a Section Through an Emplacement Room

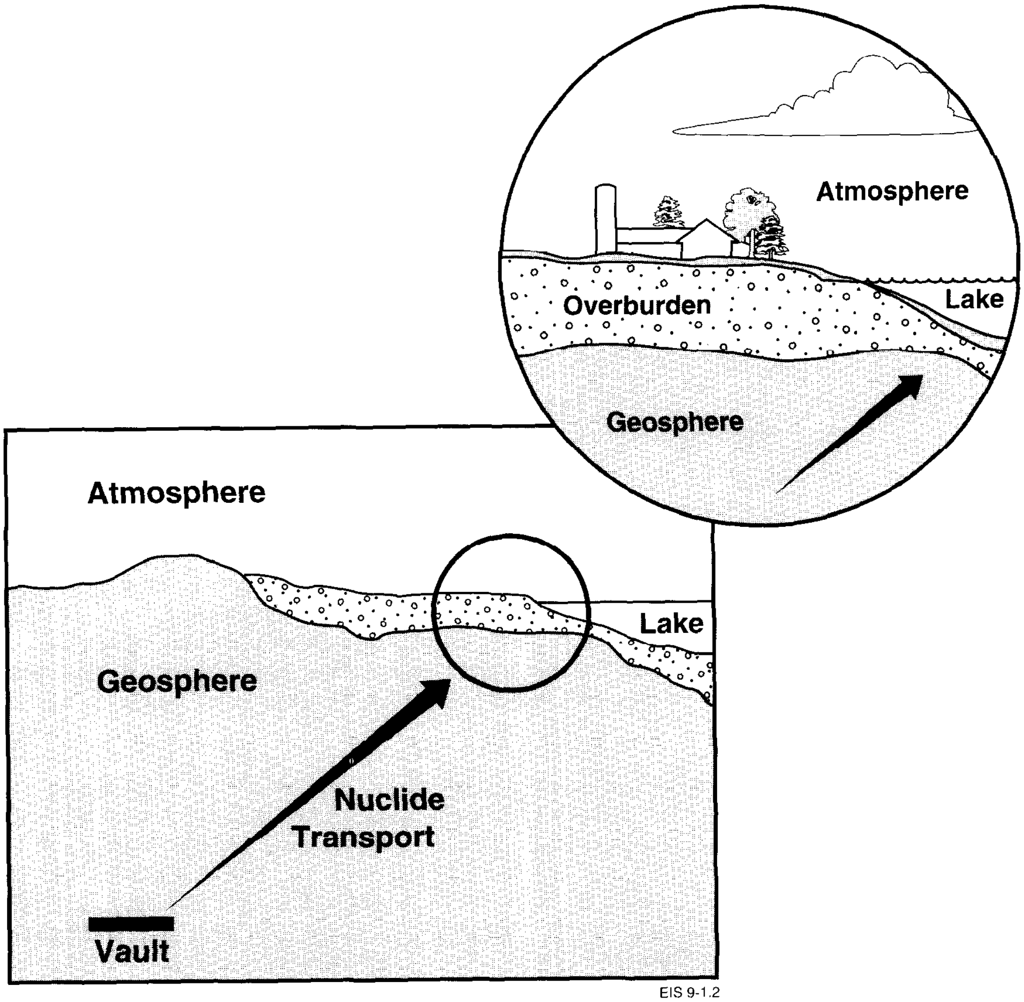

FIGURE 1-2: Schematic Representation of Groundwater Transport of Nuclides from the Vault, 500 to 1000 m Underground, to the Biosphere (Enlarged Insert)

and guidelines set by the AECB (1985, 1987). Radiological effects on humans and the environment are of primary interest, although chemical toxicity is also important.

In 1981, the governments of Canada and Ontario indicated that no site selection for a disposal facility would be initiated until a disposal concept has been accepted (Joint Statement 1981). The concept is now being subjected to a detailed scientific and a more general public review under the Environmental Assessment Review Process (EARP) administered by the Federal Environmental Assessment Review Office (FEARO). AECL is the proponent for these reviews. The terms of reference for the Federal Environmental Assessment Panel have been established (Minister of the Environment, Canada 1989) and the Panel has issued detailed guidelines (Federal Environmental Assessment Review Panel 1992) for preparing the Environmental Impact Statement (EIS) (AECL 1994a).

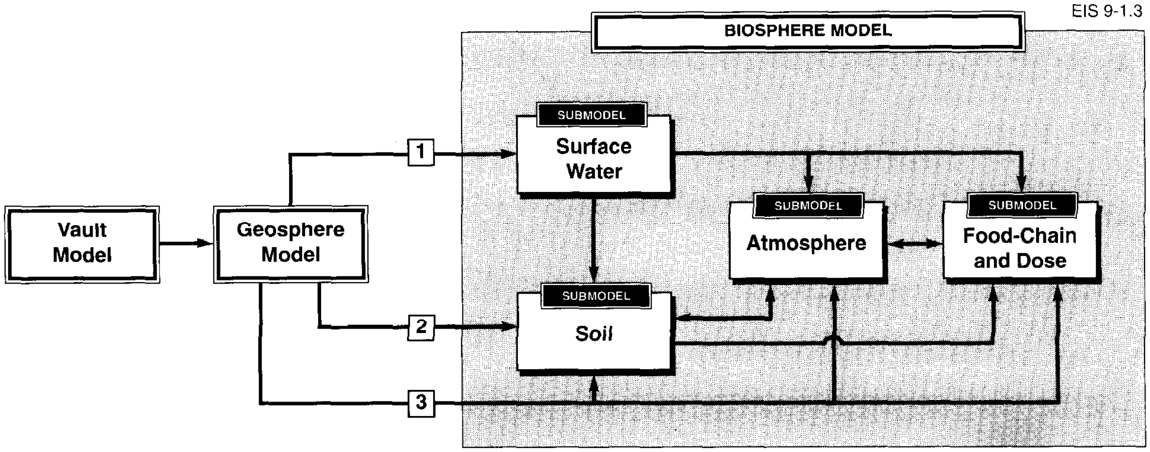

FIGURE 1-3: Schematic Representation of the Three Main Assessment Models (Vault, Geosphere and Biosphere) for the Disposal Concept Assessment, and of the Main Nuclide Transfers Among the Four Submodels of the Biosphere Model (Surface Water, Soil, Atmosphere, and Food-Chain and Dose) and Between the Geosphere and Biosphere Models. Discharges from the geosphere to the biosphere model are: (1) aquatic, (2) terrestrial and (3) well.

The EIS is supported by nine primary references, of which this is one (Figure 1-4). Following the scientific and public reviews of the EIS, the Panel will make recommendations to assist the governments of Canada and Ontario in reaching decisions on the acceptability of the disposal concept. Acceptance may lead to site selection. The search for a site might focus on Ontario because this province is currently the major user of nuclear-generated electricity in Canada.

The vault, geosphere and biosphere models are based on extensive experimental and theoretical work (Iverson et al. 1982, Johnson L.H. et al. 1987, Whitaker 1987, Zach 1985a, Zach et al. 1987), involving many Canadian universities and consulting firms as well as AECL and Ontario Hydro. The entire NFWMP has been reviewed from the outset by an independent Technical Advisory Committee (TAC), consisting of senior scientists nominated by Canadian professional societies. The Bioscience Subcommittee of TAC has focussed its attention on work in the biosphere, and reported its findings in a series of annual reports (e.g., TAC 1992). There have been two interim assessments of the concept; the first was completed in 1981 (Lyon et al. 1981, Johansen et al. 1981, Wuschke et al. 1981) and the second in 1985 (Gillespie et al. 1984, Johansen et al. 1985, Mehta 1985, Wuschke et al. 1985a, 1985b). These interim assessments have been valuable because they helped to identify critical radionuclides, pathways and processes, as well as areas needing further information and understanding (Zach 1985a).

The present document describes the biosphere model developed to assess the performance of the conceptual disposal system in the postclosure phase. For convenience, the model has been given the acronym BIOTRAC (BIOsphere TRansport And Consequence).

FIGURE 1-4: The EIS, the Program Summary and the Nine Environmental Impact Statement (EIS) Primary References

1.2 RADIOLOGICAL ASPECTS

1.2.1 Nuclides of Concern

The vault would contain many radionuclides and chemically toxic elements. Johnson L.H. et al. (1994a) provide a list of the nuclides present in the used fuel, together with their inventories and radioactive decay chains. Where radiological properties are not pertinent to a given discussion, we will refer to the vault contents generically as nuclides.

The amount of each nuclide that may eventually reach the biosphere depends on the performance of the containers, the sealed vault and the geosphere, and is calculated as a function of time in each probabilistic simulation or model run. Table 1-1 lists the nuclides that are considered in the postclosure assessment (Goodwin and Mehta 1994). This list does not include all the nuclides present in the used fuel, nor all of the radionuclides generated through radioactive decay. It includes those nuclides that might conceivably represent a hazard in the biosphere, as established by very conservative screening calculations (Section 1.5.6; Goodwin and Mehta 1994). Some nuclides in the vault inventory are present in such small amounts that they would be unlikely to pose any hazard in the biosphere. Others, although present in substantial amounts in the used fuel, would never enter the biosphere because of radioactive decay and chemical retardation over the very long times and distances required to travel from the vault to the biosphere. Still others can be accounted for by adding their inventories to those of their daughters in cases where the daughter represents a greater risk than the precursor. By screening out these types of nuclides from consideration, the models can be simplified significantly without compromising the assessment. Great care was taken in the screening process to ensure that no nuclide of any significance was dropped from consideration. Most of the nuclides in Table 1-1 are not expected to reach the biosphere, but are nevertheless included to be conservative.

Even though transit times through the geosphere are expected to be very long, certain short-lived radionuclides may still be present at any point in the geosphere or biosphere as the decay products of long-lived precursors. Some of these daughters may be in secular equilibrium with their precursors. At secular equilibrium, the activities of a daughter and its precursor are equal (Wehr and Richards 1967, Zach and Sheppard 1992). Secular equilibrium occurs for daughters with half-lives much less than those of their precursors, at times that are large compared with the half-life of the daughter. The activity of the daughter resulting from the decay of the precursor is at a maximum at secular equilibrium.

The transport of short-lived daughters is not modelled explicitly in the geosphere (Davison et al. 1994b). Activities and concentrations of the daughters at the geosphere/biosphere interface can be easily derived from those of the precursors through the assumption of secular equilibrium (Section 4.4). The decay and ingrowth of radionuclides, and the assumption of secular equilibrium for the biosphere are discussed in Section 2.5.4.

All the nuclides listed in Table 1-1 are considered in the postclosure assessment. However, depending on the exact need, all or only some of them are used in our studies and evaluations.

TABLE 1-1

RADIONUCLIDES WITH DECAY CONSTANTS AND CHEMICALLY TOXIC ELEMENTS CONSIDERED IN BIOTRAC

Radionuclide	Decay Constant, λ^{i} (a^{-1})	Radionuclide	Decay Constant, λ^{i} (a^{-1})	Radionuclide	Decay Constant, λ^{i} (a^{-1})
^{225}Ac	2.53×10^{1}	^{32}P	1.77×10^{1}	^{219}Rn	5.50×10^{6}
^{227}Ac	3.18×10^{-2}	^{231}Pa	2.12×10^{-5}	^{220}Rn	3.94×10^{5}
^{228}Ac	9.90×10^{2}	^{233}Pa	9.38×10^{0}	^{222}Rn	6.60×10^{1}
^{241}Am	1.60×10^{-3}	^{234}Pa	9.06×10^{2}	^{125}Sb	2.50×10^{-1}
^{39}Ar	2.58×10^{-3}	^{234m}Pa	3.11×10^{5}	^{126}Sb	2.04×10^{1}
^{217}At	6.80×10^{8}	^{205}Pb	4.85×10^{-8}	^{126m}Sb	1.92×10^{4}
^{10}Be	4.33×10^{-7}	^{209}Pb	1.87×10^{3}	^{79}Se	1.07×10^{-5}
^{208}Bi	1.88×10^{-6}	^{210}Pb	3.11×10^{-2}	^{32}Si	1.54×10^{-3}
^{210}Bi	5.06×10^{1}	^{211}Pb	1.01×10^{4}	^{126}Sn	6.93×10^{-6}
^{210m}Bi	2.31×10^{-7}	^{212}Pb	5.73×10^{2}	^{90}Sr	2.38×10^{-2}
^{211}Bi	1.70×10^{5}	^{214}Pb	1.36×10^{4}	^{182}Ta	2.20×10^{0}
^{212}Bi	6.03×10^{3}	^{107}Pd	1.07×10^{-7}	^{99}Tc	3.25×10^{-6}
^{213}Bi	7.98×10^{3}	^{210}Po	1.83×10^{0}	^{125m}Te	4.36×10^{0}
^{214}Bi	1.83×10^{4}	^{211}Po	4.23×10^{7}	^{227}Th	1.35×10^{1}
^{14}C	1.21×10^{-4}	^{212}Po	7.17×10^{13}	^{228}Th	3.62×10^{-1}
^{41}Ca	4.95×10^{-6}	^{213}Po	5.21×10^{12}	^{229}Th	9.44×10^{-5}
^{113m}Cd	5.09×10^{-2}	^{214}Po	1.33×10^{11}	^{230}Th	9.00×10^{-6}
^{135}Cs	3.01×10^{-7}	^{215}Po	1.23×10^{10}	^{231}Th	2.38×10^{2}
^{221}Fr	7.59×10^{4}	^{216}Po	1.46×10^{8}	^{232}Th	4.93×10^{-11}
^{223}Fr	1.67×10^{4}	^{218}Po	1.20×10^{5}	^{234}Th	1.05×10^{1}
^{3}H	5.61×10^{-2}	^{238}Pu	7.90×10^{-3}	^{206}Tl	8.68×10^{4}
^{182}Hf	7.70×10^{-8}	^{239}Pu	2.88×10^{-5}	^{207}Tl	7.63×10^{4}
^{129}I	4.41×10^{-8}	^{240}Pu	1.06×10^{-4}	^{208}Tl	1.19×10^{5}
^{40}K	5.42×10^{-10}	^{241}Pu	4.81×10^{-2}	^{209}Tl	1.65×10^{5}
^{81}Kr	3.30×10^{-6}	^{242}Pu	1.84×10^{-6}	^{232}U	9.63×10^{-3}
^{85}Kr	6.48×10^{-2}	^{223}Ra	2.21×10^{1}	^{233}U	4.37×10^{-6}
^{93}Mo	1.98×10^{-4}	^{224}Ra	6.93×10^{1}	^{234}U	2.83×10^{-6}
^{93m}Nb	5.10×10^{-2}	^{225}Ra	1.71×10^{1}	^{235}U	9.85×10^{-10}
^{94}Nb	3.41×10^{-5}	^{226}Ra	4.33×10^{-4}	^{236}U	2.96×10^{-8}
^{59}Ni	9.24×10^{-6}	^{228}Ra	1.21×10^{-1}	^{238}U	1.55×10^{-10}
^{63}Ni	7.22×10^{-3}	^{87}Rb	1.47×10^{-11}	^{90}Y	9.50×10^{1}
^{237}Np	3.24×10^{-7}	^{187}Re	1.39×10^{-11}	^{93}Zr	4.53×10^{-7}

Chemically Toxic Elements

Br
Cd
Cr
Cs
Mo
Sb
Se
Sm
Tc

1.2.2 Radioactive Decay Constants

Each radionuclide has a characteristic radioactive decay constant, λ^i, that defines the fraction of radioactive material decaying per unit time. Decay constants are radionuclide-specific, with the associated half-lives varying from a fraction of a second to millions of years. The radioactivity per unit mass is high for radionuclides with large values of λ^i. However, the decay rate is an incomplete measure of the potential hazard of a radionuclide. Biomobility and the types and energies of radiation emitted during decay are also important (Myers 1989, Zach and Sheppard 1992).

Decay constants are used in many places in BIOTRAC. The values that we have adopted for the postclosure assessment are listed in Table 1-1. Radioactive decay has been studied intensively for many years, and decay rates for most radionuclides are known precisely. The values shown in Table 1-1 have been based on radiological half-lives published by the International Commission on Radiological Protection (ICRP 1983).

1.2.3 Human Protection and the Annual Effective Dose Equivalent

One of the key concerns in nuclear fuel waste management is protecting humans from radiation effects. Although the vault will contain some chemically toxic elements, there are many more radionuclides (Table 1-1) that might lead to radiation exposure. The main quantity calculated for human radiation protection is the committed effective dose equivalent, or *simply* dose. This measure was introduced by the ICRP (1977, 1979). Since its inception in 1928, the ICRP has been dedicated to the radiation protection of humans. The ICRP has exercised its influence through a series of recommendations such as ICRP 26 (ICRP 1977). Most of these recommendations have found broad acceptance, although sometimes with a delay.

For the postclosure assessment, we have followed ICRP 26 (ICRP 1977) and related recommendations for calculating doses to humans. Further details concerning this, the interaction of radiation with biological tissue and the induction of health effects are discussed by Myers (1989) and by Zach and Sheppard (1992). The committed effective dose equivalent is briefly discussed below. At this point, it is important to point out that dose conversion factors (DCFs) (Section 8.5.2) are key parameters for calculating doses to humans and other biota. In essence, these factors represent the dose per unit intake of, or exposure to, a given radionuclide. However, the committed effective dose equivalent applies to humans only.

The basic dosimetric quantity is the dose, D (Gy), defined as the amount of energy absorbed from radiation in a mass of material such as human tissue (ICRP 1977). Since D does not fully account for the biological consequences of various types of radiation, the dose equivalent, H_T (Sv), is used instead. For a given organ or tissue, H_T is given by the product of D and a radiation quality factor, Q (unitless), which describes the effectiveness of a given type of radiation in causing biological damage. An effective dose equivalent, H_E (Sv), for the whole body can be found by summing the H_T values for all tissues and organs, weighted by organ factors, W_T (unitless), according to their radiosensitivity.

Radionuclides inhaled or ingested into the body continue to irradiate the body until they have decayed or are eliminated with waste products. The body is therefore committed to receiving a dose over an extended period of time following the intake of a radionuclide. The total internal dose is usually calculated over a 50-a period, and is called the 50-a committed effective dose equivalent, H_{50} (Sv). Since committed doses were originally developed to assess the occupational safety of radiation workers, the 50-a interval was chosen to approximate the working life span of humans (ICRP 1979). Currently, the life expectancy of Canadians is about 75 a. However, lifetime committed doses are difficult to calculate because they include all life stages from fetus to old age. On the basis of reasonable assumptions about age dependence, Johnson J.R. (1982a) concluded that the 50-a interval used here would overestimate the lifetime committed dose.

The ICRP has recently introduced age-dependent DCFs, with a 70-a commitment for some radionuclides (ICRP 1989). These factors are suitable for assessing accidental exposure of the public, but less so for the chronic exposure situations one might encounter in nuclear fuel waste management. Evaluation of the ICRP 56 (ICRP 1989) values has shown that our DCFs may lead to underestimation of doses for some radionuclides and modes of exposure (Zach and Sheppard 1992). We have not used ICRP 56 DCFs because of the limited number of values available and the conclusion reached by Johnson J.R. (1982a) that our values would overestimate the lifetime committed dose.

Traditionally, H_{50} is defined as the internal dose accumulated over 50 a following a single intake of radioactive material (ICRP 1977). However, releases from a disposal vault would result in the chronic intake of radionuclides. Conveniently, H_{50} as defined for an instantaneous input, I_o, is mathematically the same as the dose equivalent received in the fiftieth year following an intake I_o of material in each of the preceding 50 a, provided body changes with age are ignored (Healy 1981, Johnson J.R. 1982b, 1985). Therefore, the extensive information available on H_{50} can be used to predict the annual dose to an individual continuously exposed to a contaminated environment. This dose rate will be constant over the lifetime of the individual as long as the radionuclide intake remains constant. This will likely be the case for nuclear fuel waste management because radionuclide flows out of the geosphere, and radionuclide concentrations in the biosphere, would change only very slowly with time.

External dose does not involve a dose commitment. Rather, an individual receives a dose only when actually exposed to an external radiation field. The annual external dose is found by summing all the external doses received in the course of the year. The annual dose will be constant from year to year as long as the radionuclide concentrations in the environment remain constant. Internal and external doses can be added to produce an annual effective dose equivalent. One of the main aims of the biosphere model is to estimate this dose rate ($Sv \cdot a^{-1}$) for comparison with regulatory criteria, as presented by Goodwin et al. (1994).

The risk from radiation exposure and the calculation of doses are under constant review by the scientific community and various committees (e.g., UNSCEAR 1988, BEIR 1990). New knowledge and interpretations have led to revised recommendations by the ICRP (ICRP 1991a), and may lead to still further revisions. We have not used the DCFs based on these recommendations (ICRP 1991b) since an evaluation of them showed that the DCFs we use do not lead to systematic underestimation of doses (Zach and Sheppard 1992). However, for some radionuclides, notably ^{129}I, doses based on our factors would be slightly lower than those based on the revised values, as indicated in the postclosure assessment (Goodwin et al. 1994).

There have been several important recent changes in DCFs, including age dependence, a 70-a commitment (ICRP 1989), and revised risk estimates (ICRP 1991a). Most of these changes would have relatively minor effects on our dose predictions, based on ICRP 26 (ICRP 1977) risk estimates and methodologies. Eventually, age-dependent DCFs for the public based on ICRP 60 (ICRP 1991a) will appear, and will be used by us for further studies in the NFWMP.

1.2.4 Environmental Protection

ICRP 26 (ICRP 1977) assumes that environmental protection, i.e., the protection of populations of various plants and animals, can be assured by the measures adopted to protect humans. This assumption has been convenient because it implies that environmental protection does not need to be addressed separately. The assumption has been challenged repeatedly, but several reviews and evaluations (Myers 1989, IAEA 1992, UNSCEAR 1992) lend support to it. Nevertheless, it has become clear that radiation protection of the environment deserves separate consideration (Federal Environmental Assessment Review Panel 1992). One difficulty is that there are no well-established procedures, criteria or guidelines, as is the case for human radiation protection. This difficulty is increased by the numerous organisms and biological or ecological end points that might be considered.

As discussed in detail in Chapter 13, we have developed a threefold methodology to address environmental protection. The first approach assumes that humans can serve as a sensitive indicator species for other biota, and thus protection of humans at the individual level also ensures protection of other organisms at the population or species levels. This is in essence the ICRP (1977) assumption for radiological protection. The second approach consists of evaluating surface water and soil concentrations of nuclides predicted by BIOTRAC, and comparing them with the relevant regulatory criteria and guidelines, as well as with environmental baseline concentrations and their variability (Amiro 1992a). We do not focus on concentrations in air because they are a function of concentrations in surface water and soil, given an underground source of contaminants. The evaluation of environmental concentrations is a highly integrated assessment approach because the well-being of plants and animals depends on the quality of the surface water and the soil. Furthermore, environmental concentration can be readily monitored (Simmons et al. 1994). Finally, the third approach relies on predicting radiological doses to a set of generic target organisms and comparing these doses to doses for which the occurrence of effects has been documented or their absence noted (Amiro 1992b). We also show how this approach can be applied to specific rather than to generic target organisms.

As in the case of human protection, we are concerned here only with presenting assessment methodologies. Goodwin et al. (1994) describes the application of these methodologies and the postclosure assessment results.

1.3 CHEMICAL ASPECTS

As shown in Table 1-1, we have identified nine potentially chemically toxic elements for quantitative assessment based on the assessment by Goodwin et al. (1987a). Most of these elements occur as both radiologically and chemically toxic nuclides, but bromine (Br), chromium (Cr) and samarium (Sm) are of concern only because of their chemical toxicity. In a sense, these elements can be regarded as radiologically stable nuclides and they can thus be modelled in the same manner as all the other nuclides. This and other aspects related to chemical toxicity are discussed in detail in Chapter 13, and the corresponding postclosure results are presented by Goodwin et al. (1994).

Because the evaluation of radiological and chemical effects on humans and the environment has traditionally proceeded along different avenues, it is difficult to integrate the two into a single assessment approach and risk estimate. Insufficient data are available to decide on compensation, additivity or synergism of effects. Therefore, we have developed separate but conservative methodologies (Section 1.5.6).

1.3.1 Human Protection

Although the movement of chemically toxic nuclides can be modelled in the same manner as that of other nuclides, there are important differences. Chemically toxic nuclides, like radionuclides, can induce cancer, hereditary and other defects (Friberg et al. 1979), but no detailed dosimetric procedures are available to estimate consequences (Butler 1980). This means that no human doses can be calculated. For chemical contaminants, health protection is usually accomplished by regulating contaminant concentrations in environmental media, such as water and soil, and we have followed this procedure for the postclosure assessment using BIOTRAC predictions. In many instances, health protection is assured through regulatory requirements for environmental protection.

1.3.2 Environmental Protection

The situation for environmental protection is similar to that for human protection. Thus, environmental protection is accomplished by regulating the concentrations of chemically toxic contaminants in surface water and soil. We have followed this procedure for the postclosure assessment by comparing the BIOTRAC predictions with environmental criteria and guidelines. Furthermore, predicted concentrations can also be compared with baseline concentrations in the environment (Section 13.1.3, Amiro 1992a).

1.4 THE BIOSPHERE AND BIOSPHERE MODEL

In the present context, the biosphere includes those parts of the terrestrial environment that lie above the water table (unsaturated soils and the atmosphere) as well as surface waters, including wetlands, and the mixed layer of lake sediments. These are the parts of the environment that contain abundant living organisms, and that are readily accessible to humans. Because the disposal vault, if approved, might be located in plutonic rock in Ontario, we focus on a biosphere characteristic of the Canadian Shield in Ontario. However, Canadian Shield regions outside Ontario are not expected to be substantially different. The region of saturated overburden and rock beneath the water table, excluding surface waters and mixed sediments, is treated as part of the geosphere.

The biosphere as a whole also includes the oceans. However, the AECB (1985) has stated that the performance of the disposal system will be judged, as far as humans are concerned, on the basis of its impacts on the critical group (Section 1.5.4). By definition, members of the critical group, because of their lifestyle and place of residence, are at the greatest risk from nuclides entering the biosphere. We assume that the critical group lives near the point at which nuclides discharge to the biosphere, and that members of the group consume only local food and water. Oceans could not provide a direct exposure pathway to such individuals, and are, therefore, not included in the biosphere model. Furthermore, concentrations of nuclides from a vault in the ocean would always be lower than concentrations in local food and water.

The biosphere provides the focus for the postclosure environmental assessment of the disposal facility because it is here that any consequences would be felt. Furthermore, the criteria against which the performance of the disposal system will be judged are mainly biosphere criteria. The AECB (1987) stated as a basic regulatory requirement that "radioactive waste disposal options shall be implemented in a manner such that there are no predicted future impacts on the environment that would not be currently accepted." In addition, "the predicted radiological risk to individuals from a waste disposal facility shall not exceed 10^{-6} fatal cancers and serious genetic effects in a year." Risk is to be calculated from the predicted doses using a fixed risk conversion factor.

Accordingly, one of the main objectives of the biosphere model is to calculate nuclide concentrations in the various environmental compartments, and to estimate the radiological dose to humans from all the credible exposure pathways. The model achieves this by simulating the transport of nuclides from the point where they discharge from the geosphere through the biosphere to humans (Figure 1-5). We have developed four separate but closely linked submodels representing the surface water (Bird et al. 1992), the soil (Sheppard M.I. 1992), the atmosphere (Amiro 1992b) and the food chain (Zach and Sheppard 1992). Permanent and temporary, or seasonal, wetlands are considered through the surface water and soil submodels. Nuclides emerging from the geosphere are assumed to enter a lake or a well, and may also enter the lowest layer of the soil profile. The soil root zone may become contaminated as the nuclides move upward by capillary action or as irrigation water is applied to the surface. Nuclides suspended from the soil and the lake may become entrained in the atmosphere. Food supplies grown in this environment may take up nuclides. Crops grown on the soil take up nuclides through their roots, or through their leaves following deposition from the atmosphere. Animals may ingest nuclides with water, food or fodder and soil. Humans living in this environment may be subject to nuclide exposure from many sources. They may receive an internal dose through ingestion of contaminated food and water and by inhalation; they may receive an external dose through immersion in air and water, and by exposure to contaminated soil and building materials.

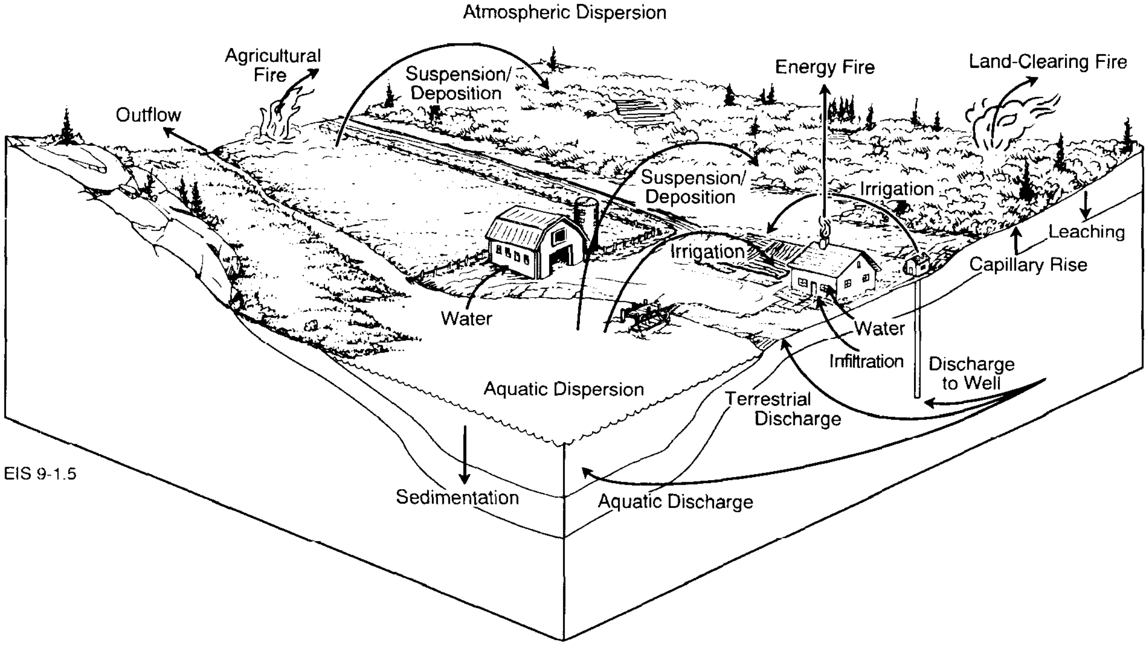

FIGURE 1-5: Main Environmental Transport Processes Considered for the Postclosure Phase of the Disposal Concept Assessment

Besides radiation doses for humans, BIOTRAC also allows the calculation of concentrations of nuclides in surface water and soil, and radiological doses to non-human organisms to help ensure protection of the environment (Section 1.2.4). As in the case of humans, environmental protection must focus on the area occupied by the critical group, where the risk from nuclides released from the vault is highest. For this reason, the biosphere model, primarily designed for human radiation protection, can also be used for environmental protection. However, the AECB has not provided quantitative regulatory requirements for environmental protection, and thus potential environmental effects must be evaluated in other ways, as discussed in detail in Chapter 13.

Another objective of BIOTRAC is to calculate concentrations of chemically toxic elements in surface water and soil to help ensure protection of both humans and the environment (Section 1.3). Here too, concerns focus on the area occupied by the critical group, and thus the biosphere model designed for radiation protection is also suitable for assessing chemical toxicity. Appropriate guidelines or criteria are available, or can be established, to evaluate model predictions, as discussed more fully in Chapter 13.

1.5 ASSESSMENT PHILOSOPHY

The form that any model takes depends strongly on the processes it is meant to simulate and the questions it is intended to answer. BIOTRAC was developed specifically to assess the postclosure impacts associated with a used-fuel disposal vault. The geological disposal concept is unique in a number of ways, among them the underground location of the source of potential contaminants, the great length of time over which wastes must be isolated from the biosphere and its biota, and the need to assess the concept before a vault location is selected. Several underlying philosophical approaches have been adopted to tailor the assessment to these special needs. Various aspects of the assessment philosophy are discussed below to explain the way in which the biosphere model was developed and the form that it currently takes.

1.5.1 Scenario Analysis

Nuclides from the vault could reach the environment and humans in a number of ways, some more likely than others. The important processes and pathways included in our assessment models were identified and selected through an extensive scenario analysis. By the term scenario, we mean a combination of factors (features, events and processes) that could affect the isolation of the waste in its disposal facility (Cranwell et al. 1987, NEA 1989). Scenarios were constructed using a systematic procedure that involved searching for all possible factors; screening and eliminating factors on the basis of probability of occurrence, physical reasonableness and regulatory criteria; and combining the remaining factors into scenarios in all possible ways (Goodwin et al., in preparation). The focus in scenario analysis has been on human radiation protection.

The analysis identified one scenario as being by far the most likely to occur - slow degradation and failure of the waste containers, release of nuclides from the waste into the groundwater within the vault, diffusion through saturated buffer and backfill materials, convection and diffusion through faults and interconnected pores in the geosphere to surface water and soil, and finally environmental and human exposure through a multitude of pathways in the biosphere. The postclosure assessment focuses on this groundwater intrusion scenario, which is referred to as the central group of scenarios because it includes all but one of the potential alternative scenarios identified (Goodwin et al., in preparation). This alternative scenario is concerned with open or unsealed boreholes. The central group of scenarios has been assigned a probability of occurrence of one in the postclosure assessment (Goodwin et al. 1994).

BIOTRAC has been specifically designed to treat the central group of scenarios. The biosphere factors considered in the scenario analysis are listed in Appendix C and by Goodwin et al. (in preparation). The list was generated from literature reviews, expert opinion and brainstorming sessions. The factors were classified in several ways to help identify any that were initially missed. When the list was as comprehensive as possible, factors were either included or not included in the central group of scenarios, depending on their expected importance and probability of occurrence. Those selected have been incorporated into the model for quantitative assessment and are described in Chapters 4 to 8. They include the major exposure pathways described in Section 1.4, as well as many less common pathways. The factors not selected are discussed briefly by Goodwin et al. (in preparation) and in the references listed in Appendix C, which also provide the reasons for their exclusion. A few factors with the potential to affect the dose to humans (notably, the pathways associated with continental glaciation (Davis 1986, Elson and Webber 1991)) have not been incorporated into the model for various reasons. The impact of these factors is assessed in Chapter 12.

In the central group of scenarios, nuclides from the vault enter the biosphere from aquifers below. Traditional biosphere models have been designed to treat above-ground releases, and are unable to simulate an underground source. The biosphere model developed for the postclosure assessment has been designed to treat the unique pathways and processes associated with groundwater discharge. This is particularly true of the soil and atmosphere submodels (Chapters 6 and 7).

1.5.2 Systems Variability Analysis

The performance assessment of the disposal facility is based on a probabilistic analysis directed by the executive code SYVAC3 using Monte Carlo simulation techniques (Dormuth and Quick 1980, Goodwin et al. 1987b). The systems approach is used to predict how the vault, geosphere and biosphere will respond as a whole to a given nuclide release. It also provides a way to quantify the variability and uncertainty in the predictions of the models, which may be inexact for several reasons. Because the assessment must be conducted before a disposal location is selected, site-specific values cannot be assigned to the parameters describing the state of the vault, geosphere and biosphere system. Even if a specific location were being evaluated, it might not be possible to assign values to all the input parameters precisely. The processes being simulated may not be fully understood, and may be approximated or simplified in the models. Parts of the system may be inherently variable, and the parameters describing these parts should reflect this. Because the predictions are made for times far into the future, the models must take into account changes that may occur in the system over very long periods of time (Section 1.5.7).

Uncertainty and variability in the data and the processes are accommodated through the use of probability density functions (PDFs) to define many of the parameter values. This requires specification of distribution types, usually with a measure of central tendency (most probable value), a measure of variation, and truncation values and parameter correlations to avoid unreasonable values. At the beginning of each simulation, SYVAC3 selects a possible state of the system by randomly sampling a value for each parameter from its specified PDF (Figure 1-6). This set of values is then used within the model equations to calculate the nuclide transfer from the vault to the biosphere and to estimate the consequence, i.e., a set of concentrations, doses or risks, for that state. SYVAC3 repeats the selection of parameter values and estimation of consequences, typically one thousand times or more, and summarizes the results statistically. The output is the range of possible consequences and their frequency of occurrence, from which the uncertainty in the results can be estimated (Goodwin et al. 1994).

Substantial amounts of computer time are needed to perform the many simulations needed for a probabilistic assessment. To ensure that the computer requirements do not become impractical, the various transport processes were modelled in as simple and efficient a manner as possible, consistent with a sufficiently accurate and detailed description of nuclide movement through the biosphere.

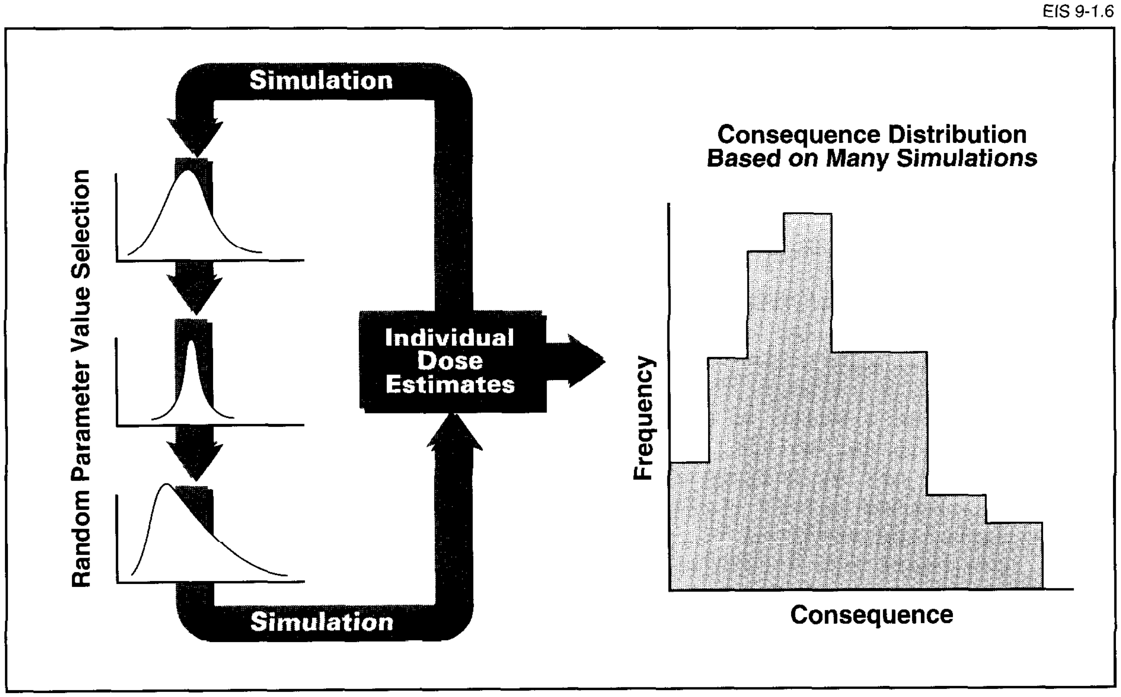

FIGURE 1-6: Schematic Representation of the Principles of Monte Carlo Simulation Showing Random Parameter Value Selection from a Series of Probability Density Functions, Calculation of Individual Consequence Estimates for Each Simulation, and Compilation of These Estimates into a Probability Distribution for the Combined Simulations

1.5.3 Generic Nature of the Biosphere Model

The NFWMP has developed a concept for geological disposal, but no site will be selected until the concept has been accepted (Joint Statement 1981). The performance assessment has been designed to demonstrate the methodology rather than the safety of a specific site. The three integrated assessment models (for the vault, the geosphere and the biosphere) reflect the lack of a specific site location in different ways, depending on their requirements and the scientific information available. The vault model (Garisto and LeNeveu 1991, Johnson L.H. et al. 1994b) is based on a particular reference design for the waste form, the containers and the vault itself (Johnson L.H. et al. 1994b). This design and the model are largely independent of the location of the vault, except at the point where the model couples with the geosphere and for groundwater characteristics. The geosphere model (Davison et al. 1994b) is based on the geology and hydrology of the Whiteshell Research Area (WRA) near Pinawa, Manitoba. Nuclide transport through the geosphere depends so strongly on groundwater flow patterns that consistent values for the local topography and the fracture structure of the rock must be specified before meaningful calculations can be made. It was decided to use information from a real location to set those parameters, rather than hypothesizing generic values. The WRA was chosen as an example because much is known about its groundwater flow pattern. It must be emphasized, however, that the WRA is not under consideration as a potential location for a disposal facility; it is being used for the postclosure assessment case study to evaluate the disposal concept and demonstrate that methods exist to characterize and assess geosphere transport at a given location on the Canadian Shield. The methods developed to characterize and model the WRA could be applied directly to other locations on the Canadian Shield.

The approach taken in the biosphere in the absence of a specified site was to develop a generic model. The distributions chosen for the biosphere parameters encompass the full range of values that could be encountered on the Ontario portion of the Canadian Shield. Exceptions to this rule are the parameters that are common to both the geosphere and biosphere models (Section 4.3). These were set equal to their WRA values to ensure consistency between the models. The surface water submodel (Chapter 5), which provides the main interface with the geosphere, therefore has some site-specific features. In general, the suite of biosphere parameter values sampled in each simulation of the model represents no particular location, but one of a range of possible Canadian Shield locations in Ontario. The PDFs also include values that are characteristic of the biosphere at the WRA. The biosphere model therefore includes the WRA without specifically modelling it.

1.5.4 Human and Environmental Exposure Situation

The individual human for whom doses are calculated in the postclosure assessment belongs to a postulated relatively homogeneous group of people expected to receive the greatest exposure because of their location and lifestyle (AECB 1987). This all-inclusive group is called the critical group, a concept commonly employed when evaluating doses to the public from conventional nuclear power installations. Collective or population doses need not be calculated for the concept assessment (AECB 1985).

Given a performance assessment that extends far into the future, it is not possible to precisely identify the critical group and its lifestyle because of uncertainties about population distributions, living habits and climate. Instead, the lifestyle of the critical group has been based on present human behaviour using conservative (Section 1.5.6), yet reasonable assumptions. The diet and metabolic characteristics of the group are based on present knowledge. We have not considered possible future changes in technology, culture, or human physiology or anatomy, because such changes are unpredictable, as indicated by the AECB (1987).

We assume that the critical group is composed of a sequence of self-sufficient rural households, of variable size, living near the point at which nuclides discharge to the biosphere (Zach and Sheppard 1992). The effects of dispersion and dilution are minimal at this point, and hence environmental concentrations and doses to humans and all the other organisms are at their highest. At a given time, the household could be one in a succession of families that has inhabited the location continuously since the vault was decommissioned and closed. Alternatively, it could have pioneered the location after it had been uninhabited for some time. In either case, we assume that members of the household live their entire lives at the discharge zone, having access only to those parts of the biosphere that are most highly contaminated. They are assumed to be entirely self-sufficient, drawing all of their resources, including food, water, heating fuel and building materials, from the local environment. Summer produce is stored for winter consumption, so potentially contaminated food is eaten throughout the year. The critical group depends mainly on agriculture for food, but may also harvest wild foods. They are more self-sufficient than is customary for most people today, but this is conservative and consistent with the critical group concept.

Members of the critical group exhibit some individuality, particularly with respect to diet, which is described by distributed parameters. They also choose their source of water (a well or a lake), and the extent to which they practise aerial irrigation. These choices are based on current behaviour patterns of people, including aboriginals, on the Canadian Shield.

The biosphere inhabited by the critical group can be visualized as a typical Canadian Shield environment consisting of rock outcrops, bottom lands with pockets of soil, wetlands and surface water bodies, and uplands with meadows, bush and forests. The biosphere model does not require the actual distribution and sizes of most of these features to be specified. We simply assume that the land in the area is sufficient to grow the crops and raise the animals required by the critical group. Similarly, we assume that there is sufficient forest to provide the required resources to build and heat homes. The critical group is assumed to draw all of its resources from the most contaminated part of the environment.

In calculating doses, the critical group is assumed to have a garden available for raising plant foods, such as vegetables, cereals, fruits and berries. It is also assumed to have a forage field that provides feed or forage for domestic and wild food animals. Thus, both the garden and forage field may include wild foods. A wood lot and a peat bog are two additional areas assumed to be available for supplying building material or fuel.

For dose prediction purposes, each member of the critical group is assumed to be represented by reference man as defined by the ICRP (1975). Reference man is a typical adult individual, about 20 to 30 years old, 170 cm high and weighing 70 kg. The gastrointestinal tract, lungs and other organs are assumed to obey specific physiological models that make it possible to calculate the dose received from the rate of exposure to radiation. Reference man includes both male and female features that are important in dose prediction.

The focus on ICRP reference man in the assessment reflects the large amount of information available on radionuclide metabolism in the human body and consequent radiation exposure. Much of the information pertains to adults, but doses predicted for adults and infants tend to be similar (Zach and Mayoh 1984, Grondin et al. 1994). Furthermore, the radiation risks for reference man (lethal cancers and heritable genetic effects) are based on human population data, which include both sexes and all ages. Accordingly, our dose calculations for ICRP reference man are state-of-the-art, and are representative of the critical group as a whole.

The critical group concept is primarily related to human radiation protection. Because contaminant concentrations can be assumed to be greatest in the area occupied by this group, the concept is also relevant for chemical protection of humans and the environment (Sections 1.2.4 and 1.3).

ICRP reference man has been specifically defined for assessing human radiation exposure, but there are no analogous non-human organisms for this purpose. We have therefore defined four generic target organisms for assessing radiation exposure (Section 13.3.1). These plants, fish, birds and mammals live their entire lives in this potentially most contaminated area. Thus, their exposure situation is entirely analogous to that of reference man. Since we have based chemical protection of both humans and the environment on nuclide concentrations in surface water, soil and air, no target organisms are required for assessing chemical protection (Section 1.3). However, such organisms are considered indirectly in the regulatory criteria and guidelines used to evaluate postclosure predictions in Goodwin et al. (1994).

1.5.5 Time Scale and Model Applicability

The AECB (1987) has defined the period over which mathematical models must be used for quantitatively demonstrating compliance with its regulatory criteria to 10 000 a. Where predicted risks do not peak before 10 000 a, there must be reasoned arguments that at longer times the rate of nuclide release to the environment will not increase suddenly and dramatically, and that individuals will not be subject to acute radiological risks. The reasons for specifying the 10 000-a limit are increased uncertainty in environmental conditions over time, and the increased likelihood of continental glaciation (Section 3.3.2). We have run the assessment models for up to 100 000 a, but without considering glaciation (Goodwin et al. 1994). Although confidence in the calculations decreases as the simulation period increases, the mathematical results provide one way to gauge consequences far into the future.

The parameters sampled at the beginning of each SYVAC3 simulation are held constant throughout the simulation. Accordingly, the state of the biosphere is assumed not to change within each simulation, or even to exhibit its characteristic fluctuating behaviour during the assessment period (Davis 1986). Since the parameter PDFs are based on data gathered under current biosphere conditions, the calculations have been made for a biosphere in a permanent interglacial state. Although the PDFs represent spatial variability, they likely cover much of the temporal variability that might be expected to occur at a given location in the absence of glaciation (Section 1.5.7). We discuss environmental change generally in Chapter 3, and glaciation in Chapter 12.

BIOTRAC was designed with the AECB 10 000-a time limit in mind, and is not suited for making predictions beyond about 100 000 a even if interglacial conditions are assumed to persist. The model does not allow for the lateral transfer of soil or sediment (or the nuclides associated with them) by wind or water erosion, or other geomorphological processes that might become important on long time scales. Moreover, in some instances, the nuclide inventory in a source compartment is not reduced when nuclides move to another compartment (Section 2.3.3). The absence of lateral transfer in the model and of an exact mass balance has little effect on concentration in the short term, but predicted concentrations become unreasonably high over geological time scales. This would result in concentration and dose estimates that may become overly conservative if the model is applied over time scales greatly exceeding 10 000 a.

We have made no attempt in the model to incorporate temporal changes in man's cultural or social behaviour, in his physiology, or in the changes in the biosphere caused by anthropogenic effects (Davis 1986). Very dramatic changes along these lines are quite possible over the duration of the assessment period, but such changes are not possible to predict, and it would be futile to try to model them. They are accounted for to a limited extent through the critical group concept, and by assessing all the credible exposure pathways on the basis of present-day human behaviour.

As noted in Section 1.2.3, one of the main predictions of the biosphere model is dose to a member of a critical group and to other organisms on an annual basis. BIOTRAC is therefore designed to predict concentrations averaged over periods of one year. This is achieved through the use of annually averaged parameter values and through the structure of the model itself. Processes occurring on time scales shorter than one year are not modelled unless they affect annual doses.

1.5.6 Conservatism

The biosphere model has been developed to simulate the transport of nuclides through the environment to humans as realistically as possible. However, the biosphere is extremely complex, and not all of the transport processes are fully understood. In areas where our knowledge is incomplete, and where realistic models cannot be formulated or validated, we have made conservative assumptions in accordance with AECB (1985) recommendations. Here and elsewhere in this report the term conservative will be used to describe an assumption or result that overestimates the true consequences. Examples of conservatism can be found in three key aspects of BIOTRAC:

1. Assessment philosophy - The concept of the critical group was adopted, in part, to overcome the difficulties in defining appropriate exposure pathways to humans far in the future and the likelihood of human presence when and where nuclides reach the biosphere.

2. Models - Detailed modelling of incompletely understood processes is avoided by making conservative assumptions. For example, nuclides can be suspended into the atmosphere by a variety of mechanisms, both natural and anthropogenic, most of which are not well understood. We model these mechanisms using a dust loading approach (Amiro 1992b) in which all aerosols from any source are conservatively assumed to be contaminated to the same extent as the local soil.

3. Input data - Parameters such as the sediment transfer rate, α^i, are difficult to measure, and data are often not available to define PDFs for all nuclides of interest. In such cases, values are chosen that are believed to be conservative.

Conservatism should not be carried too far because it could lead to the rejection of an acceptable option in favour of a less desirable one (Shaeffer 1980). Accordingly, we have tried not to make BIOTRAC grossly conservative. Wherever possible, we have adopted conservative assumptions to balance the uncertainties. We have tried to develop realistic models for pathways that contribute significantly to consequences, and to employ the more conservative approaches to simplify and minimize the effort devoted to the less important pathways. The aim in applying the conservative philosophy has been to provide a margin of safety without unduly biasing the predicted environmental concentrations and doses. Furthermore, uncertainty regarding BIOTRAC predictions is not so much related to the accuracy of the predictions, but rather to the degree of overestimation of the true consequences. Conservatism and uncertainty can only be relaxed as new research data become available.

1.5.7 Variability and Uncertainty

A basic requirement in model development is the need to account for variability and uncertainty. This section briefly summarizes the main sources of variability and uncertainty, and how we account for them in BIOTRAC.

Variability and uncertainty are very important aspects in a mathematical model, such as BIOTRAC, because they relate directly to the accuracy and precision in model predictions, and therefore error. We have implemented BIOTRAC in SYVAC3 using Monte Carlo simulation techniques (Section 1.5.2), and this means that variability and uncertainty are accounted for in the distributed parameter values. Variability and uncertainty are not tracked separately, but are reflected in a combined manner in the distributions of the predicted consequences. These distributions have been statistically analyzed in the postclosure assessment (Goodwin et al. 1994). Combining the two is not unreasonable because they have a similar effect on the accuracy and precision of the predictions. In this sense, we use the term uncertainty to describe both variability and uncertainty. However, variability and uncertainty have been considered separately in constructing BIOTRAC, and in establishing its parameter values and distributions.

What are the major sources of uncertainty? O'Neill and Gardner (1979) recognized three main sources of error related to uncertainty in ecological models. These error sources are also relevant to three key aspects of BIOTRAC:

1. Model structure - Uncertainty arises from constructing a simple mathematical model to represent a complex system. The simple model may miss some important factors or misrepresent others. This can result in inaccurate predictions.

2. Model parameters - Uncertainty arises because parameter values and distributions may be derived from inaccurate or insufficient data. This can result in inaccurate or imprecise predictions.

3. Natural system variability - Most systems modelled are not static: they may change as a result of factors, such as ecological succession, climate change or even human activity. Natural system variability has both temporal and spatial components, and ignoring it can result in inaccurate predictions.

We recognize that there are many uncertainties in a model such as BIOTRAC, which is designed to make predictions far into the future. By necessity, our model must be very simple in comparison with the real world, but it is based on an extensive scenario analysis to make sure all the important features, events and processes are included (Section 1.5.1). Furthermore, complex models do not necessarily perform better than simple ones in making reliable predictions (O'Neill 1971, Crick and Simmonds 1984). We have attempted to capture the essential features of the real world in our model with the help of simplifying assumptions. Most of these are conservative to avoid underestimating the consequences (Section 1.5.6). Furthermore, we have validated three of the four BIOTRAC submodels as far as possible, and subjected BIOTRAC itself to a code comparison study (Chapter 11). These procedures help to establish confidence in the model, although the uncertainty resulting from the model structure cannot be quantified separately from the overall uncertainty in the model predictions.

BIOTRAC has many model parameters and distributions, the derivations of which are documented in great detail in the four submodel reports (Amiro 1992b, Bird et al. 1992, Sheppard M.I. 1992, Zach and Sheppard 1992) and are summarized in this report. The distribution of each parameter reflects the unique combination of variability and uncertainty in its values. A common set of guidelines was established for defining PDFs (Stephens et al. 1989). The quality and quantity of the data available for establishing parameter values and PDFs varied greatly, and in several instances analogies and expert opinion had to be relied on. We have carried out sensitivity analysis to guide our efforts on those parameters that most strongly influence model predictions (Chapter 10). Here too, conservatism has played an important role in dealing with uncertainty. Furthermore, validation of some of the submodels has also played an important role in addressing error in model parameters. The PDF for each parameter was chosen to account for all the uncertainties in the observed values of that parameter.

Natural systems tend to be variable in space and time. This is related to the fact that they are governed by diverse physical, chemical, and biological processes that interact with each other and do not remain static. Because of this, and the diverse array of organisms, there is constant change. Some of these changes are relatively short-term fluctuations, whereas others are long-term transitional processes (Section 3.1). This makes it difficult to predict the state of the natural system at any given instant. We have made BIOTRAC generic to deal effectively with natural system variability so that the model can describe sites anywhere on the Ontario portion of the Canadian Shield. As discussed in Section 3.1, the variability at any given site on the Canadian Shield over the next 10 000 a may well be equivalent to or less than the spatial variability we see today over the entire Canadian Shield. The parameter PDFs are intended to include values that could occur on the Canadian Shield today. Therefore, the PDFs also account for temporal variability. This philosophy or assumption only leaves transitional processes such as continental glaciation for separate assessment (Chapter 12). As in the case of the other two main sources of uncertainty, we have also attempted to allow for uncertainty from natural system variability through conservatism.

Our assessment approach does account for variability and uncertainty both directly and indirectly. An outcome of this is that BIOTRAC predictions span a broad range of possible outcomes and tend to be conservatively high. This is a reflection of our current level of understanding. It is anticipated that ongoing research by us, and others, will significantly reduce uncertainty (although not necessarily variability) and predicted consequences as new understanding is substituted for conservative assumptions. Similarly, if BIOTRAC is used to assess a particular site, the uncertainty in its predictions will be lower than when it is used to cover a range of possible sites across the Canadian Shield. This is discussed more thoroughly in Chapter 14.

1.6 REPORT OBJECTIVES AND OUTLINE

The main objectives of this report are to describe and justify the biosphere model developed for the postclosure assessment case study of the concept for disposal of Canada's nuclear fuel waste. We will show how the available experimental data and current theoretical understanding have been synthesized into a model capable of predicting nuclide concentrations throughout the biosphere, and doses to humans and other organisms. We will show that the approaches and the level of detail used are sufficient to represent the processes involved and that, where validation is not possible, the model and parameter values are conservative. Evidence for these claims will be drawn from the extensive body of research on contaminant transport in the biosphere (Figure 1-7), as published in the open literature, AECL documents, and the reports describing the four submodels of BIOTRAC (Amiro 1992b, Bird et al. 1992, Sheppard M.I. 1992, Zach and Sheppard 1992). Key aspects of the four submodels have also been published in the open literature.

Chapter 2 discusses the historical development of biosphere modelling and identifies the major transport processes and exposure pathways involved in predicting consequences. It presents the overall mathematical framework of BIOTRAC. The philosophy behind distributed parameter values and the guidelines used to derive the PDFs from experimental data are also described.

Chapter 3 discusses the way in which the environment may be expected to change over the course of the postclosure assessment period, and how such changes are accounted for in the model. We show that the effect of fluctuating processes on predicted consequences is implicitly accounted for through the use of distributed parameter values. We review the long-term transitional processes that could occur during the assessment period. Human activities and glaciation are identified as processes that could affect dose predictions. We outline the way in which human activities are accounted for in the assessment, deferring a discussion of glaciation to Chapter 12.

Chapter 4 describes how the biosphere model is coupled to the geosphere model. It provides a brief overview of the geosphere model and summarizes predictions of discharge zone locations and rates of nuclide discharge to the biosphere. Chapter 4 also indicates how the output from the geosphere model is used as input to the biosphere model to drive the surface water and soil submodels.

The four main BIOTRAC submodels (the surface water, soil, atmosphere, and food-chain and dose submodels) are presented in Chapters 5 through 8.

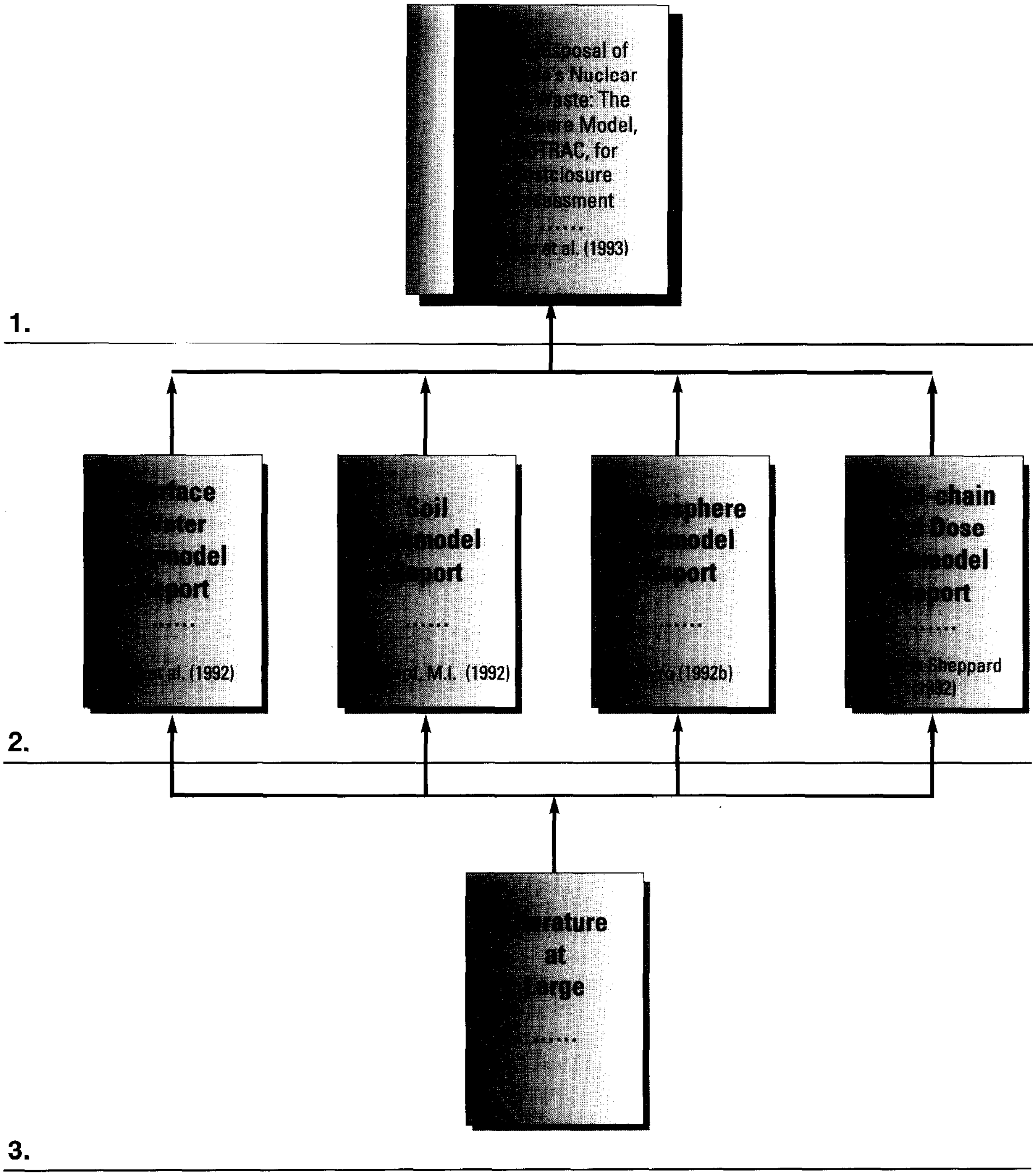

FIGURE 1-7: Schematic Representation of the Documentation of BIOTRAC. Level 1 constitutes the biosphere model report; Level 2 involves the surface water, soil, atmosphere, and food-chain and dose submodel reports; and Level 3 represents the literature at large, ranging from various AECL documents to journal publications.

The structure of each of these chapters is similar, and each draws heavily on its associated submodel report. The transport processes and exposure pathways are presented for each compartment from both descriptive and mathematical points of view. The parameters required to run the models are identified, and their PDFs are defined and justified. The way in which each submodel interfaces with the other submodels and with the geosphere model is described. Throughout these chapters we emphasize the reliance that has been placed on field data, experimental data and theoretical understanding in developing the models and the parameter values.

Chapter 9 ties the four submodels together to present an integrated view of BIOTRAC. It deals with issues and parameters common to all the submodels, and describes how the biosphere model is implemented within SYVAC3. It discusses the extent to which a nuclide mass balance is preserved in BIOTRAC. It also compares BIOTRAC with the biosphere model used to assess the preclosure phase case study of the waste disposal concept.

The sensitivity analysis of the biosphere model is the subject of Chapter 10. The methods used to perform the analysis are described, together with a discussion of the results. The parameters to which the model output is most sensitive are identified, as are the pathways and nuclides that contribute significantly to predicted concentrations and doses.

Chapter 11 discusses the validation of BIOTRAC. We emphasize the work done to measure model performance against experimental and field data, and discuss efforts regarding model testing, model intercomparisons and peer review. The aim of this discussion is to demonstrate that the models adequately represent the processes of nuclide transport through the biosphere.

The impacts of continental glaciation on nuclide transport through the biosphere are discussed in Chapter 12. We develop an approach that we believe is credible and manageable and that provides an adequate assessment of the effect of glaciation on predicted doses. We describe the changes made to the model and its parameter values to allow it to treat glaciation, and report human doses predicted for different stages of the glacial cycle.

Chapter 13 is dedicated to environmental protection. We put this topic into an ecological context and discuss in detail the approach we developed to help demonstrate environmental protection. We discuss the role of the biosphere model in our approach and provide information for evaluating model predictions.

Chapter 14 describes the changes required to make the biosphere model and its parameter values suitable for site-specific assessments. Although no location has yet been selected for the disposal facility, we must demonstrate the ability to characterize and assess specific locations before the concept of geological disposal can be accepted (Joint Statement 1981, AECB 1985). Accordingly, this chapter identifies the parameters whose site-specific values would differ from their generic ones, and indicates how the site-specific data would be obtained. Similarly, we identify pathways that may have to be treated differently in a site-specific assessment, and indicate the form that a site-specific model would take.

The text of the report concludes in Chapter 15 with a brief summary of the model, its main assumptions, and an evaluation of its suitability for assessing the concept for disposal of Canada's nuclear fuel waste.

There are several appendices to assist the reader. Appendix A summarizes the acronyms, names and abbreviations, and Appendix B lists the parameters and their related symbols alphabetically, together with their definitions and units. Appendix C includes a brief description of scenario analysis, and a list of the features, events and processes considered in developing BIOTRAC. A sample BIOTRAC calculation is presented in Appendix D, and Appendix E includes a short glossary. Finally, Appendix F contains a list of recent AECL references that have not been cited in any of the submodel reports or the biosphere model report. These references support our models and parameter values. They also give an indication of ongoing work on the behaviour, transport and effects of nuclides in the environment.

2. MODELLING THE BIOSPHERE

2.1 CHARACTERISTICS OF THE SHIELD BIOSPHERE

In Ontario, the Canadian Shield spans a range of 1350 km from north to south, and 1500 km from east to west, underlying about two thirds of the province (Figure 2-1). Environmental conditions across this vast area vary widely (Grondin et al. 1994). The characteristics of the present-day Canadian Shield biosphere are described briefly below to provide a picture of the physical, chemical and biological setting of a disposal facility. In the absence of human interference it is unlikely that this biosphere will undergo major changes before the next glacial advance, but there could be minor changes (Section 3.3).

The majority of the landforms on the Ontario portion of the Canadian Shield are of glacial origin (Chapman and Putnam 1966). Overburden depths exceed 2 m over 90% of the area, most of which is forested; only about 2% of the region is exposed bedrock. The relief is characteristically low, except in a narrow band near the Lake Superior and Georgian Bay shores. About 4% of the area is covered by surface water bodies.

In general, the Canadian Shield is poorly drained and contains many wetland areas, particularly in the north where up to 50% of the land area may be considered wetland (National Wetlands Working Group 1988). Most lakes are small, with a mean area of about 7 ha, and are located in drainage basins about 10 to 20 times their size (Minns 1984). Lake water is typically renewed at rates ranging from once in 10 a to ten times per year, driven by a runoff of 0.3 to 0.4 m water$\cdot$a^{-1}. The average mean depth of the lakes is about 5 m.

The geochemistry of Canadian Shield lakes is more uniform than their widespread locations would suggest. Biological productivity, nutrient levels, alkalinity and sediment load tend to be low. Suspended solids, which are approximately 50% organic, are deposited on the lake bottom at a typical rate of 0.2 kg dry sediment$\cdot$m$^{-2}\cdot$a^{-1}. The bottom sediments are thin near

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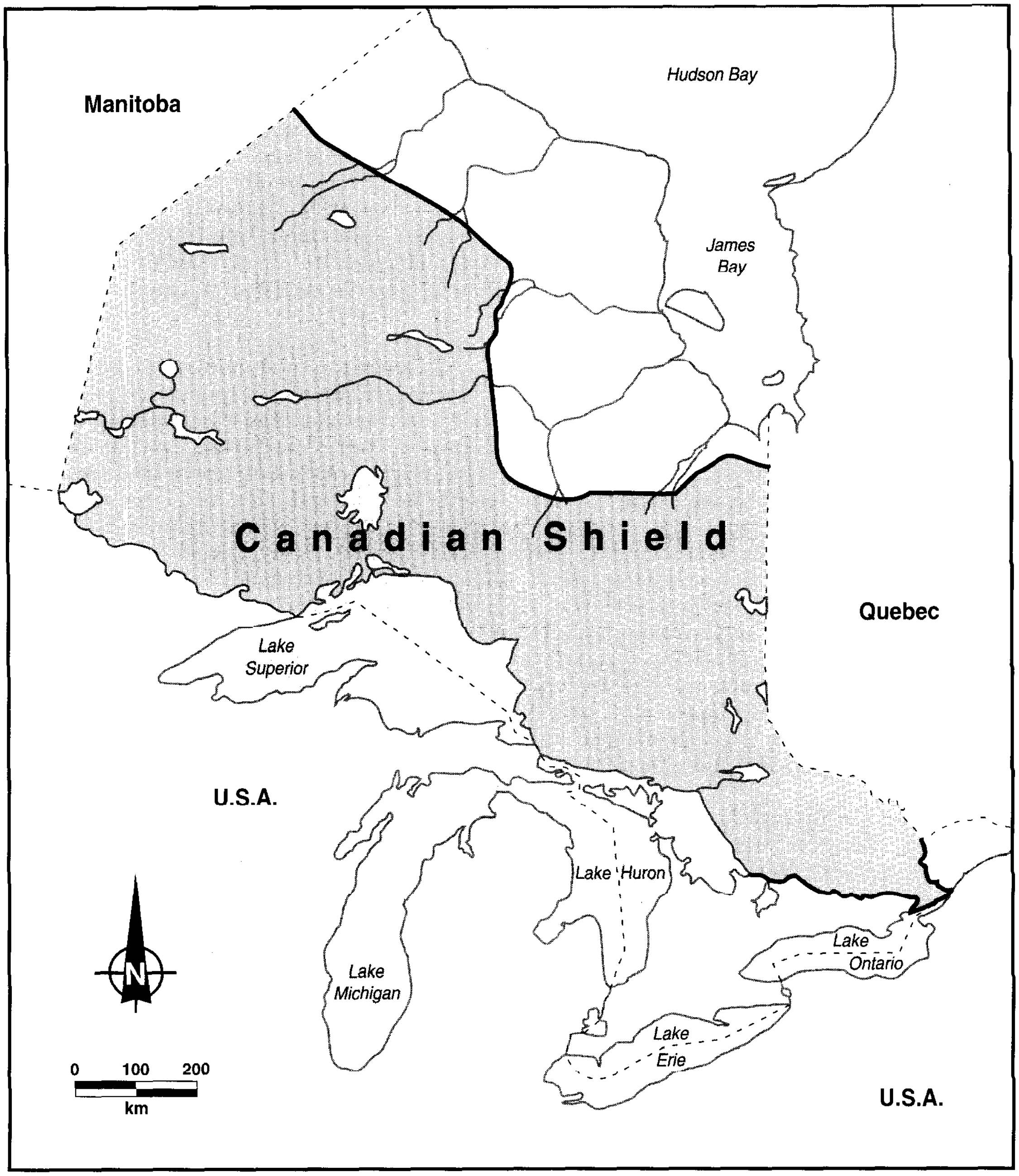

FIGURE 2-1: Extent of the Canadian Shield in Ontario on which the Biosphere Model is Based

the shore, and thicken to a depth of about 4 m in deeper parts of lakes. They reflect the composition of the material suspended in the water column.

The pronounced seasonal changes in climate on the Canadian Shield dramatically affect major lake processes, including rates of productivity, turnover and sedimentation. Lakes are frozen from four to seven months of the year and rivers about one-half month less. Shallow lakes may be well mixed throughout the year, whereas deeper lakes usually undergo complete mixing only in the spring and autumn, and are otherwise thermally stratified. On a geological time scale, the Canadian Shield has only recently (12 000 to 8 000 a ago) emerged from the last glaciation, and Canadian Shield lakes are in various stages of succession. Wetlands may be in transition to terrestrial areas, whereas deep oligotrophic (nutrient-poor) lakes are in the early stages of infilling.

The dominant soils of the Canadian Shield in Ontario are podzols, which are typically acidic, coarse-textured soils with pronounced layers, or horizons (Bentley 1979). A vegetation litter layer on the surface is underlain by an intensively leached white horizon of silica sand, and then by layers of iron-rich subsoil materials. However, the region is heterogeneous, and there are numerous other mineral soil textures present, as well as organic soils. If soil texture (particle size distribution) is used for classification, sandy soils cover about 55% of the Shield, clay soils 23%, silts 4% and organic soils including peat, 8%. In the remaining 10% of the area, soils are non-existent, or are too thin to categorize. Clays are largely restricted to the northern parts of the region, whereas silts are generally confined to the south. The proportion of organic soils increases in lowland areas. The depth to the water table ranges between 0 and 20 m, with typical values lying between 1 and 5 m, depending on the topography and soil texture. The soil profiles are relatively mature in the south, but are still undergoing development in the north.

The Canadian Shield in Ontario experiences a humid continental climate, which is characterized by extremes of temperature and sufficient precipitation to support agriculture (Fisheries and Environment Canada 1978). The south is generally warmer than the north. Mean July daily temperatures decrease from 20°C in the south to 15°C in the north; mean January daily temperatures decrease from -10°C to -20°C. The northwest, which receives a total of about 0.55 m water$\cdot$a^{-1} of precipitation, is drier than the southeast, which receives about 1.0 m water$\cdot$a^{-1}. The ratio of snowfall to rainfall increases from about 0.25 in the south to 0.50 in the north; the ground is snow-covered for about four months of the year in the south and six months of the year in the north. Evapotranspiration decreases from about 0.6 m water$\cdot$a^{-1} in the south to 0.3 m water$\cdot$a^{-1} in the north. Despite these general patterns, many locations show local anomalies, particularly near the Great Lakes, which tend to moderate temperatures and increase precipitation. Annual average wind speeds are typically 14 km$\cdot$h^{-1} at the standard 10 m observation height (Environment Canada 1982a). Prevailing wind directions depend strongly on local topography, but show a westerly component at most locations.

The weather on the Canadian Shield in Ontario is largely controlled by extensive air masses that sweep from west to east across the continent. A given set of atmospheric conditions can therefore be experienced at almost any site for at least a brief period of time, so that extreme climate conditions are similar across the region.

The Ontario portion of the Canadian Shield belongs to the Boreal Forest Region, except for the southern edge, which lies within the Great Lakes - St. Lawrence Forest Region (Rowe 1972). White spruce and black spruce are the most characteristic species of the boreal forest, but tamarack, balsam fir and jack pine are also common. Although the forest is primarily coniferous, there is a general admixture of broadleaved trees, such as white birch and trembling aspen. The Great Lakes - St. Lawrence region is a highly mixed forest characterized by eastern white pine, red pine, yellow birch and eastern hemlock. Other common broadleaved species are sugar maple, red maple, red oak and many other species. A broad array of shrubs and herbaceous plants is associated with both forest regions.

The Canadian Shield has a diverse fauna, and most species are widely distributed. Lakes and streams are inhabited by brook trout, lake trout, lake whitefish, white sucker, northern pike, smallmouth bass, yellow perch, walleye and numerous minnow species (Scott and Crossman 1973). Several species of salamanders, newts, toads, frogs, turtles, lizards and snakes occur in the region, particularly in the south (Cook et al. 1984). Many species of resident and migratory birds can be encountered. The common loon, common raven and the numerous ducks and wood warblers are especially characteristic (Godfrey 1966). Upland game birds include the spruce grouse, ruffed grouse and sharp-tailed grouse. Among the more characteristic mammals are moose, caribou, white-tailed deer and black bear (Banfield and Brooks 1974). The beaver is also common, as are many other fur bearers. In addition, there are many small species of rodents, bats and insectivores, and innumerable invertebrates.

The Canadian Shield in Ontario is only sparsely populated by humans, with most of the population concentrated around resource-industry centres in the southern regions. Southeast of North Bay, more than half the land area has population densities above 1 $p \cdot km^{-2}$. Further north and west, this density occurs on less than 20% of the land area. Forestry is the major land-based industry, supplying resources for pulp and paper products and construction materials. Mining of gold, silver, copper, zinc, nickel and other minerals is carried out at many widespread localities. Hunting, trapping and fishing, on both a commercial and sport basis, are practised at a fairly low level. Recreational land use is relatively intense near developed public parks, and wherever vehicle access is possible. Farming is practised near populated areas, with much of this on relatively small farms supplying local needs for milk, eggs, meat and vegetables. The length of the frost-free period largely dictates the type of crops grown, with local conditions of drainage, soil texture, slope, stoniness and historic land use playing a secondary role.

If and when a disposal vault has been decommissioned and closed in the postclosure phase, its physical presence would likely have no significant effect on any aspect of the Canadian Shield biosphere. Its greatest potential effect, other than from nuclide releases, would be to increase soil temperatures since the used fuel in the vault would be a heat source. However, calculations using conservative assumptions (Goodwin et al. 1994) have shown that temperature increases at the soil surface would be too small to have any effect on the biosphere, and are in fact less than the changes in temperature that are observed between various locations on the Shield.

Biosphere characteristics in each model simulation are set by sampling parameters such as lake area, soil type and precipitation. A slightly different biosphere is modelled in each simulation. Each biosphere describes a generic site typical of environments that are presently found on the Canadian Shield, or that could develop over the next 10 000 a. It is not necessary to model all biosphere compartments in detail; values are required only for those parameters that affect nuclide migration and dose to humans and other organisms. Since the critical group is usually assumed to practise agriculture, the local environment is assumed to have the characteristics of a farming area. Accordingly, we assume there is sufficient arable land for the garden and forage field (Section 1.5.4) among the rocks, trees and lakes of our Canadian Shield region at the vault discharge zone. The areas of these fields are calculated from the size of a typical household belonging to the critical group and from their food requirements (Section 9.1.1). Similarly, the areas of the woodlot and peat bog are calculated to provide the resources required to build and heat homes. For most purposes, the shapes of the fields and their locations relative to each other, to the various farm buildings, and to the surrounding physical features, need not be specified. Similarly, the location of the lake does not have to be defined precisely; we assume that the location of the farm within the discharge zone is such as to allow water supplies to be readily drawn from the discharge lake, or from a well drilled into the contaminated groundwater plume. The exception is in the atmosphere model (Section 7.3.1.1), where simple assumptions concerning the physical layout of the farm are required before air concentrations can be calculated.

A schematic representation of the Canadian Shield environment and of a farming settlement in the vicinity of the discharge zone is shown in Figure 2-2. Although the exposure situation is specifically defined for humans, it is equally suitable for assessing human and environmental impacts (Section 1.5.4).

2.2 HISTORICAL DEVELOPMENT OF BIOSPHERE MODELLING

Mathematical modelling of contaminant transport through the biosphere on a regional scale is a relatively young science. It began in the 1950s as the outgrowth of developments in two other areas: the advent of analog and digital computers, which made it feasible to solve the complex equations and to store and access the large amounts of data required to simulate contaminant behaviour in the biosphere; and the growing awareness that human activities can lead to a general, widespread and continuing degradation of the environment (Neely 1980). The contaminants of concern were originally pesticides and radioactive fallout from nuclear weapons testing. In subsequent years the list expanded considerably to include mercury, polychlorinated biphenyls (PCBs), sulphur dioxide and the nitrogen oxides responsible for acidic precipitation, and carbon dioxide, which contributes to greenhouse warming of the earth's climate (Section 3.3.1). Biosphere modelling received a further impetus in the early 1970s when first the United States, and then other countries, began to require formal environmental impact statements for major projects.

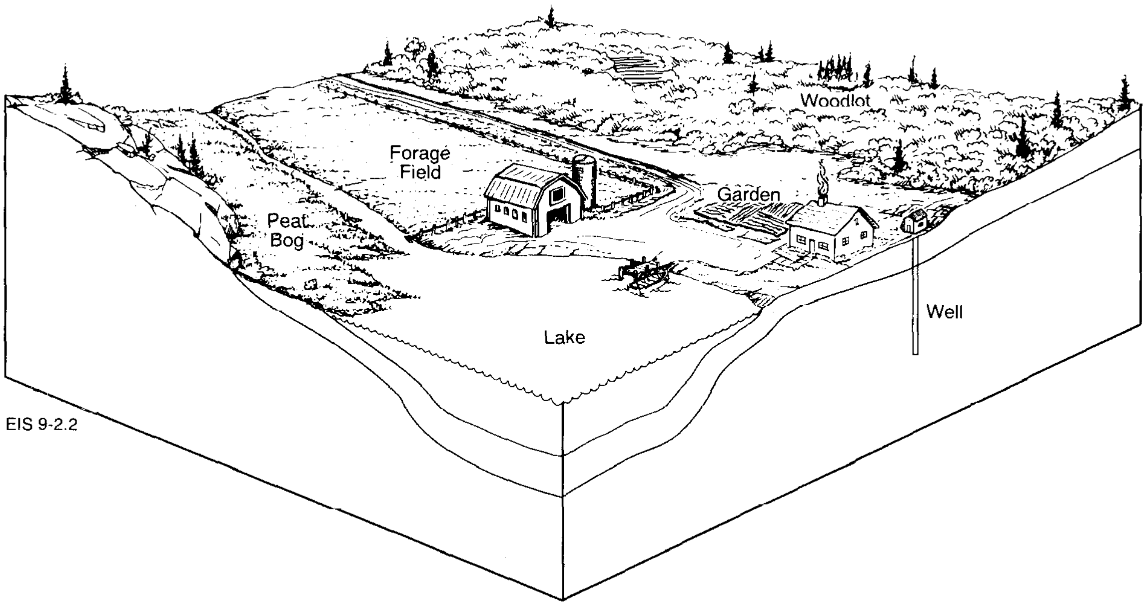

FIGURE 2-2: Generic Farm on the Canadian Shield Occupied by the Critical Group in BIOTRAC

Although biosphere modelling is a relatively new science, it draws extensively on the accumulated work of several mature disciplines. The four basic compartments of the biosphere (surface water, soil, atmosphere and food chain) have each been studied individually for over a century. The physical, chemical and biological processes that occur within each compartment are reasonably well understood, and our understanding of contaminant behaviour in each is only slightly less advanced (Whicker and Schultz 1982, Pasquill and Smith 1983, Coughtrey et al. 1985, Sibley and Mytermaere 1986, Eisenbud 1987, Santschi and Honeyman 1989). Early models of contaminant transport through the compartments were relatively simple and were solved analytically. Since the development of computers, more complex and sophisticated numerical models have also become available. The achievement of biosphere modelling has been to bring together the relevant aspects of these various disciplines to provide a coherent description of contaminant behaviour and impacts in the biosphere as a whole. Over the last three decades, biosphere models have evolved from simple, somewhat crude expressions to more sophisticated and realistic representations of environmental systems (HRI 1976, Frenkiel and Goodall 1978).

The development of biosphere models has been driven, in part, by the need to assess the impact of radionuclides in the environment. Three distinct types of radionuclide release can be recognized: fallout from nuclear weapons testing, operation of nuclear power plants and nuclear waste management. Routine monitoring and surveys of fallout radionuclides from the late 1940s to the 1960s and beyond were followed by laboratory and field studies of deposition and transfer through terrestrial food chains (Garner 1971, Whicker and Schultz 1982). Information collected in these studies was used to formulate the first models describing radionuclide deposition, environmental transfer and dose prediction. Much of this work focussed on ^{137}Cs, ^{131}I and ^{90}Sr, important fallout nuclides.

Radionuclide releases from operating nuclear power plants accompanied the initial growth of the nuclear power industry in the 1960s and 1970s. Radionuclides other than those from fallout became of interest, as did their behaviour and transport in the atmosphere and in surface waters. Many models were developed for assessing power plant releases (Hoffman et al. 1977). These led to the development of systems analysis methods (Auerbach 1984) for comprehensive regional models (Fletcher and Dotson 1971) that included all the important environmental transport processes, and both internal and external dose predictions. Regulatory agencies began to promulgate generally accepted assessment models for nuclear power plants (USNRC 1977, CSA 1987), and more and more effort began to be devoted to parameter value determination, model validation and uncertainty analysis.

During the late 1970s and early 1980s, efforts began to shift toward nuclear waste management assessments. Different radionuclides such as ^{14}C, ^{129}I, ^{99}Tc and the actinides became important in these analyses. Because many of the waste management options involve burial, the models started to treat radionuclide behaviour and transport in soils at the same level of detail as in the other environmental compartments. Although many of the modelling methodologies established earlier continued to apply, the waste management application presented unique challenges in terms of the underground location of the source, the large number of nuclides and pathways involved, and the very long time frame of the assessment.

In developing the BIOTRAC model, we have found solutions to these challenges largely through our own extensive research program, which started in 1978 (Iverson et al. 1982, Zach 1985a, Zach et al. 1987). We also learned and applied a great deal from ongoing waste management research programs in other countries, particularly Finland, Germany, Sweden, Switzerland and the United States. Furthermore, we benefitted from advances in fields such as ecology, environmental toxicology, environmental physics, environmental chemistry, analytical science and computing. In particular, we were among the first (Dormuth and Quick 1980) to adapt Monte Carlo simulation techniques (Raeside 1976) to an environmental assessment, thereby providing a systems variability framework for dealing with variability and uncertainty (Section 1.5.2).

It is important to be able to assess the consequences to the environment and humans of contaminants. This is relatively easy to do in the case of radionuclides. Information relating environmental concentrations to health effects has been available for some nuclides since the 1930s from studies of radiation workers. An important advance in dosimetry was made in 1976 when ICRP 26 (ICRP 1977) introduced a new system for calculating dose to man, in which all internal and external doses became fully additive, and were related to risk (Section 1.2.3). Further advances are now being made in ICRP 60 (ICRP 1991a).

The prediction of radiological doses to non-human biota is also important, but is not as well established as for humans. However, much background information is available and suitable methodologies can be developed, as shown in Section 13.3. Here too, further advances are being made, stimulated by the current high level of environmental awareness and data from such events as the Chernobyl nuclear accident.

In summary, although biosphere modelling is a relatively young science, it has developed quickly. It is founded on a number of advanced disciplines that have developed over a century of study. When applied to radionuclides in the environment, it can draw on a vast amount of information on transport and dosimetry accumulated over several decades in many countries. Research programs established in Canada (Joint Statement 1978) and elsewhere about fifteen years ago have succeeded in providing solutions to the special challenges raised by the waste management application.

2.3 RADIONUCLIDE TRANSPORT PROCESSES

Radionuclide behaviour in the biosphere is governed by a number of physical, chemical and biological processes that together determine the rate of radionuclide transport and accumulation in the various environmental compartments. We assume that radionuclide behaviour and transport is element-specific, and that all the isotopes of a given element, be they radioactive or not, behave identically. This is not an unreasonable assumption because behaviour and transport are mainly determined by chemical properties, which tend to be element-specific. The most important of the biosphere processes are identified and discussed briefly here. The processes that are included in the model (and the way in which they are simulated) are discussed in Chapters 4 to 8 and Chapter 13, which describe the geosphere/biosphere interface, the four main submodels, and the model for assessing environmental effects in detail. These chapters also list the processes relevant to the compartment in question that are not modelled, and provide justifications for their exclusion. An overall perspective on nuclide transport through the biosphere, as an introduction to the detailed discussions that follow, is presented here.

2.3.1 Transport Processes

Groundwater from the vault will likely discharge into a topographic low containing a surface water body such as a lake, a river, or a wetland (Bird et al. 1992). Any nuclides present will be carried upward through the compacted sediments of the water body by advection and diffusion, although some will be lost to the sediments through sorption (Figure 2-3). Nuclides may also reach the lake in water that has run over contaminated soil, or through deposition from a contaminated atmosphere. Once in the water column, the nuclides will be mixed and diluted, and some will be flushed out of the immediate water body into downstream parts of the drainage system away from the critical group. In predicting doses to the critical group, these nuclides need not be considered further because they are no longer accessible. The situation is similar for assessing environmental effects. Nuclides may be lost from the water column in a variety of other ways. Some may be suspended into the atmosphere through processes such as wave breaking or bubble bursting. Volatile nuclides may be lost to the atmosphere through gaseous evasion.

Some nuclides may be taken from the lake with water that is withdrawn for domestic use, or irrigation purposes. Others may be taken up by aquatic

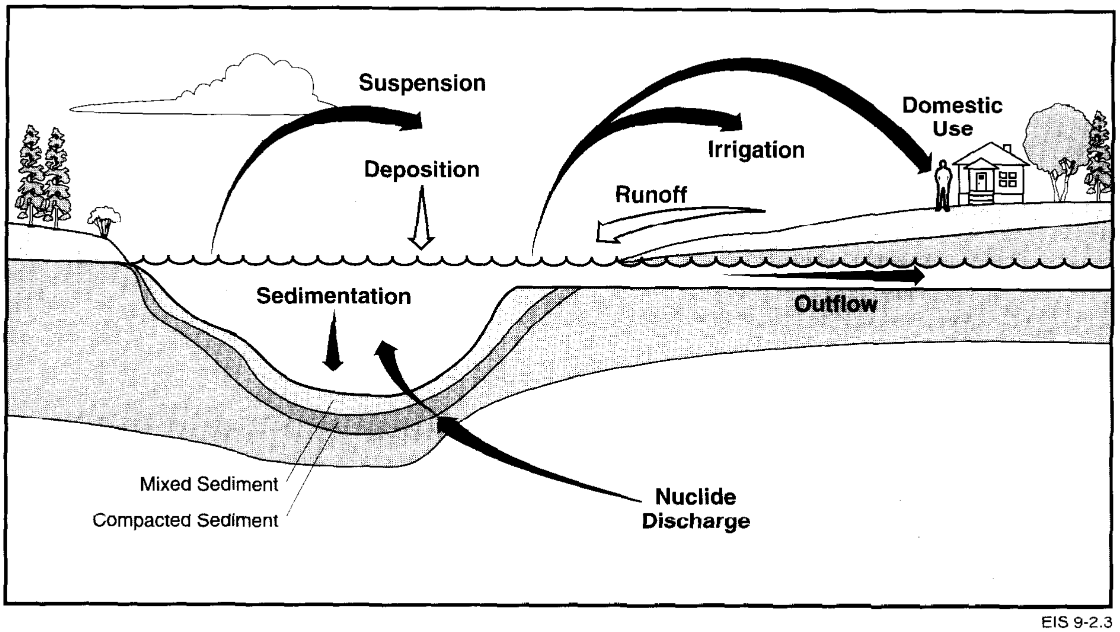

FIGURE 2-3: Transport Processes in the Surface Water Body. Closed arrows indicate explicit consideration in BIOTRAC, and open arrows indicate implicit consideration.

plants or animals, although these may be returned to the water when the organisms die. Still others may sorb onto suspended material in the water column and be deposited on the lake bottom as mixed sediment. These may be buried by subsequent sedimentation, or they may be resuspended back into the water column. Radioactive decay continuously removes radionuclides from the system, but some new nuclides are generated locally by the ingrowth of daughters, which may or may not be radioactive. The interplay of these and other less important processes determines concentrations in the surface water system at any time.

Although discharge is expected to occur mainly into water bodies, it is possible that some of the contaminated groundwater could come into contact with the bottom of the unsaturated soil zone (Figure 2-4, Sheppard M.I. 1992). The soil surface may also become contaminated by deposition from the atmosphere or by aerial irrigation with contaminated water. Once in the soil, nuclides are transported with the water flow, which is controlled by topography and meteorological conditions. During and immediately after precipitation events, nuclides are leached downward through the soil profile. When evapotranspiration occurs, they may be drawn upward by capillary rise. In either case, nuclides are exchanged continuously between the solid and aqueous phases of the soil through sorption mechanisms. Nuclides may be lost from the soil through a number of processes. They may drain out of the bottom of the profile and be carried away by groundwater flow. They may be suspended into the air by mechanisms such as wind erosion, or by gaseous evasion if they are volatile. They may be transported with surface runoff to local water bodies. Some nuclides may be taken up by

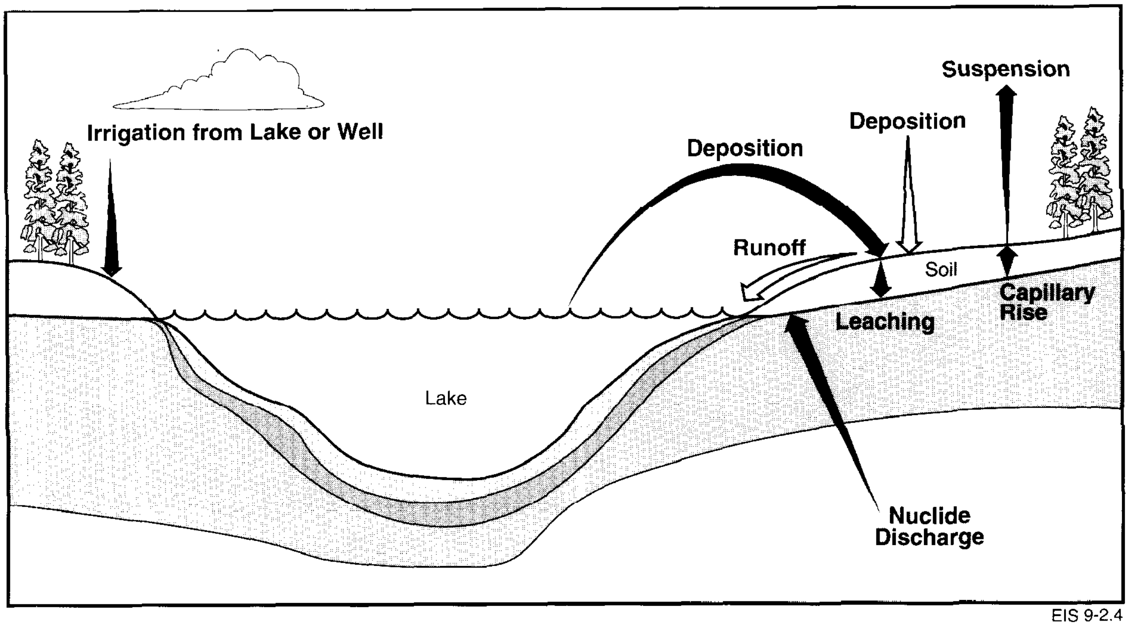

FIGURE 2-4: Transport Processes in the Soil. Closed arrows indicate explicit consideration in BIOTRAC, and open arrows indicate implicit consideration.

terrestrial plants, although a portion of these will return to the soil when the plant dies and materials are recycled. Together with radioactive decay and ingrowth, these are the main processes governing nuclide concentrations in soil.

Contaminants from an underground source can reach the atmosphere by suspension from the underlying surfaces: from water bodies through processes such as wave breaking and evasion; from the soil by processes such as wind erosion and gaseous evasion; and from vegetation via pollen release and fires (Figure 2-5, Amiro 1992b). The fires can arise through the annual burning of crop residues, periodic burning to clear new agricultural land, or the burning of wood or peat for energy. Once in the air, the nuclides undergo dispersion and deposition back to the surface. Deposition may occur through the scavenging of aerosols and gases during precipitation events, or by processes such as gravitational settling, particle impaction and surface sorption in the absence of precipitation. Because atmospheric processes occur on very short time scales, radioactive decay and buildup are generally not important in determining air concentrations.

The above processes govern atmospheric concentrations in outdoor air. Additional processes come into play when calculating indoor concentrations. Volatile nuclides, including radon (^{222}Rn), may diffuse from the soil into buildings, and nuclides can be released from contaminated water that is used in humidifiers or for showers. Indoor concentrations can build up to relatively high levels if air exchange rates with the outside are low.

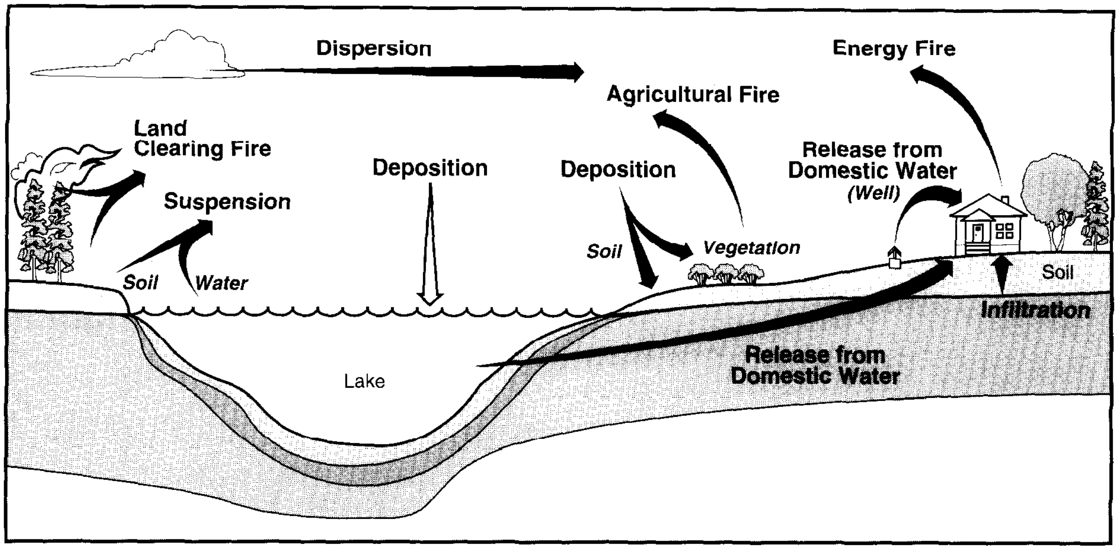

FIGURE 2-5: Transport Processes in the Atmosphere Considered in BIOTRAC. Closed arrows indicate explicit consideration in BIOTRAC, and open arrows indicate implicit consideration.

Plants and animals grown or raised in a contaminated environment may themselves become contaminated (Figure 2-6, Zach and Sheppard 1992). Nuclides reach plants from the soil by root uptake, from the air by deposition to leaf surfaces, and from water through leaf interception and deposition of irrigation water. Terrestrial animals accumulate nuclides through ingestion of contaminated fodder, water and also soil. Nuclides reach fish through ingestion of food and sediment, and through the osmotic exchange of tissue water and lake water. Nuclides taken up by all biological species are in large part returned to the physical environment in waste products or through the death or decay of the organism.

Human activities also contribute to nuclide transport. Domestic wells may provide a direct link between contaminated groundwater and the human and other food chains. Nuclides in water drawn from wells or lakes are transferred to soil and plants (through aerial irrigation) and to the immediate human environment in homes. Contaminated plants are removed from agricultural fields at harvest time. Activities such as ploughing, driving on gravel roads or burning contaminated wood for heat all release nuclides to the atmosphere. Contaminated sand or gravel may be used as building materials, lakes may be drained to create new agricultural fields, and lake sediments may be dredged and transported to fields for use as a soil conditioner. Many nuclides are recycled back to the environment with waste products, and from human use of water, crops and animals.

2.3.2 Microbial Effects

Many transport processes in the biosphere that we explicitly model are affected by the presence of microbes (bacteria, algae, fungi, protozoans and other small organisms). Microbes are ubiquitous in the biosphere, and

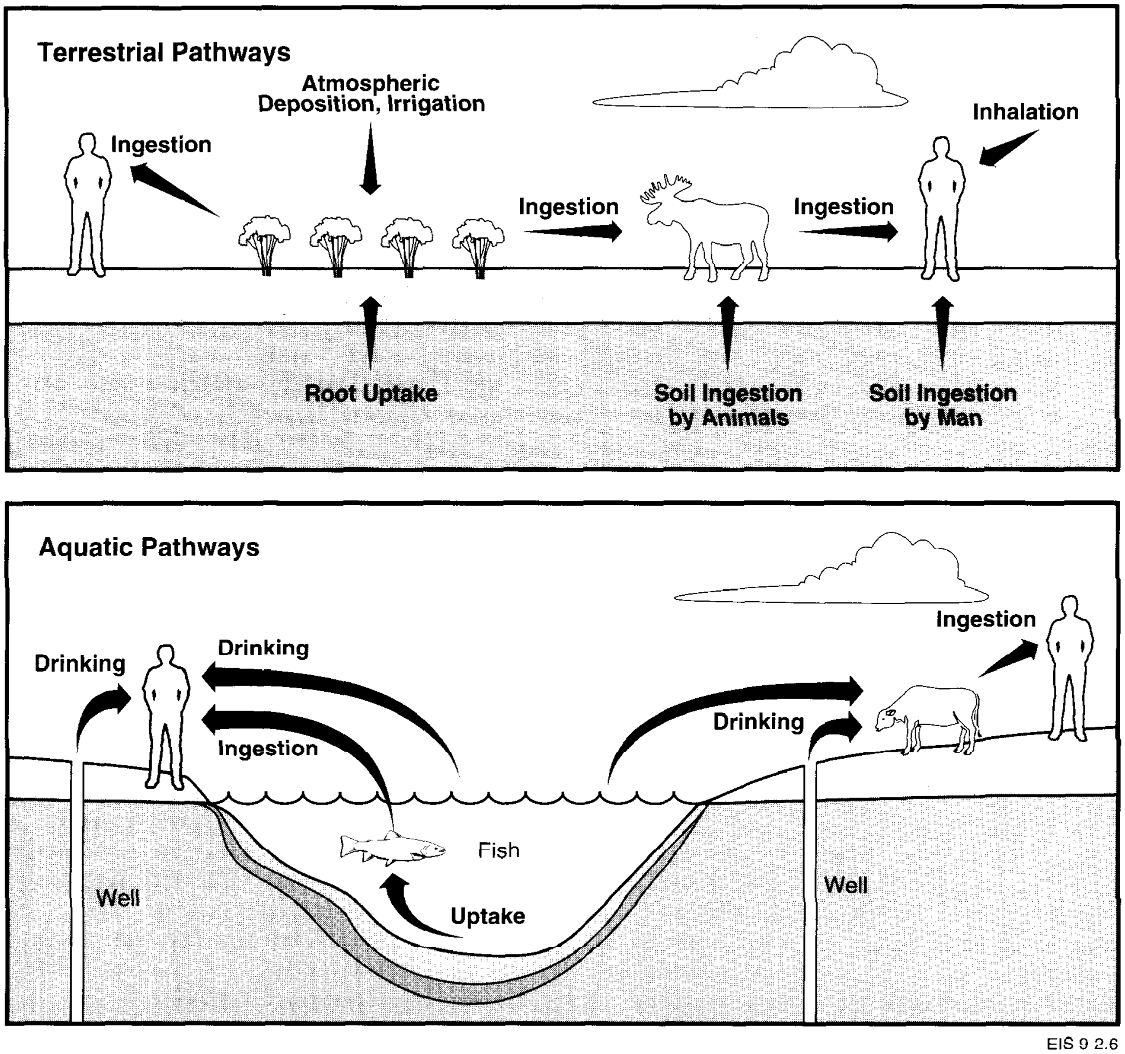

FIGURE 2-6: Transport Pathways in the Food Chain Considered in BIOTRAC

may influence the behaviour and transport of nuclides by changing their physical or chemical characteristics (Beijer and Jernelöv 1979, Loewen and Flett 1984). Microbes can change the oxidation state of some elements through metabolic activity, and thereby increase or decrease their mobility. They can create toxic mobile organic compounds through methylation in sediments and soils. Various microbially generated chelating agents can target otherwise immobile elements and form organic complexes with increased mobility and bioavailability. Microbes may change the absorption of some elements during root uptake by plants and digestion by herbivores. The exact extent by which microbes affect nuclide transport in the biosphere is variable. For example, root uptake of elements by many plants depends on mycorrhizal associations between the host plant and soil-living fungi. The effects of microbes on the behaviour and transport of nuclides in surface waters, soils and food chains are discussed in more detail by Bird et al. (1992), Sheppard M.I. (1992) and Zach and Sheppard (1992).

Virtually all field and laboratory studies investigating contaminant transfer in the biosphere (including those conducted specifically in support of BIOTRAC) have been conducted under natural conditions in which microbes were present and functional. Microbial effects on the behaviour and transport of nuclides have therefore been implicitly taken into account in BIOTRAC and its parameter values.

2.3.3 Nuclide Mass Balance

An exact nuclide mass balance is not maintained in BIOTRAC. Instead, mass is conserved internally within each environmental compartment (the surface water body, the soil and the atmosphere), but not when nuclides are transferred between compartments. In general, in the model the inventories of the donor compartments are not depleted when nuclides migrate to a new compartment. For instance, surface water inventories are not reduced when water is drawn from the lake for domestic use or irrigation; soil inventories are usually not depleted when nuclides are lost to the atmosphere through suspension; and air inventories are not reduced when nuclides are lost through deposition to underlying surfaces.

Although assumptions such as these result in an apparent generation of mass and radioactivity, they allow complex processes such as runoff, recycling, and atmospheric suspension and deposition to be treated very simply. Furthermore, they allow concentrations in both donor and receptor compartments to be calculated conservatively, which is reflected in higher predicted consequences. If a mass balance is enforced, a transfer coefficient that produced a conservative result in one compartment would underestimate the concentration in the other. The multitude of exposure pathways makes it difficult to identify the compartment that contributes most to the dose and for which concentrations should not be underestimated. In all cases where source inventories are not depleted, the nuclide flux out of the source compartment is small compared with the total inventory in the compartment, and the mass generated in the model is not large. A fuller discussion of these concepts as they apply to each submodel is given in Chapters 5 to 8.

A second assumption in the biosphere model also results in apparent mass generation. In each simulation, we assume that contaminated groundwater discharges partly to a lake and partly to a terrestrial soil area (Section 4.2). However, concentrations in the lake are calculated assuming that the entire discharge occurs to the lake; some of these nuclides therefore represent a duplication of mass. As discussed in Section 5.2, this assumption allows the runoff of nuclides from the soil to the lake to be modelled very simply. Near steady state, the nuclide flux retained in the soil is a very small fraction of the flux out of the geosphere, so the duplication of mass is not large.

Where mass conservation is violated in BIOTRAC, mass is created, never destroyed. The same is true for radioactivity. By foregoing an exact mass balance, we can simplify the model greatly, and simultaneously add a degree of conservatism to it. In Section 9.3 we calculate the amount of mass generated through these assumptions, and estimate the effect on the predicted nuclide concentrations and doses.

2.4 EXPOSURE PATHWAYS

The mathematical expression of the transport processes described above results in the prediction of nuclide concentrations in the various environmental compartments. Humans living in the contaminated environment receive a radiological dose via a multitude of exposure pathways. BIOTRAC treats all the pathways that are commonly recognized as contributing to human exposure (Zach and Sheppard 1992). Internal doses are caused by radionuclides taken into the body through ingestion or inhalation (Figure 2-7). As noted in Section 1.2.3, internal exposure results in a dose commitment to the body. Ingestion doses arise from the daily intake of contaminated plants, animal products, fish, water and soil. The water source may be either a well drilled into the contaminated groundwater plume, or the lake in the discharge zone. Inhalation doses arise from breathing contaminated air, both indoors and out.

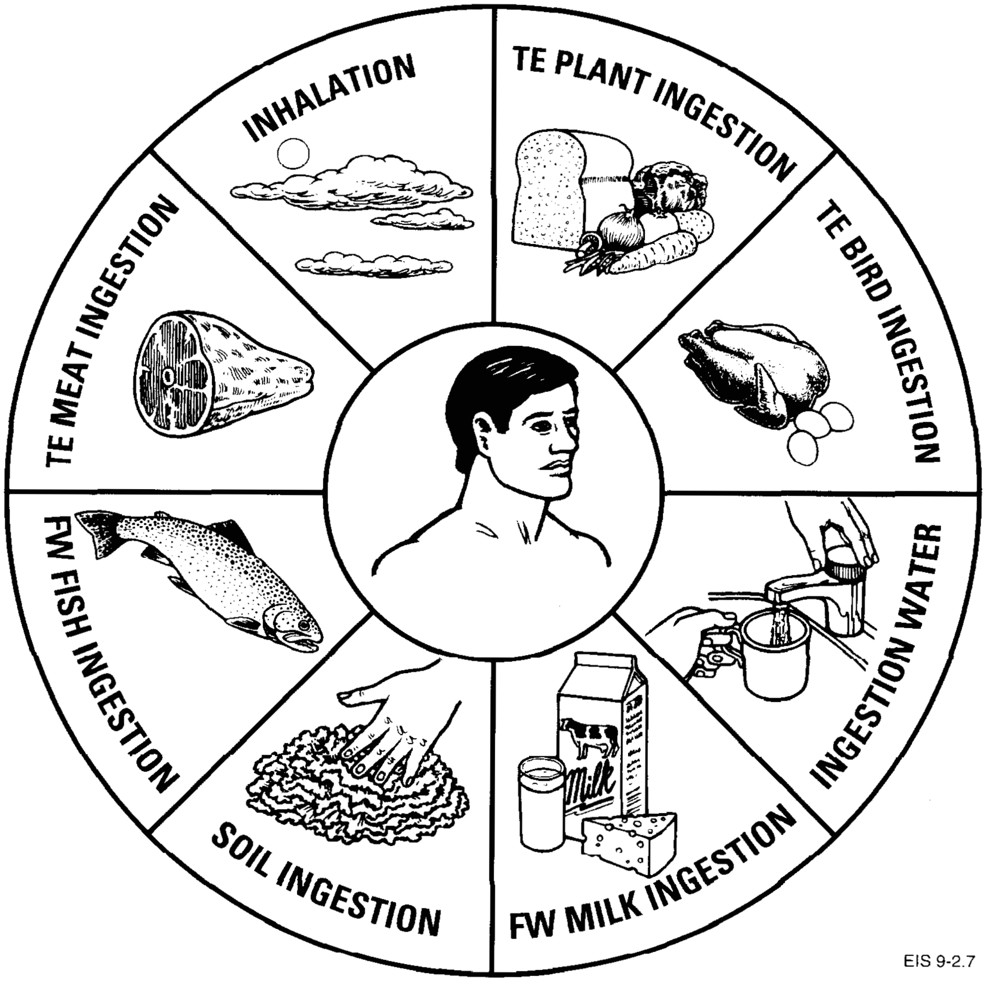

FIGURE 2-7: Internal Exposure Pathways for Man Considered in BIOTRAC. TE refers to terrestrial food types and FW refers to freshwater food types.

External doses arise from radiation fields given off by contaminated parts of the environment (Figure 2-8). They involve no radionuclide intake or dose commitment; rather, the magnitude of the dose depends on the duration of the exposure. Humans may receive an external dose from exposure to contaminated ground or by living in houses built of contaminated materials. External doses may also arise from immersion in air, either indoors or out, and from immersion in water while swimming or bathing.

Biota other than humans may also be exposed to radiation through internal and external pathways (Chapter 13). Furthermore, the principles of food-chain transfer for humans and animals are very similar.

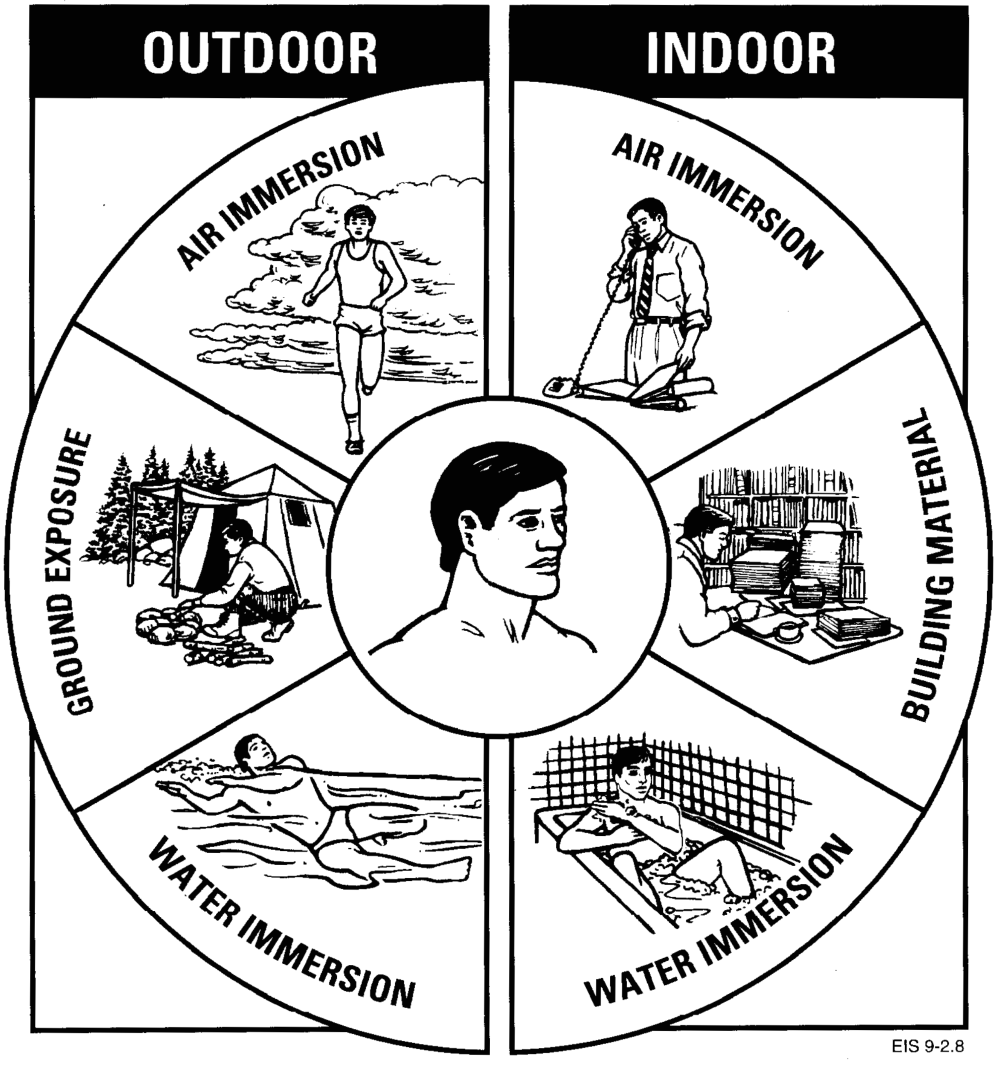

FIGURE 2-8: External Exposure Pathways for Man Considered in BIOTRAC

The internal and external exposure pathways included in BIOTRAC are the same for naturally occurring radionuclides (e.g., ^{14}C, ^{129}I and ^{40}K) and radionuclides that may be released from the vault.

2.5 SPECIAL RADIONUCLIDES

Environmental concentrations for the vast majority of nuclides in the vault are calculated by modelling their movement through the biosphere according to the transport processes discussed in Section 2.3. Similarly, doses to humans for most of the radionuclides are calculated taking into account contributions from all of the exposure pathways discussed in Section 2.4. In general, each nuclide could be in any number of possible chemical forms. We do not specify the chemical species, but adopt parameter values that are representative of the most mobile form of each nuclide in the environment.

A few radionuclides exhibit special properties that require alternative approaches to transport modelling and dose calculation for humans. In this section we identify these radionuclides, and outline the way in which they are treated in BIOTRAC.

2.5.1 Tritium

Hydrogen is ubiquitous in the biosphere because of its presence in water. As part of biological and water cycles, hydrogen is extremely mobile, and moves quickly from one compartment to another. When the radioactive form of this element (tritium, or ^{3}H) is introduced into the biosphere, it tends to become associated with water, and to share in water's mobility (NCRP 1979). The stable and active forms of hydrogen are assumed to become thoroughly mixed, so that the specific activity of tritium (the ratio of the tritium concentration to the total concentration of hydrogen in all forms) becomes uniform throughout the biosphere. If the specific activity is known in any one compartment, the radionuclide concentration can be deduced in any other compartment because the concentrations of the stable forms are known throughout the biosphere. This method of calculation is known as the specific activity approach. When applicable, it provides a very effective means of estimating concentrations without the need for detailed transport modelling.

To apply the specific-activity model to tritium, we assume that specific activities in soil, air and the food chain are equal to the specific activity in the lake or in well water. This is a very conservative assumption since uncontaminated water flowing into these compartments will tend to dilute the tritium concentration. Thus, the tritium specific activity will not be higher anywhere else in the biosphere than in the lake or well water. The specific activity in lake and well water is found from the tritium concentrations in these compartments, which are calculated with the methods used for all the other nuclides discussed in Chapters 4 and 5. Since the concentration of hydrogen atoms in humans is well known (ICRP 1975), the tritium concentration in humans can be deduced directly without considering food-chain transfer. All internal doses can then be calculated using a single dose conversion factor that takes into account food and water ingestion, inhalation and skin absorption (Zach and Sheppard 1992). External doses are not calculated for tritium since they are very low compared to the internal dose (Holford 1989).

2.5.2 Carbon-14

Carbon-14 is also very mobile in the environment, and has traditionally been treated using a specific activity approach in models for assessing nuclear power plant emissions (NCRP 1985, CSA 1987). This is a reasonable approach for that situation. When the release occurs directly to the atmosphere, a constant specific activity can be expected locally in air, plants and humans. Since ingestion of plants following air deposition is the main exposure pathway, the value of the specific activity in the other biosphere compartments is immaterial.

In contrast, when the source is underground, as in this assessment, many pathways contribute to the total exposure. The specific activity in soils, air and biota is likely to be very much less than that in lake or well water because of the very large and mobile carbon pools in the former compartments. Because the specific activity is not uniform across the entire biosphere, a specific-activity model is not appropriate. BIOTRAC calculates the ^{14}C concentrations using the same transport model as for the other nuclides. However, we limit the total internal ^{14}C dose to humans on the basis of its predicted specific activity in the geosphere. Groundwaters on the Canadian Shield contain small amounts of stable carbon, which would move in concert with any radioactive carbon that migrated from the geosphere to humans. Since it is reasonable to assume that the ratio of active to stable carbon cannot be higher in the biosphere than in the geosphere, the specific activity in the groundwater imposes an upper limit to the specific activity in the human body. The ratio would be reduced in the biosphere because of ^{14}C dilution by the presence of large amounts of stable carbon. This specific-activity limit has been taken into account in calculating man's internal doses arising from ^{14}C (Zach and Sheppard 1992). This limit has not been implemented for other biota.

2.5.3 Iodine-129

Iodine entering the human body accumulates almost exclusively in the thyroid gland (NCRP 1983). The internal dose to humans from ^{129}I can therefore be calculated by considering the thyroid alone (ICRP 1979). Iodine is an essential element (Guthrie 1983), and is metabolically regulated so that its level in the thyroid stays within narrow limits regardless of intake. Once the thyroid becomes saturated, excess iodine is rapidly excreted from the body.

Stable iodine, ^{127}I, is present in the environment both in the biosphere and the geosphere. Man's intake of iodine would therefore involve both active and stable forms. Assuming that the two forms are handled identically by the body, the thyroid cannot become saturated with ^{129}I. We have taken this into account in the iodine dosimetry model for man by implementing a limited specific-activity model in which the ratio of active to stable iodine in the thyroid is set equal to the ratio at intake to the body (Zach and Sheppard 1992). The intake of stable ^{127}I is set to a low value; the intake of active ^{129}I is calculated using the ordinary transport model, so that the iodine model is identical to that of most other nuclides up to the point of actual dose calculation. Using a low intake of stable ^{127}I is conservative because it increases the specific activity of ^{129}I and so increases the dose.

Since Canadian Shield groundwaters contain appreciable amounts of stable iodine, a limit similar to that placed on ^{14}C is applied to the internal ^{129}I dose based on the predicted specific activity of ^{129}I in the geosphere (Zach and Sheppard 1992). As in the case of ^{14}C, ^{129}I would be diluted in the biosphere by the presence of large amounts of stable iodine, and so use of this limit in dose prediction is conservative. This limit has not been implemented for non-human biota.

2.5.4 Short-Lived Radionuclides

Radionuclide concentrations in any biosphere compartment are controlled by two processes: direct transfer into and out of the compartment, and radioactive decay and daughter buildup within the compartment. For example, a radionuclide and its radioactive daughter may be taken up directly by a plant from soil. While in the growing plant, the decay of the precursor may further contribute to the concentration of the daughter (Ibrahim and Whicker 1988). Transfers between compartments are handled by the methods described in Chapters 5 to 8. The treatment of decay and ingrowth depends on the half-life of the daughter radionuclide (Zach and Sheppard 1992) and on the biosphere compartment under consideration.

As noted in Section 1.2.1, some radionuclides are assumed to be in secular equilibrium in the geosphere. These radionuclides will not necessarily remain in equilibrium once they reach the biosphere. Secular equilibrium is maintained from one compartment to another only if the precursor and daughter are both transferred between compartments at the same rate. This may not be the case because the precursor and daughter invariably represent different elements, and transfer is often element-specific. If the transfer rates are similar, an approximate secular equilibrium can be achieved provided the decay rate of the daughter is very much greater than its rates of transfer into and out of the compartment.

Very short-lived daughters (those with half-lives less than one day) are assumed to be in secular equilibrium with their precursors throughout the biosphere. Decay rates for these radionuclides are very much greater than transfer rates between any of the biosphere compartments. Furthermore, residence times in the soil, surface waters and food chain are all much longer than one day, and provide sufficient time for secular equilibrium to be achieved. The time required for atmospheric processes is generally less than one day; however, transfer rates to or from the atmosphere are not radionuclide-dependent for short-lived daughters, so secular equilibrium will be maintained as long as it exists in the adjoining compartments. Accordingly, the assumption of secular equilibrium is justified for these radionuclides throughout the biosphere, and there is no need to model them individually. Instead, their contributions to the dose to man are accounted for through their precursors (Zach and Sheppard 1992). For internal exposure, they are considered through the DCFs, which include the effects of daughters produced in vivo. For external exposure, their DCFs are added to those of the precursor. The 28 radionuclides with half-lives less than one day are listed in Table 2-1.

All radionuclides with half-lives greater than one day are tracked explicitly through the biosphere. Those in secular equilibrium in the geosphere are separated from their precursors as soon as they enter the biosphere,

TABLE 2-1

SHORT-LIVED RADIONUCLIDES CONSIDERED IN BIOTRAC

Radionuclides with Half-Lives Less than 1 d	Radionuclides with Half-Lives Between 1 d and 20 a
^{228}Ac	^{225}Ac (^{225}Ra)*
^{217}At	^{210}Bi (^{210}Pb)
^{211}Bi	^{93m}Nb (^{93}Mo)
^{212}Bi	^{93m}Nb (^{93}Zr)
^{213}Bi	^{32}P (^{32}Si)
^{214}Bi	^{233}Pa (^{237}Np)
^{221}Fr	^{210}Po (^{210}Bi)
^{223}Fr	^{223}Ra (^{227}Th)
^{234}Pa	^{224}Ra (^{228}Th)
^{234m}Pa	^{225}Ra (^{229}Th)
^{209}Pb	^{228}Ra (^{232}Th)
^{211}Pb	^{126}Sb (^{126}Sn)
^{212}Pb	^{182}Ta (^{182}Hf)
^{214}Pb	^{125m}Te (^{125}Sb)
^{211}Po	^{227}Th (^{227}Ac)
^{212}Po	^{228}Th (^{228}Ra)
^{213}Po	^{228}Th (^{232}U)
^{214}Po	^{231}Th (^{235}U)
^{215}Po	^{234}Th (^{238}U)
^{216}Po	^{90}Y (^{90}Sr)
^{218}Po	
^{219}Rn	
^{220}Rn	
^{126m}Sb	
^{206}Tl**	
^{207}Tl	
^{208}Tl	
^{209}Tl	

* Precursors shown in brackets.

** Not considered in the postclosure assessment because the parent, ^{210m}Bi, has a very low vault inventory.

and are treated individually thereafter (Section 4.4). Decay and ingrowth are handled rigorously in the soil and surface water submodels, which are time-dependent. Because time scales in the atmosphere are on the order of minutes, decay and buildup are insignificant for radionuclides with half-lives greater than one day, and are not considered. On the other hand, time scales in the food chain are on the order of months. Ingrowth can therefore be neglected in the food-chain submodel for daughters with half-lives greater than about 20 a. Ingrowth is more important for daughter radionuclides with half-lives between 1 d and 20 a, but cannot be modelled rigorously because the food-chain submodel is not time-dependent. Instead, ingrowth is approximated by assuming secular equilibrium of the daughters with their precursors. The total radionuclide concentration in the various compartments of the food chain is found by adding the contribution arising from ingrowth calculated in this manner to the concentration arising from direct transport. The 20 radionuclides treated in this way are listed in Table 2-1.

Calculating ingrowth by secular equilibrium is an approximation. There is insufficient time for secular equilibrium to be achieved for the longer-lived daughters in most of the food-chain compartments. However, the activity of the daughter resulting from decay of the precursor is a maximum at secular equilibrium, so this approach is conservative. Zach and Sheppard (1992) show that this approach is also reasonable by comparing secular equilibrium estimates of ingrowth with more detailed calculations.

2.5.5 Noble Gas Radionuclides

Because the noble gases ^{39}Ar, ^{81}Kr, ^{85}Kr and radon are inert, they can be treated very simply in BIOTRAC. Argon and krypton move rapidly through the biosphere, independently of the food chain and without accumulating. They do not cause an appreciable internal dose to humans because they are not retained in the body. Also, the air immersion dose is dominant for these radionuclides (ICRP 1979). Accordingly, only air concentrations are needed to calculate the total dose to man. These were estimated conservatively by assuming that the argon and krypton fluxes to the atmosphere equal the predicted fluxes out of the geosphere, without any losses to other compartments, so that all the geosphere releases are considered in the dose calculations (Section 7.3.2.2).

Radon is also inert, but has such a short half-life (Table 1-1) that it essentially moves in concert with its precursor, ^{226}Ra. Radon decays into particulate daughters, which can impart a significant inhalation dose; ingestion doses are very small in comparison and may be ignored (UNSCEAR 1982). External doses arise through all external pathways, but adequate data exist to evaluate air immersion only. Radon doses from the other three external pathways (water immersion, groundshine and exposure to contaminated building materials) are accounted for through the precursor radionuclide, ^{226}Ra (Section 2.5.4). The indoor and outdoor air concentrations needed for the inhalation and air immersion doses are calculated from the radon fluxes to the atmosphere from soils and surface waters. These fluxes are estimated from the ^{226}Ra concentration in soil and the radon concentration in water, which are predicted with the usual transport models.

2.5.6 Other Gaseous Radionuclides

Amiro (1985) reviewed the vault inventory and identified ^{14}C, ^{79}Se and ^{129}I as the only nuclides, apart from tritium and the noble gases, that could be in volatile form in the biosphere. Carbon-14, ^{79}Se and ^{129}I are treated in the same way as all the non-volatile nuclides in BIOTRAC, except that they are allowed to escape from the soil (Section 6.5.4), and some of them from the lake (Section 5.3.4), through gaseous evasion. The outgassed material is taken into account when air concentrations are calculated.

2.6 MATHEMATICAL FRAMEWORK

Transport processes in the atmosphere and in the food chain occur very rapidly, with time scales on the order of minutes to months. The transient aspects of these processes are generally not important when annual average doses are calculated. Accordingly, the atmosphere and food-chain submodels are steady-state models; concentrations in air, plants, animals and humans are assumed to adjust instantaneously to changes in concentration in the donor soil and surface water compartments. Expressed mathematically, the atmosphere and food-chain concentrations are directly proportional to the soil and water concentrations, and their evaluation is straightforward. This is also true for the model for predicting doses to non-human biota, which is similar to the food-chain and dose submodel (Chapter 13).

On the other hand, transport processes in the soil and lake occur slowly. Soil processes have time scales on the order of hundreds or thousands of years; processes in surface waters and sediments are more rapid, but still occupy tens of years. Accordingly, concentrations in these compartments change slowly with time and are best calculated using time-dependent models.

The models used to describe nuclide behaviour and transport in the vault and geosphere are also time-dependent. For reasons of consistency and efficiency, a common solution method has been implemented in SYVAC3 to solve all time-dependent systems. This method involves the convolution of an impulse response function with the rate of nuclide input to the compartment (Dormuth and Quick 1980). The impulse response function describes the model output (nuclide mass in the compartment, nuclide concentration or nuclide flow rate) as a function of time following a unit input of the nuclide to the compartment at some previous point in time. The model output at any time is found by summing the contributions from all previous inputs through the use of a convolution integral.

We illustrate these ideas for a model whose output is the total nuclide mass, M (mol), in an arbitrary volume. The mathematical expression for M at time t is

$$M(t) = \int_0^t I(t') \cdot RF(t,t') \, dt' \quad . \qquad (2.1)$$

Here $I(t')$ is the time-dependent rate at which the nuclide enters the volume (e.g., $mol \cdot a^{-1}$). $RF(t,t')$ is the impulse response function, which defines the fraction of nuclide remaining in the volume at time t following a unit impulse input at time t' (where $t' \leq t$). The convolution integral defined in Equation (2.1) provides the total mass in the volume by considering the input function I as an infinite sequence of impulse inputs, and adding up the contribution from each impulse.

Equation (2.1) can be evaluated once I and RF have been specified. RF can be deduced quite simply for a particular class of models called compartment models. A compartment model describes a system that exhibits no spatial dependence in any of its properties, or in the concentration of nuclides introduced into it. The loss of nuclides from such a system often occurs at a rate that is proportional to the total mass of the nuclide in the compartment. Furthermore, the concentrations of intermediate members of radioactive decay chains can be found by considering only the member itself and its immediate precursor. (For models with spatial dependence, the concentration of the intermediate member depends on all of its precursors.) Both the soil and surface water models have been expressed as compartment models.

Compartment models are generally expressed mathematically through the mass balance equation

$$\frac{dM(t)}{dt} = I(t) - \beta \cdot M(t) \quad . \tag{2.2}$$

Here, β is the fractional transfer rate resulting from all loss mechanisms, including radioactive decay. $I(t)$ is the rate at which nuclides are introduced into the system, and may include the ingrowth of the nuclide under consideration as a result of the decay of its precursor. If we take $M(0) = 0$ as the initial condition, Equation (2.2) can be solved by the method of Laplace transforms to give

$$M(t) = \int_0^t I(t') \exp[-\beta \cdot (t - t')] \, dt' \quad . \tag{2.3}$$

Comparing Equations (2.1) and (2.3), we conclude that the impulse response function for a compartment model is

$$RF(t,t') = \exp[-\beta \cdot (t - t')] \quad . \tag{2.4}$$

In this case, note that RF is a function of $(t - t')$ only, and not of t and t' individually.

Impulse response functions for the surface water and soil models are derived in detail in Chapters 4, 5 and 6. These are used in conjunction with the input $I(t')$ computed by the geosphere model to calculate concentrations in lake water, sediment and soil.

The simulation of radionuclide transport through the vault, geosphere and biosphere is directed by the executive code, SYVAC3 (Goodwin et al. 1987b). Time series are calculated for each time-dependent model variable in turn. The order in which the variables are treated is carefully chosen to ensure that the information required at each point in the model is available from previous calculations. Radionuclide chains are treated member by member, starting with the precursor and working through the daughters. Variables associated with the vault are handled first, followed by those pertaining to the geosphere and biosphere. The time series for a given variable is generated from values calculated at points in time that depend on the rate at which the variable changes with time. Where the variable shows only a slow change, the time points are far apart; where the variable changes quickly, the time points are close together to ensure that all important behaviour is captured. Operations such as addition, multiplication and convolution of time series are directed by SYVAC3 as needed. The value of any variable at any time is available through interpolation in the time series.

2.6.1 Notation

For convenience, a symbol convention has been adopted to aid in the mathematical presentation of BIOTRAC. Concentrations are expressed as

$(C^i_j)_k$

where C is the concentration

i is the nuclide

j is the environmental compartment (surface water, soil, air, etc.) and

k is the pathway.

This convention has also been applied to some flux terms and transfer coefficients. In some instances, subscripts, superscripts and other notations have been suppressed for the sake of simplicity. All the mathematical symbols are listed in Appendix B, together with their definitions and units.

2.7 TREATMENT OF PARAMETERS

2.7.1 Theoretical Considerations

As noted in Section 1.5.2, the performance assessment of the disposal concept is based on a systems variability analysis approach. The model output is a statistical expression of the consequences predicted in a large number of individual simulations. The state of the system in each simulation is defined by the parameter values sampled at the beginning of the simulation. The PDFs from which the values are chosen must be carefully defined to ensure that the model output is statistically meaningful (Stephens et al. 1989).

Because the biosphere model is generic, a different hypothetical environment is considered in each simulation. The PDF for each parameter must therefore encompass the full range of values that could be encountered on the Ontario portion of the Canadian Shield. They must also cover the range of values that could occur on the Shield during interglacial conditions over the next 10 000 a or more. Finally, they must cover the range of values resulting from uncertainty in the model or the data (Section 1.5.7). Such uncertainty may arise from measurement error, or from the need to extrapolate data obtained under one set of conditions to another set. Uncertainty may also arise when parameter values are used to offset deficiencies in our knowledge of the processes being modelled, or when parameters are combined to describe the net effect of several processes in a simplified bulk transfer model. To the extent possible, the PDFs should be constructed to reflect the probability that, in the most accurate description of the system, the parameter (in the form required by the model) will assume a particular value, taking into account spatial and temporal variations and the dominant sources of uncertainty.

Stephens et al. (1989) set out guidelines for defining PDFs for the parameters that appear in the concept assessment models. The number of such parameters is very large, and the number and quality of the data available for each are extremely variable. Accordingly, the guidelines consist of background information and suggestions rather than detailed procedures.

Specification of the PDF type may begin by plotting the available data in the form of a histogram to obtain a visual picture of the shape of the distribution. Theoretical arguments, statistical analysis and the meaning of the available data are all used to help establish the most appropriate distribution type (e.g., normal or lognormal). The attributes of the distribution (e.g., central tendency and measure of variation) may be obtained by statistical analysis once a PDF type is chosen, or may be set subjectively using arguments based on the most probable value of the parameter, and on its upper and lower limits. Both the PDF type and its attributes may be modified subjectively to account for uncertainty in the data or for changes that could occur in the parameter values over the course of time. A continuous PDF may have to be truncated at its upper or lower end to eliminate physically impossible values. Similarly, it may be necessary to correlate two or more parameters to avoid unreasonable combinations of values. We have followed these guidelines to the extent possible in defining the PDFs for the biosphere parameters.

2.7.2 Practical Application

The biosphere model contains well over 100 parameters, many of which are radionuclide- or element-specific. The data available for defining PDFs are very uneven with respect to number and quality. In general, the parameters that define the physical state of the relevant biosphere (precipitation, soil type, lake size, etc.) have been observed for many years, and their values are well established in space and time. The physical and chemical processes that occur in soils, water bodies and the atmosphere have also been well studied, and their associated parameters are reasonably well known. But we do not have the same understanding of the biological processes at work in the environment, or of the behaviour of contaminants released into the biosphere. The environmental behaviour of some nuclides has been well studied, but information for many others is less complete. Parameters such as the soil partition coefficient (Section 6.5.3) and the plant/soil concentration ratio (Section 8.5.1.1) describe a variety of complex processes and are subject to much uncertainty. In some cases, data obtained under appropriate field conditions are too limited to define PDFs for the biosphere parameters. The available information must usually be augmented by laboratory data, theoretical considerations and expert opinion.

PDFs for most biosphere parameters were obtained in the following way, as described in the four submodel reports. All the relevant data were first assembled. Where possible, data from the Ontario portion of the Canadian Shield were used exclusively, but often it was necessary to include information from other areas. The data were then plotted in the form of a histogram, and a PDF type was selected. This could occasionally be done objectively using standard statistical tests (Sokal and Rohlf 1981). More commonly, the limited amount of data dictated a more subjective approach, and the PDF type was deduced from an overall appraisal of the shape of the distribution. Theoretical arguments can also be useful in this regard. For example, many parameters can be assumed to be distributed lognormally since they are expressed as the product or quotient of other variables

(Sheppard S.C. and Evenden 1988). The common occurrence of lognormally distributed environmental concentrations has also been explained on the basis of successive independent random dilutions (Ott 1990). However, some environmental parameters are known to be normally distributed. For both lognormal and normal distributions, mean values and, occasionally, standard deviations were calculated directly from the data. More commonly, standard deviations were chosen nominally to cover the observed range of values with a high probability.

PDFs constructed in this way reflect spatial variability and uncertainty in the data. They also reflect temporal variability over the relatively short time periods represented by the data (Section 1.5.7). It is not possible to account objectively for changes in the parameter values that could occur over the next 10 000 a. Long-term temporal variability is incorporated subjectively into many of the PDFs by assuming somewhat more variability than the data for present-day conditions imply. Thus, with a selection of extreme parameter values, the model may simulate more extreme fluctuations than one might expect in the real system.

As a rule, the PDFs for the biosphere parameters are not truncated. Exceptions are made only to avoid physically impossible situations, to avoid conflicts with assumptions made about the critical group, or to maintain consistency with the site-specific characteristics of the geosphere model. For example, the PDF for the radon aquatic transfer coefficient (Section 7.5.1.4) is truncated at its lower end at zero because negative values are not possible. The distribution for mean lake depth is truncated at its lower end at 1 m. Shallower lakes would freeze to the bottom during winter, and could not act as a supply of water and fish for the critical group. A truncated PDF is automatically rescaled in SYVAC3 so that its integral continues to equal unity.

Correlations are imposed between pairs of biosphere parameters where the data or theoretical considerations suggest that correlations exist. In this way, unrealistic combinations of parameter values are avoided, and the corresponding exaggerated variability in the calculated doses is reduced. For example, the plant/soil concentration ratio, Bv, is assumed to be negatively correlated with the soil solid/liquid partition coefficient, Kd. Nuclides that are strongly bound to soil solids (and therefore have a high Kd value) are not readily available for root uptake (and therefore have a low Bv value). In SYVAC3, the value x_c of a parameter that is correlated with a second, independently sampled parameter with the value x_i is given by (Kleijnen 1974)

$$x_c = \mu_c + r \cdot (x_i - \mu_i) \cdot \sigma_c / \sigma_i + \sigma_c \cdot \Gamma \cdot (1 - r^2)^{1/2} \qquad (2.5)$$

where μ_c and σ_c are the arithmetic mean and standard deviation of the correlated parameter,

μ_i and σ_i are the arithmetic mean and standard deviation of the second, independently sampled parameter, which has a sampled value of x_i,

r is the linear product moment correlation coefficient, and

Γ is a random number chosen from a normal distribution with an arithmetic mean 0 and standard deviation 1.

Equation (2.5) can be used in SYVAC3 only for parameters that are normally or lognormally distributed. In the case of lognormal PDFs, Equation (2.5) is applied to the logarithms of the parameter values. All of the correlated parameters appearing in BIOTRAC are normally or lognormally distributed, so Equation (2.5) can be used in all cases.

The difficulties involved in defining a PDF usually increase as the amount of data decreases. However, the smaller the amount of data, the larger the uncertainty, and the more desirable it becomes to represent the parameter using distributed values. We have adopted the position that, wherever possible (and with a few exceptions as noted below), every parameter should be described by a PDF. The procedures used to construct the PDFs where limited data exist depended on the parameter in question and on the amount and type of data available. Data are totally missing only for a few nuclides in the case of some nuclide-specific parameters. PDFs for these nuclides were based on the distributions derived for other nuclides with similar physical and chemical properties. For the other incompletely defined parameters, the PDF type was decided on theoretical grounds, or specified through expert opinion. Mean values were determined from the available data, and standard deviations (SDs) and geometric standard deviations (GSDs) were set large enough to cover the observed or expected range of values. In each case, every effort was made to ensure that the PDFs chosen would cover most or all of the physically possible values in order to overestimate the consequences.

A few biosphere parameters were not distributed deliberately. Fixed values were adopted for such parameters as the radioactive decay constants, which have well-defined values (Table 1-1). Some of the parameters describing the biosphere/geosphere interface were fixed to maintain consistency with the geosphere model (Chapter 4). Fixed values were also occasionally used for parameters that showed relatively little variation, that had little impact on the calculated consequences, and for which a clearly conservative value could be identified. All the switch parameters of BIOTRAC were assigned fixed values (Section 2.7.3). The use of fixed values in such cases is reasonable, and makes the results easier to interpret.

The various biosphere parameters are addressed in detail in Chapters 4 to 9 and Chapter 13. The data available for each parameter are summarized, together with a description and justification of the PDF selected for each. The selection of parameter values is discussed in more detail in the four submodel reports.

2.7.3 Switch Parameters

BIOTRAC contains a number of switch parameters that could strongly influence consequence predictions. These parameters determine the occurrence of irrigation of plant crops (Section 6.5.5.2), the use of lake sediments as soil (Section 6.5.5.4), the use of peat as heating fuel (Section 7.5.2.7) and the exact source of domestic water (Section 9.1.2). All these parameters have fixed probability values to decide on a choice, e.g., the use of well water or lake water for domestic purposes. Literature data for determining switch-parameter values are limited, and we have attempted to make conservative yet reasonable choices. The switch parameters are closely related to the scenario analysis (Section 1.5.1).

2.7.4 Normal and Lognormal Distributions

Most of the parameters in BIOTRAC are assumed to be distributed normally or lognormally. We characterize a normal distribution by the arithmetic mean, μ, and by its SD, σ. We define a lognormal distribution using the geometric mean (GM) of the parameter and its GSD. Note that

$$GM = e^{\mu_e} = 10^{\mu_{10}} \tag{2.6}$$

and

$$GSD = e^{\sigma_e} = 10^{\sigma_{10}} \tag{2.7}$$

where μ_e (μ_{10}) and σ_e (σ_{10}) are the arithmetic mean and standard deviation of the $\log_e$ ($\log_{10}$) transformed distributions. The GM also approximates the median of the untransformed lognormal distribution.

The criteria proposed by the AECB (1987) for assessing the performance of the disposal facility involve the prediction of the arithmetic mean of the estimated doses at a given time in large sets of SYVAC3 simulations. Arithmetic means tend to be dominated by any large values in the sample. This means that a lognormal distribution for a given parameter will tend to produce a more conservative result than a normal distribution, assuming large values of the parameter lead to high doses. Similarly, the arithmetic mean becomes more conservative as the lognormal distributions are widened.

3. ENVIRONMENTAL CHANGE

3.1 TIME DEPENDENCE AND BIOSPHERE MODELLING

The biosphere, and the processes that control nuclide transport through it, are fundamentally time-dependent (Davis 1986). On the shortest time scales, variations in the amount of solar radiation reaching the earth's surface drive diurnal and seasonal cycles in the atmosphere, which are reflected in other biosphere compartments. In contrast, geological processes, such as mountain building and continental drift, occur over hundreds of millions of years. Processes with time scales of about 10 000 a or less will affect the Canadian Shield over the assessment period. Environmental changes should be taken into account if they significantly alter the rate or pattern of nuclide migration and their consequences on the environment and humans.

The parameter values sampled at the beginning of each SYVAC3 simulation are held constant throughout the simulation. Although the parameter PDFs reflect, in part, variations in time, they are used to describe different permanent states of the vault, geosphere and biosphere, rather than a system that varies in time. Therefore we do not explicitly model environmental change on any time scale. But we do take time dependence into account in other ways, which are described in the rest of this chapter and in Chapter 12.

3.2 FLUCTUATING PROCESSES

Fluctuating processes are those that show relatively rapid variations in time about some mean condition. For example, precipitation varies from day to day, although its annual average value at a given location remains fairly constant. Similarly, strong seasonal fluctuations occur in the rate of nuclide transfer from the water column to sediments in a lake, but annual average rates are much the same from year to year. The variability in most processes tends to decrease as the time scale increases. The standard deviations of PDFs constructed from annually averaged parameter values are generally less than those formed from daily averaged data.

Because means and fluctuations can be defined on all time scales, the relevant scale for a given application depends on the desired result. The main end point of BIOTRAC is to calculate annual doses to humans and other organisms (Sections 1.2.3 and 1.2.4). Accordingly, averaging times of one year are appropriate. Although fluctuations on scales less than one year are common, they generally have no effect on the calculated annual dose. Because doses are additive, and because the surface water, atmosphere and food-chain submodels are linear (Chapters 5, 7 and 8), doses calculated for these compartments using annual average values are equal to the annual sum of doses calculated at finer time scales. On the other hand, the soil submodel (Chapter 6) is non-linear. Annual average nuclide concentrations in soil depend on weather patterns throughout the year, as well as on their annual values. In this case, short time-scale fluctuations are important, and are taken into account by basing the model on daily average data.

The biosphere also varies on time scales greater than one year. For example, the climate exhibits a continuous spectrum of fluctuations ranging up to decades and centuries even during interglacial conditions (Morner and Karlen 1984). Although we do not model these processes explicitly, we account for their effects through the PDFs (Davis 1986), as explained in the following comparison.

Consider a biosphere that is constant except for a single process, characterized by a parameter, Pr, that varies on a time scale, τ, that is substantially less than the total simulation time, T. We assume further that Pr has no spatial variability, and can be measured precisely, so that the distribution of Pr values reflects only the time variation in Pr. In general, some values in the distribution will lead to high predicted doses and some to low ones.

Using the random sampling methodology employed in SYVAC3, the value of Pr selected at the beginning of a particular simulation is held constant throughout the simulation. This value leads to a particular dose curve, which is a smooth function of time, as shown schematically in Figure 3-1(a). Doses remain zero until the first nuclides reach the biosphere; they then increase as the nuclide flux from the geosphere increases. If T is sufficiently large, radioactive decay or depletion of the vault inventory eventually leads to a decrease in the geosphere flux, and doses decline.

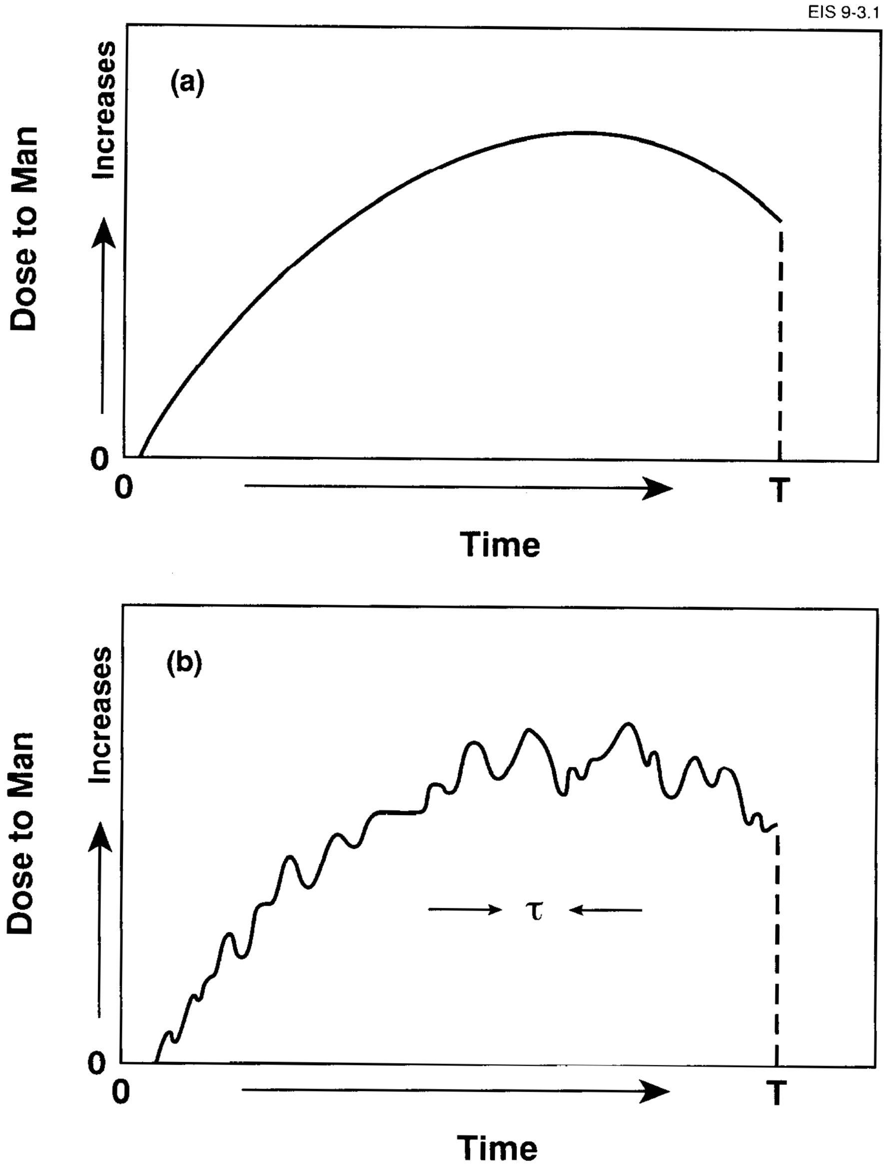

FIGURE 3-1: Schematic Representations of Dose Curves to Time, T, when (a) Pr is Held Constant Throughout the Simulation (Histogram H_c), and (b) Pr is Sampled Throughout the Simulation at Frequency τ (Histogram H_v)

Repeated simulations with new values of Pr produce similar curves. One of the primary outputs of SYVAC3 is a consequence-frequency histogram based on the doses calculated in each simulation at some arbitrary point in time, t_p. We will denote this histogram H_c.

Consider now the case in which time dependence is modelled explicitly by choosing new values of Pr throughout the simulation at a rate $\Delta t \sim \tau$.

Assuming that the predicted dose responds rapidly to changes in Pr, the dose curve (Figure 3-1(b)) will no longer vary smoothly with time, but will show fluctuations of periodicity τ. The consequence-frequency histogram, H_v, would be constructed in the same way as before, but from a different set of doses.

The histogram H_c is made up of doses determined by values of Pr sampled at time $t = 0$. On the other hand, H_v is constructed from doses determined by values chosen at $t = t_p$. The two histograms will be statistically identical if the PDF for Pr is independent of time. This is in fact the case for PDFs associated with fluctuating processes because they describe short time-scale variations for a system that is in the same basic state from $t = 0$ to $t = t_p$. Accordingly, the effect of fluctuating processes on predicted doses has been implicitly accounted for through the use of distributed parameter values.

The above argument relies on the assumption that the predicted dose responds so rapidly to changes in Pr that the dose at time t_p depends only on the value of Pr chosen at $t = t_p$, and not on the history of Pr values selected prior to t_p. This condition is satisfied for the rapid processes in surface waters, the atmosphere and the food chain, but not necessarily in soils, where the time to steady state is much longer than the time scales of the processes driving the system. A fluctuation in the controlling parameters may push the real soil system in one direction or another. But before steady state can be reached, another fluctuation may push it in another direction. The final soil concentrations in the real system will reflect all perturbations, but will tend toward an average value if the system experiences a large number of fluctuations over the period of interest. In contrast, the predictions of the soil model will tend to extreme values because the input parameters are assumed to remain constant over the entire simulation period, even if they take on values that are more representative of fluctuations than of means. The model will therefore tend to overpredict or underpredict concentrations; however, the arithmetic mean over many simulations will be conservative because the overpredictions will dominate the mean, given a lognormal distribution (Section 2.7.4).

3.3 TRANSITIONAL PROCESSES

The biosphere is subject to changes other than those imposed by fluctuating processes. These may take the form of specific, short-term, nonperiodic events, such as meteorite impact or major seismic activity. They may also appear as persistent trends caused by processes such as continental glaciation or tectonic drift, with time scales that are long compared with the time frame of interest here. In either case, these transitional processes cause quasi-deterministic changes in the biosphere that could affect the rate of nuclide migration through it.

It is difficult to predict the state of the Canadian Shield environment far into the future. The forces driving environmental change are, at best, incompletely understood, and at worst, unpredictable. In addition, the time-dependent behaviour of the various biosphere parameters during a given transitional event must be deduced from proxy data, and so cannot be predicted with any certainty. We believe that a detailed, time-dependent treatment of transitional processes is not warranted since the results would not be credible. Instead, we have developed an alternative approach based on the following guidelines (Davis 1986).

1. Processes are considered only if their temporal variations could have a significant effect on health or environmental consequences.

2. The prediction of future events is based on the assumption that the geological history of the biosphere will be repeated. This will likely be the case if there is no interference from humans. The human potential for altering the biosphere is great; however, the nature of human impact is difficult or impossible to predict, and our attempts to model it would not be productive.

3. The stochastic nature of the assessment is exploited by addressing as many time-dependent processes as possible through variations in parameter values or distributions rather than through explicit modelling.

4. Supported logical arguments are used in place of mathematical modelling to address transitional events extending beyond 10 000 a (AECB 1987).

Merrett and Gillespie (1983) and Davis (1986) reviewed the transitional processes that have affected the Canadian Shield in the past. They found that the majority of the processes do not require treatment because their potential to affect nuclide migration through the biosphere in the future is small. The probability that the vault area will be subject to tectonic activity, volcanism or meteorite impact over the next few million years is so low that these processes may be excluded (AECL 1994a, Goodwin et al. 1994). Tectonic drift, denudation and fluvial erosion on the Shield occur so slowly that they will induce no significant change in the biosphere as far as the disposal facility is concerned. Seismic activity is low and localized, and its effects can be minimized by careful vault siting (Davison et al. 1994a). Reversals of the earth's magnetic field can be expected to occur in the future, but their effects on the biosphere cannot be predicted or modelled (Goodwin et al., in preparation).

Of all the processes considered, Davis (1986) concluded that only human activities and continental glaciation (including glacially induced faulting and succession in a glacial regime) have the potential to alter the biosphere and the rate of nuclide migration through it. The treatment of these processes from a biosphere perspective is described in the following sections, and in Chapter 12. The impact of glaciation on the disposal facility itself has been discussed by Ates et al. (1991), and the effects of glaciation on groundwater flow and nuclide transport through the geosphere are described by Davison et al. (1994b).

3.3.1 Anthropogenic Effects

3.3.1.1 Effects on the Biosphere

Man continually disrupts the biosphere locally through activities such as agriculture, mining, forestry and recreation. Human activities have begun to alter the biosphere on much larger scales. Acid rain is modifying the chemical balance of soils and lakes on the Canadian Shield. The reduction of the ozone layer may allow higher levels of ultraviolet light to reach the earth's surface. These processes could cause changes in the types of flora and fauna that the Shield environment can support.

The injection of large amounts of particulates and "greenhouse" gases (particularly CO_2 from fossil-fuel burning) into the atmosphere may be changing the earth's climate. Although the effects of elevated CO_2 levels are under debate (Seidel and Keyes 1983, USNRC 1983a, Idso 1984, AMS 1991), it is generally agreed that one result will be an increase in the mean global temperature, at least in the short term. Significant alterations in the global wind pattern, and hence in precipitation regimes, could follow. If the warming persists, melting of the Antarctic and Greenland ice sheets could occur, resulting in a global rise in sea level and flooding. There could be many changes on the Canadian Shield, including a northward shift in climatic zones, changes in the amount of precipitation, a northward shift of vegetation zones, degradation of permafrost, and extirpation of some plant and animal species (Environment Canada 1991). It should be noted that similar and much more drastic changes have occurred in the past as a result of natural climate change. Although acid rain, ozone depletion and increased atmospheric CO_2 concentrations are established environmental facts, their long-term effects remain open to speculation.

In the case of global warming from the buildup of greenhouse gases, changes in climate variability may have a greater impact than changes in average climate conditions. For example, some researchers have indicated that temperature variability may decrease, and precipitation variability increase as the climate warms (Mearns 1991). However, there is still much uncertainty regarding human-induced global warming, both because the climate data accumulated over the past century are insufficient to decipher the exact nature of changes that have taken place, and because the climate is characterized by substantial variability over different time scales (AMS 1991). Given this uncertainty, it is difficult to evaluate the impact of climate change from the buildup of greenhouse gases on the biosphere. All that can be done is to use broad distributions for the climate parameters in BIOTRAC to cover a wide variety of possible outcomes.

Future human interaction with the biosphere is difficult to predict. Technology is advancing at such a rate that it is conceivable that it will lead to control of the biosphere in the not too distant future, through the genetic manipulation of plants and animals and the control of climate, including glaciation. Humans may also learn to control the rate of nuclide migration through the biosphere, or mitigate the effects of any nuclides that do reach the biosphere. On the other hand, civilization may return to a primitive state as a result of war, exhaustion of natural resources, severe degradation of the environment, or problems associated with overpopulation. In all these cases, the biosphere will suffer a parallel change, at least temporarily. In the face of this uncertainty, any attempt to predict the effect of humans on the evolution of the biosphere would be futile. Accordingly, we have assumed that future human activities, whether for better or worse, will not alter the biosphere in any fundamental way over long periods of time.

To the extent that acid rain, the destruction of the ozone layer and global warming by greenhouse gases have already affected the biosphere, they are implicitly reflected in our database and models. Allowance for future effects of these processes can be made by using broadly distributed model parameter values. Future interactions of humans with the biosphere are unpredictable and we have made no allowance for them in BIOTRAC. However, future interactions with the disposal facility can be evaluated (Section 3.3.1.2).

3.3.1.2 Human Intrusion

One final aspect of man's activities is the possibility of direct intrusion into the vault through drilling, mining or the use of explosives. In addressing this question, Merrett and Gillespie (1983) concluded that accidental intrusion would be unlikely because of the small size of the vault relative to the total area of the Canadian Shield, its great depth, and its location in a type of rock that is plentiful at the surface and of low economic value. One of the siting criteria for the disposal facility will be the absence of any valuable natural resource such as minerals (Davison et al. 1994a). Deliberate intrusion as a result of sabotage or curiosity is unlikely because of the magnitude of such an operation; it could likely not be carried out without the knowledge or approval of the authorities of the day. Deliberate intrusion to salvage the materials placed in the vault is a possibility, but Merrett and Gillespie (1983) concluded that a society advanced enough to want the materials, and capable of retrieving them, would also be aware of the hazards involved and take steps to minimize them.

We have investigated a number of inadvertent human intrusion scenarios in detail (AECL 1994a). The consequences of a bedrock well drilled into the groundwater plume are calculated explicitly in the model (Davison et al. 1994b). Well depth is a lognormally distributed parameter, but is truncated at 200 m because of the subsurface fracture geometry at the WRA (Section 4.3). Wells up to 500 m deep are known, but are extremely rare. The probability that a well will exceed 500 m in depth can be estimated as 5×10^{-4} by extrapolating the well-depth distribution. This is so low that a well drilled directly into the vault is highly unlikely. Even so, we have considered an intrusion scenario of drilling into the vault, as discussed below.

Four other intrusion scenarios were assessed by Wuschke (1992), whose analysis is summarized here. The specific scenarios were:

1. Exposure to undispersed waste of a member of a crew drilling a core into the vault.

2. Exposure to undispersed waste of a geotechnical laboratory technician examining core material taken from the vault.

3. Exposure to waste dispersed by previous intrusion of a worker building a house on the exposed waste.

4. Exposure to waste dispersed by previous intrusion of a resident in a house built on the exposed waste.

These scenarios were considered likely to present the highest risk to the intruder and each is intended to be generally representative of a set of similar scenarios with lower probabilities or consequences. The assessment was done with a package of well-documented codes produced by Battelle Pacific Northwest Laboratories for the United States Nuclear Regulatory Commission (USNRC) (Wuschke 1992). These codes were chosen as a consistent package with the capability to model a variety of external exposure geometries.

In order to assess the risk from these intrusion scenarios, both the concentrations of nuclides in the vault and the probabilities associated with each scenario were required. The concentrations in the vault were computed from the estimated initial inventories, based on ten-year-cooled fuel irradiated to a burnup of 685 $GJ \cdot kg^{-1}$ uranium, and the reference design for the vault. The assignment of probabilities to each scenario was more complex and less exact. Each scenario implies a series of events such as selecting a drilling site, missing controls or warnings about the vault, and continuing drilling to vault depth. A probability was assigned to each of these events, based largely on present experience, geotechnical considerations and technology. Time-dependent probabilities were assigned where appropriate. The probabilities for the full scenarios were then computed as the products of the probabilities of the implied events.

The estimated radiation doses were significant (i.e., they exceeded the AECB (1987) risk criterion if the scenario was certain to occur) for all four scenarios. Inhalation of dust was the most important pathway for most times. However, the probabilities were extremely low. As a result, the risks ranged from zero, at vault closure, to a peak value of 2×10^{-11} serious health effects per annum. This peak occurred after 500 a following vault decommissioning and closure, when institutional controls were assumed to have lost their effectiveness. All the risks computed were substantially below the AECB criterion of 10^{-6} serious health effects per year (AECB 1987).

3.3.2 Continental Glaciation

As part of our evaluation of the effects of glaciation on nuclide transfer and dose, we used BIOTRAC in a somewhat modified form to calculate dose to the critical group as an indicator of consequences in different glacial states. Interpretation of the results requires an understanding of the model and its parameter values. Since these topics are introduced in Chapters 4 through 10, the discussion of glaciation is provided in Chapter 12. The exception is succession in a glacial regime, which can be usefully presented at this point.

3.3.2.1 Succession in a Glacial Regime

Succession is the process whereby one type of biotic community or physical system is gradually replaced by another. The Canadian Shield shows a characteristic pattern of succession following the retreat of a glacier (Ritchie and Yarranton 1978). The climate gradually warms, and air mass movements and precipitation patterns similar to those of today become established. The large volumes of water associated with the melting ice recede, leaving behind many wetland areas and small lakes, which gradually fill in. Soils in deglaciated areas develop according to the type of parent material, drainage conditions and climate they experience. Eventually, the soil over most of the Shield develops into a podzol (Chesworth et al. 1985). In regions covered with some sort of glacial deposit, a tundra ecosystem becomes established upon deglaciation, changing to a spruce-dominated boreal forest within 2000 a. Pine and birch begin to appear with further warming, leading to a forest very similar to today's within 5000 a of the retreat of the ice (Ritchie 1984). Deglaciated areas are repopulated by animals and by humans from refugia as soon as conditions permit (Gordon 1975, Pielou 1991).

The Canadian Shield biosphere has changed significantly since the retreat of the last ice sheet, and the present rate of succession is generally slow. The climate shows fluctuations on many scales, but is in a more-or-less steady-state condition. Soil profiles are generally mature, although they are still undergoing some development, particularly in the north. Succession remains dynamic within plant and animal communities; however, the critical group experiences a biosphere that is largely agricultural in nature, and that is subject to agronomic management practices rather than to normal succession. Accordingly, for the most part, the Shield biosphere will not undergo any successional changes of consequence to the critical group as long as interglacial conditions persist. Small changes can be accounted for through our parameter distributions.

One aspect of succession cannot be treated in this way. The discharge water body, if it is a typical Shield lake, will fill in significantly over the course of the next 10 000 a. Its area and depth will decrease, the flow through it may change, and its bottom sediments may become exposed and used for agricultural purposes. The dose to humans and other biota arising from the surface water pathways may depend upon the successional state of the biosphere.

Sundblad et al. (1988) and Smith G.M. (1989) have modelled the long-term dynamics of lake ecosystems and calculated nuclide concentrations at different points in their evolution. Both studies found that nuclide concentrations in water slowly increase as the lake fills in and its volume is reduced. The total increase in concentration over the course of the change depended on the sorptive behaviour of the nuclide under consideration, but was typically a factor of two, and no more than a factor of five. Sediment concentrations also increased with time, with concentrations at the end of the infilling process ranging from a factor of three to almost three orders of magnitude greater than concentrations in the young lake. The concentrations in sediments after their transformation into soil depended strongly on assumptions concerning the location in the lake at which the sediments accumulate, nuclide mobility in the sediment after it has dried out, and the land area over which to average the concentration. However, in all cases concentrations in sediments used as soil were lower, and usually substantially lower, than concentrations in the sediments proper, primarily because nuclides were removed by leaching.

Sundblad et al. (1988) extended their calculations to estimate doses at three stages of lake evolution: a lake phase in which no significant infilling had occurred, a soil phase in which the lake had dried out to the extent that sediment had been converted to farmland, and an intermediate phase in which sediment near the lake shore was being transformed into soil. They found that doses to humans in the lake and intermediate phases were very similar. Doses in the soil phase for nuclides of high mobility were lower than for the preceding phases; for nuclides of low mobility, doses in this final phase increased by up to a factor of 100 as a result of accumulation in the sediment. The pathways responsible for increased doses after conversion of lake sediment to soil were inhalation of dust and soil/plant/man transfer. These results suggest that the particular successional stage of the discharge lake has little effect on dose to humans, and probably to other organisms. The variation in dose that does occur during the aging process can be captured by considering the final successional phase alone.

We do not treat lake infilling in detail, but we used two different ways to model the scenario in which lake sediments are used as agricultural soils. First, we assign a relatively high value to the probability that the critical group will farm organic soils (Section 6.5.1.1). Since the gradual exposure and weathering of lake sediments produces a soil with organic characteristics, this simulates in a rough way the end point in the lake succession scenario, which is the important phase with regard to dose assessment. Secondly, in 1% of all simulations, we use sediment concentrations in place of soil concentrations throughout BIOTRAC (Section 6.3.7.3). This allows a simple treatment of the consequences that could result if fresh sediment were dredged and applied to an agricultural field, or if the lake were drained and farmed. The sediment concentrations are used as predicted, without credit for loss of nuclides through leaching, degassing, conditioning, mixing with existing soil or radioactive decay before the sediments become suitable for agricultural purposes. When sediment is used as soil, the contaminated water from the lake is still assumed to be available for use in the surface water pathways (Section 9.1.2), even though the lake may have partially filled in or been drained.

4. GEOSPHERE/BIOSPHERE INTERFACE

The geosphere is the source of all the nuclides entering the biosphere from the vault. Figure 4-1 shows a schematic cross section through a generic geosphere/biosphere interface. Plutonic bedrock surrounds the vault, and is covered at the ground surface by a layer of saturated unconsolidated material (overburden) that consists mostly of glacial deposits on the Canadian Shield, and by a layer of soil. In aquatic regions, the overburden is overlain by two sediment layers, both of which are derived from material deposited from the water column. The top layer is newly deposited and kept well mixed by processes such as scouring and bioturbation. The bottom layer is older sediment that has become compacted. The geosphere model tracks nuclide flow from the vault to the top of the compacted sediment layer, at which point the biosphere model takes over. Accordingly, for modelling purposes, the interface is assumed to lie between the mixed and compacted sediment layers in aquatic regions. In terrestrial regions, the overburden is covered by unsaturated soils. Since transport within the overburden is treated by the geosphere model, and within the soil by the biosphere model, the interface is assumed to lie at the water table. A

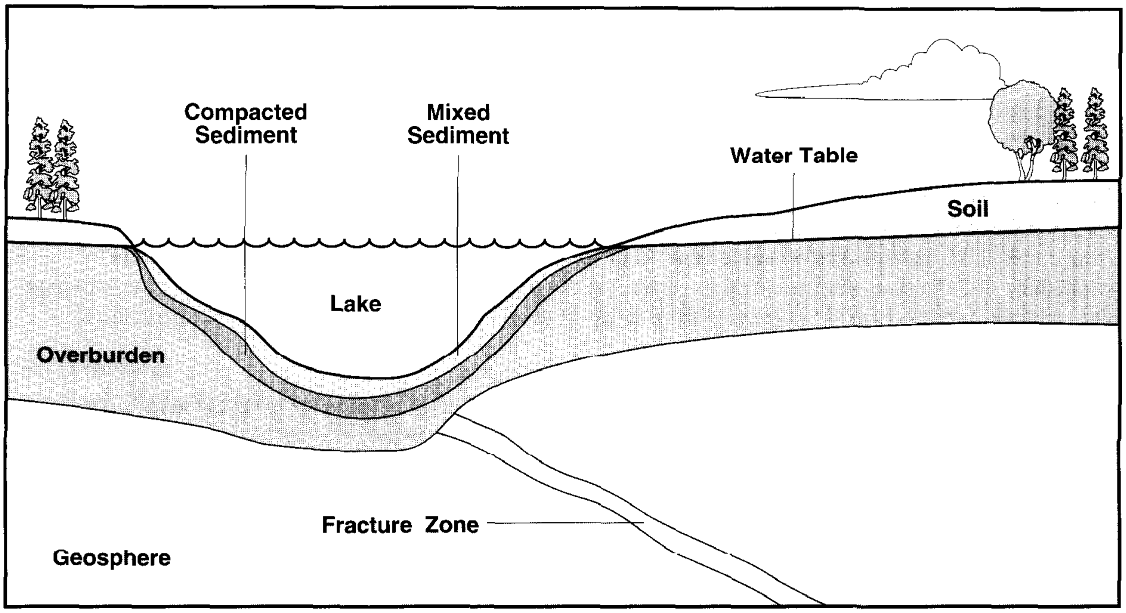

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FIGURE 4-1: Schematic Cross Section Through a Generic Geosphere/Biosphere Interface Showing the Arrangement of Bedrock, Overburden, Sediment Layers and Soil. The interface occurs in BIOTRAC between compacted and mixed sediments under the lake, and between overburden and soil in terrestrial regions.

well may be drilled into the geosphere (bedrock) to supply domestic water needs, providing a direct link between the geosphere and the biosphere (Section 3.3.1.2).

An interface model is required at these three types of contact points to ensure that the output of the geosphere model is compatible with the input requirements of the biosphere model, and because the biosphere model is generic whereas the geosphere model is site-specific. The geosphere model documented in detail by Davison et al. (1994b) is described briefly below to provide a basis for understanding the interface model.

4.1 OVERVIEW OF THE GEOSPHERE MODEL

The geosphere model is site-specific, and accordingly would be different for each potential disposal site. For the postclosure assessment, we have chosen to illustrate our modelling approach through a case study using data obtained in site characterization work at the WRA. The WRA covers about 750 km^2 in southeastern Manitoba (Figure 4-2), encompassing the Whiteshell Laboratories (WL) and the Underground Research Laboratory (URL). A major portion of the area consists of part of the Lac du Bonnet batholith, a large granitic pluton. The WRA is almost surrounded by the Winnipeg River system, and is drained by a series of lakes and rivers, including Boggy Creek, the Pinawa Channel, the Lee River, and Lac du Bonnet, which is formed by a hydroelectric dam. The geology, geophysics, hydrology, hydrogeology and geochemistry of the area have been studied and documented

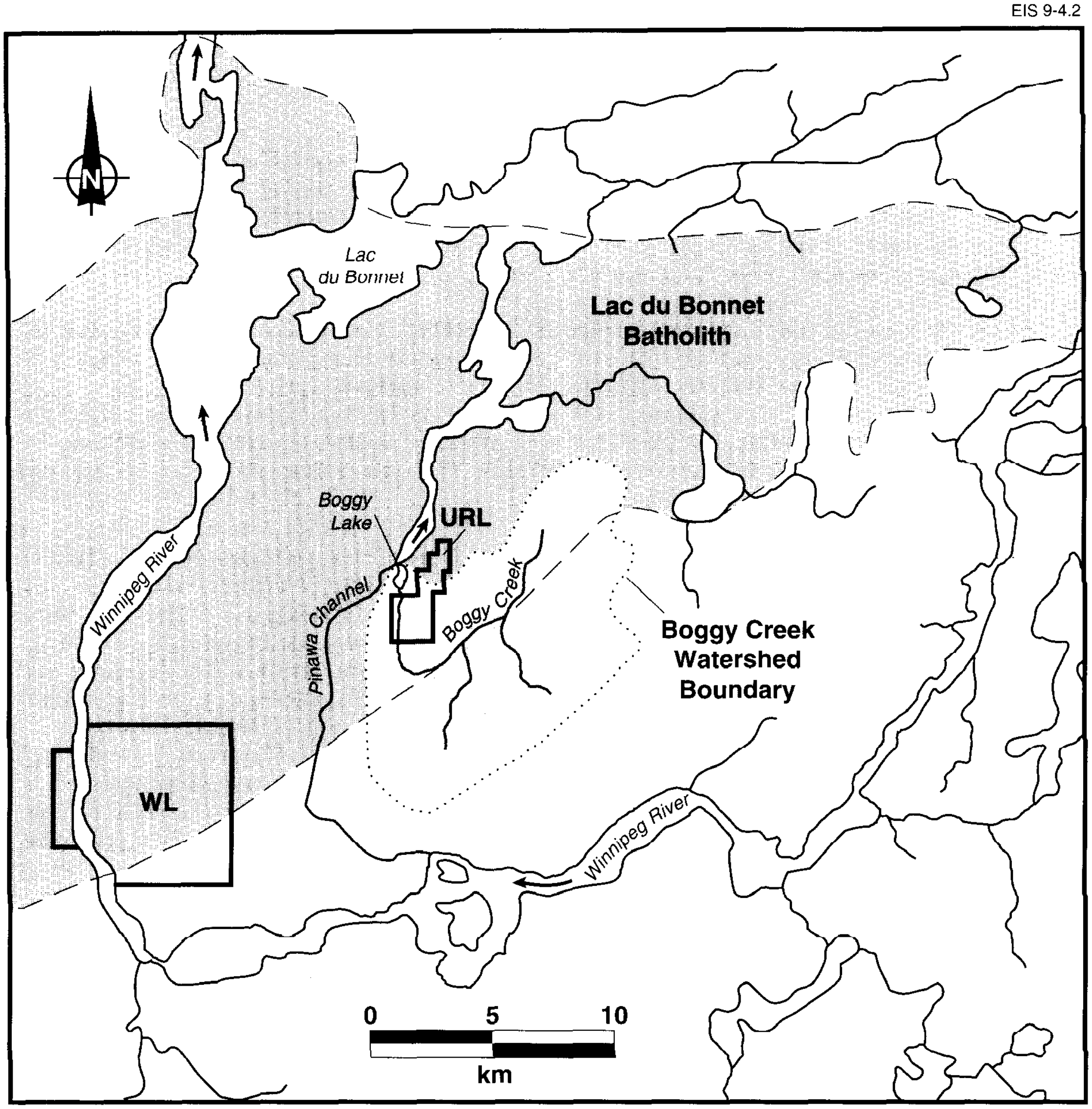

FIGURE 4-2: The Whiteshell Research Area (WRA) in Southeastern Manitoba. The Whiteshell Laboratories (WL) and the Underground Research Laboratory (URL) are shown as enclosures.

extensively (Betcher 1983, Davison 1984, Thorne 1986, Thorne et al. 1990, Griffault et al. 1992, Kozak and Davison 1992).

The geosphere model for the assessment was developed in three steps (Davison et al. 1994b, Goodwin et al. 1994):

1. A conceptual model consistent with the geological, geophysical and hydrogeological data obtained from field investigations was constructed for the subsurface structure and hydrogeology of the

WRA. This model summarizes our understanding of the hydrological units of the WRA with the help of maps, sections and other diagrams.

2. The observed groundwater flow at the WRA, containing a hypothetical vault and domestic well, was simulated using a detailed research model (MOTIF). MOTIF is a three-dimensional finite-element representation of the conceptual model to describe groundwater flow and solute transport.

3. A simplified assessment model (GEONET), suitable for Monte Carlo simulations, was developed to reproduce the groundwater flows predicted by MOTIF and incorporate chemical information to calculate the transport of nuclides from the hypothetical vault to the biosphere. GEONET is a network model consisting of one-dimensional segments that reflect the three-dimensional structure of the conceptual model.

The conceptual model (Davison et al. 1994b) describes the geological features that determine the groundwater transport pathways within the geosphere. The geological structure of the plutonic rock mass at the WRA is modelled using three rock zones (upper, intermediate and lower) to reflect the decrease in permeability and porosity of the rock mass with depth. A series of low-dipping fracture zones (designated LD0, LD1,...) cut through the rock mass and intersect the surface (Figure 4-3(a)). These fracture zones connect to vertical fractures (designated V0, V1,...) that extend from the surface into the intermediate rock zone, forming a continuous interconnected network. The fracture zones are considered to have a uniform thickness of 20 m, with a permeability two to six orders of magnitude larger than that of the rock mass. For modelling purposes, we located a hypothetical disposal vault in the conceptual model at a depth of 500 m in a region of the WRA that is most thoroughly characterized. The vault is purposely positioned to approach the plane of fracture zone LD1, which rises to the northwest to reach the surface below the open body of water, referred to as Boggy Lake, at the end of Boggy Creek. For the postclosure assessment, we assume that LD1 extends down to the proposed vault, although it does not actually do so (Goodwin et al. 1994). A region of sparsely fractured rock is assumed to separate the vault and fracture zone LD1 at their nearest approach (Davison et al. 1994a, Goodwin et al. 1994). We recognize that the presence of a major fracture zone close to the vault would be undesirable. However, the use of this hypothetical placement of the vault in the postclosure assessment case study has allowed us to examine the different actions that would be most effective in reducing undesirable consequences.

The conceptual model is completed by a domestic well near the shore of Boggy Lake that draws its water from fracture zone LD1. The well is very conservatively positioned so that it intersects the maximum amount of groundwater flow up the fracture zone from vault depth.

This conceptual model of the WRA geosphere has been translated into a research model, MOTIF (Guvanasen 1985, Chan and Stanchell 1990, Davison et al. 1994b), a three-dimensional finite-element code that can simulate the processes of groundwater flow, heat transfer and solute transport in

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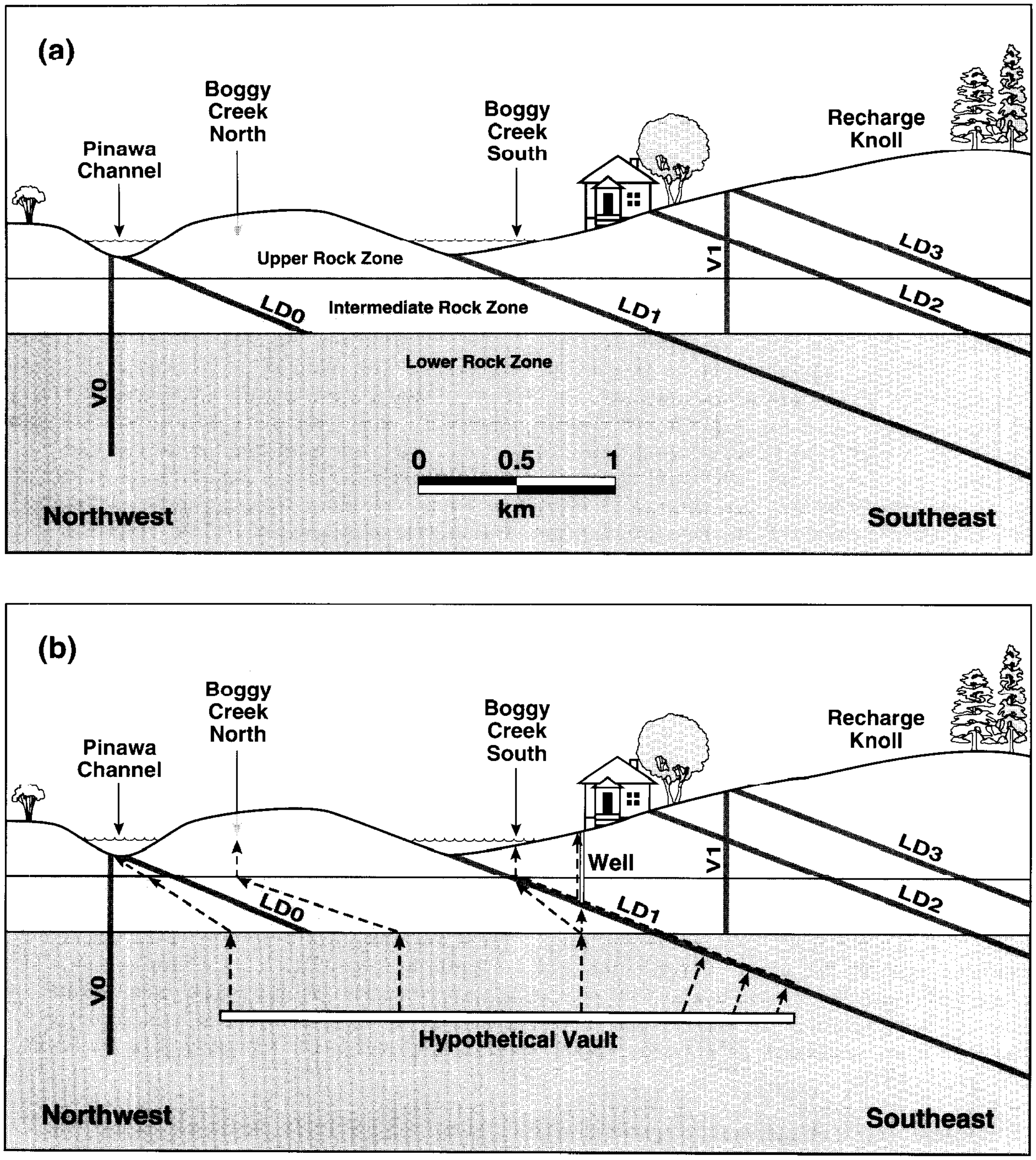

FIGURE 4-3: (a) Conceptual Model of the Subsurface Structure at the Whiteshell Research Area and (b) Groundwater Transport Pathways (Dotted Lines). In (a) the Pinawa Channel, Boggy Creek north and Boggy Creek south discharge zones are indicated. The Boggy Creek north discharge is located behind a knoll separating the Pinawa Channel and Boggy Creek south discharge zone. The vertical scale of surface features is exaggerated. Note that the fracture zone LD1 does not actually extend down to the vault level. It is only assumed to do so for the post-closure assessment.

saturated porous or fractured media. When used to model the WRA, its upper boundary condition is a prescribed hydraulic head distribution that reflects the local topography. The rock mass has fairly uniform thermal, mechanical and chemical properties, and was divided into the three rock zones based on permeability, which is constant within each layer, but decreases with zone depth. Although the permeabilities are relatively large near the surface, the upper zone has properties representative of the rock rather than of the overburden. The output of the model is the spatial distribution and time evolution of temperature and hydraulic head in the conceptual WRA geosphere.

MOTIF results were used to construct a simplified model of groundwater transport for use in the assessment. This simplified model, GEONET (Davison et al. 1994b), approximates the flow regime at the WRA by a network of one-dimensional flow paths connected in three-dimensional space (Figure 4-3(b)). Each pathway consists of one or more segments, and connects a sector of the vault to a surface discharge zone. The well is represented by a single segment in the upper part of the network. Segments can converge or diverge to allow capture by the well and leakage from the fracture zones. Physical and chemical properties are held constant along a segment. The topmost segment in each pathway (apart from terrestrial areas and the well) represents a compacted sediment layer, and the next deeper segment represents an overburden layer. Mass transport is calculated by solving a set of one-dimensional convection-diffusion equations for radionuclide decay chains of arbitrary lengths. Chemical sorption is taken into account using solid/liquid partition coefficients. The outputs of the model are the flow rates of water and nuclide mass from the geosphere to the biosphere for each pathway.

4.2 DISCHARGE ZONES

MOTIF calculations show that regional groundwater flow through the WRA geosphere is determined by the regional and local surface topography, and by the characteristics of the major fracture zones (Figure 4-3(b)). Water velocities of about 1 $m \cdot a^{-1}$ occur up fracture zone LD1, which acts as a conduit for water flow. Nuclides that reach LD1 would, therefore, be advected relatively quickly through the geosphere. Velocities are about four to six orders of magnitude lower elsewhere in the system, including LD0 and the vertical fracture zones, which are subject to very low hydraulic gradients.

At depth, the contaminated groundwater plume would have more or less the same cross-sectional area as the vault, the source of the nuclides. The plume becomes progressively narrower as it nears the surface and converges on the discharge zone. For calculation purposes in the GEONET model, the plume has been split into three distinct parts (excluding the well), which discharge in the vicinity of Boggy Lake (Figure 4-4). Nuclides originating in the southeast sectors of the vault tend to diffuse through a region of sparsely fractured rock into fracture zone LD1, where they are convected upward. Near the surface, where the fracture passes under Boggy Lake, the hydraulic gradients, and the generally higher horizontal and vertical permeability of the rock, force the plume out of the fracture and vertically upward to discharge near the south shore of Boggy Lake (Boggy Creek

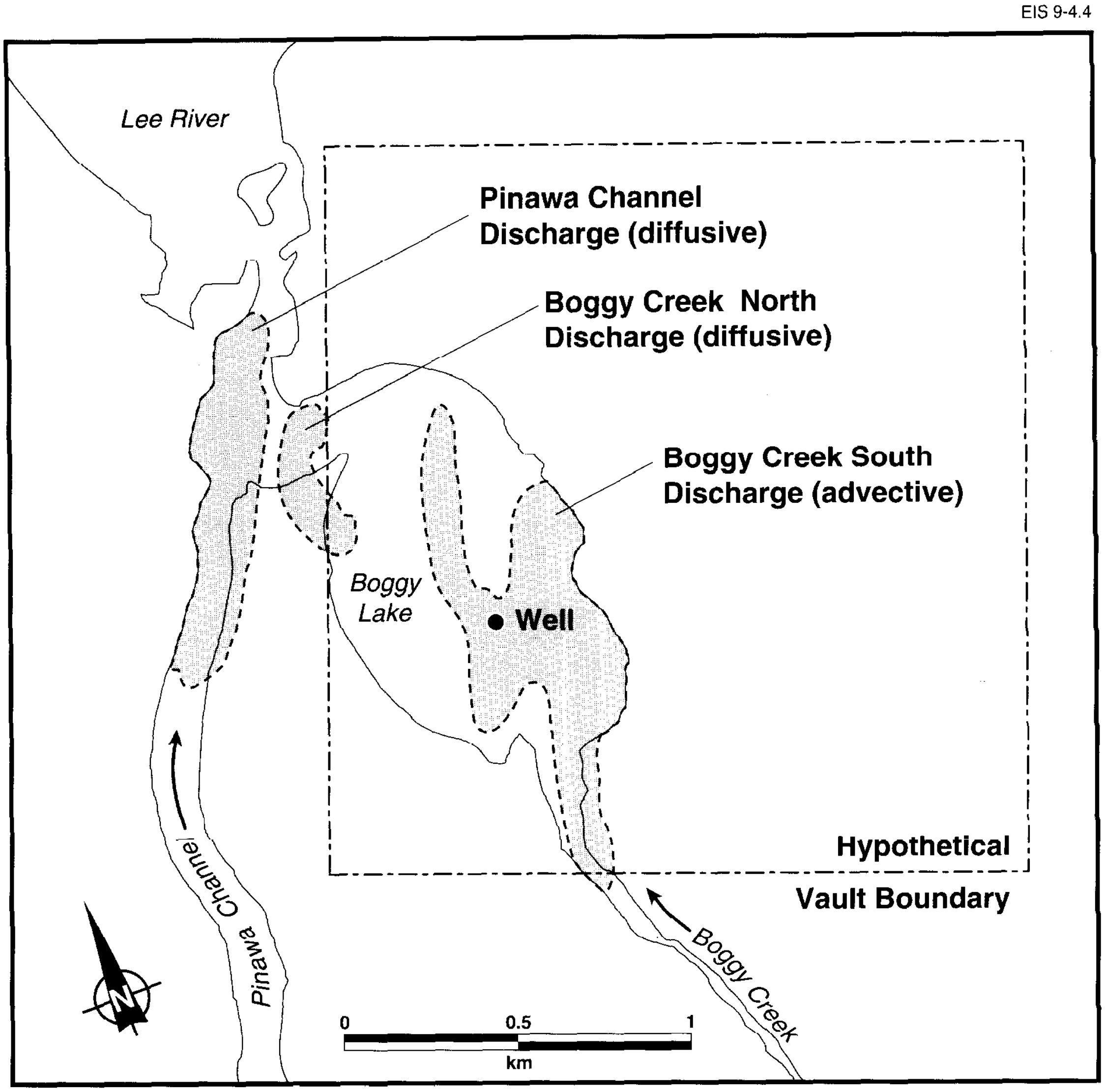

FIGURE 4-4: Zones (Shaded Area) where Groundwater Contaminated with Nuclides from the Hypothetical Vault is Predicted to Discharge to the Biosphere at Boggy Lake in the Whiteshell Research Area. The location of the well reflects its conservative placement in the potentially most contaminated area.

south discharge zone). Significant lateral dispersion is expected to occur in these near-surface layers, and the water velocities are reduced. Because nuclides following this pathway move relatively short distances through sparsely fractured rock, travel times through the geosphere can be expected to be short, of the order of thousands to tens of thousands of years (Chan and Stanchell 1990). Travel time depends strongly on the thickness of the region of sparsely fractured rock (Goodwin et al. 1994).

Transfer in the remaining two plume pathways is slower because of the greater distance nuclides must be transported through sparsely fractured rock. One pathway discharges into the north corner of Boggy Lake (Boggy Creek north discharge zone), and the other discharges into the Pinawa Channel (Pinawa Channel discharge zone). Although the overall length of these pathways is about the same as that of the Boggy Creek south pathway, travel times are much longer, typically millions of years.

Although MOTIF predicts that pathways from the vault discharge to aquatic areas only, it is possible that some pathways could discharge at a terrestrial location and come into contact with the bottom of the unsaturated soil zone. For example, relatively impervious clay layers in the overburden could channel the groundwater flow laterally under terrestrial zones near the margins of Boggy Lake. The detection of helium in the sediments and soils near the southeast shore of Boggy Lake suggests that some deep groundwater does discharge to this area (Gascoyne and Wuschke 1990, Stephenson et al. 1992), although the helium has not necessarily been transported via any of the plume pathways, or originated in the vicinity of the hypothetical vault location. Regardless, and to be conservative, we assume that a small portion of each of the three discharge zones underlies a terrestrial area that is suitable for farming or is available to terrestrial wild plants and animals. The soils of these areas may therefore become contaminated by nuclides moving upward from the water table through the soil profile.

A domestic well drilled into the contaminant plume represents another point of nuclide discharge. In order to access sufficient water to meet domestic demands, and to be consistent with the critical group concept, a bedrock well must end in a potentially contaminated fracture zone. The location of surface water bodies and the geometry of the fracture zones at the WRA dictate that a bedrock well must be drilled into LD1 in order to penetrate the nuclide plume (Figures 4-3 and 4-4). A well intersecting LD1 would modify the subsurface flow patterns, accelerate the rate of flow up the fracture between the vault and the well, and siphon off some nuclides from the Boggy Creek south discharge zone. Well demand and well depth can reduce the area of related discharge zones (Davison et al. 1994b). On the other hand, an overburden well is supplied mostly by water drawn in from near-surface layers, and would have little effect on groundwater flow or discharge patterns. To intersect the plume, the well must be located on the southeast side of Boggy Lake within a few hundred metres of the shoreline. We use the most conservative location of the well, which places it within the present boundary of Boggy Lake (Figure 4-4). Lake drainage or other changes could make this area suitable for human occupation and farming at some future time.

The MOTIF code predicts the area of each groundwater discharge zone, as well as its location (Davison et al. 1994b). In the absence of a well, the area of the Boggy Creek south discharge zone is about 2.9×10^5 m^2. This area decreases as the well depth and well demand increase, and groundwater that would have discharged into Boggy Lake is drawn up the well. The areas of the Boggy Creek north and Pinawa Channel discharge zones are predicted to be about 7.5×10^4 m^2 and 1.9×10^5 m^2 respectively. These areas are independent of the well because they are not directly connected to fracture zone LD1, the source of water for the well. The area of arable terrestrial discharge associated with each discharge zone is assumed to be given by

$$A_{TD} = \delta \cdot A_{DZ} \tag{4.1}$$

where A_{TD} is the area of arable terrestrial discharge (m^2),

A_{DZ} is the total area of the discharge zone (m^2), and

δ is the terrestrial fraction of the total area of the discharge zone (unitless).

A_{TD} and A_{DZ} apply to each of the three discharge zones separately, and δ is treated as a probabilistic parameter sampled once during each BIOTRAC simulation (Section 4.5.1).

4.3 MATCHING THE GENERIC BIOSPHERE TO THE SITE-SPECIFIC GEOSPHERE

In SYVAC3, the predictions of the site-specific geosphere model drive the generic biosphere model. For the most part, this is accomplished smoothly, since the assumptions made and the parameter values adopted in one model have no direct effect on the other model. In the case of hydrology, however, the biosphere and geosphere are closely coupled, and assumptions need to be consistent between the models. The potential for inconsistency is particularly high where the models hold parameters in common. Where such parameters appear in the geosphere model, they take on values that describe conditions as they presently exist at the WRA. Where they appear in the biosphere model, they are distributed to account for the variety of conditions that could occur over the course of time at a generic Canadian Shield location. This creates the possibility that the same parameter could be assigned different values in different parts of the system model. Here we discuss how the generic biosphere model is linked to the site-specific geosphere model in a consistent manner.

The movement of nuclides in the geosphere is controlled by the groundwater flow regime, which is driven by the hydraulic head distribution. The hydraulic head boundary conditions used in the geosphere model reflect the current hydrological conditions at the WRA. They are controlled by the ground surface topography, the elevations of local water bodies, the amount of water infiltrating the soil and the rock, and regional groundwater flow components. Assumptions made in BIOTRAC concerning any of these parameters must be consistent with the values implicit in the geosphere model.

Catchment area is one of the parameters of the surface water model (Section 5.5.1). A change in catchment area implies a change in the location of local heights of land and an associated change in hydraulic head distribution. Therefore, to maintain consistency with the boundary conditions used in the geosphere model, the catchment area used in BIOTRAC has been fixed at 1.06×10^8 m^2, the area of the Boggy Creek watershed.

Water body elevations do not appear explicitly in BIOTRAC, but are implicit in the values adopted for the depth and area of the lake. Lake depth and area are distributed parameters in BIOTRAC (Sections 5.5.2 and 5.5.3). Consistency with the geosphere model is maintained by assuming that changes in lake dimensions take place without affecting lake elevation significantly. Shallower or deeper lakes can be readily achieved through changes in the elevation of the lake bottom rather than the lake surface (Figure 4-5(a)). Similarly, smaller lakes are possible with only minor decreases in surface elevation because the existing lake is quite shallow (Figure 4-5(b)). On the other hand, larger lakes cannot be achieved without significant increases in elevation because of the relatively steep topography of the surrounding land (Figure 4-5(c)). Accordingly, lake area

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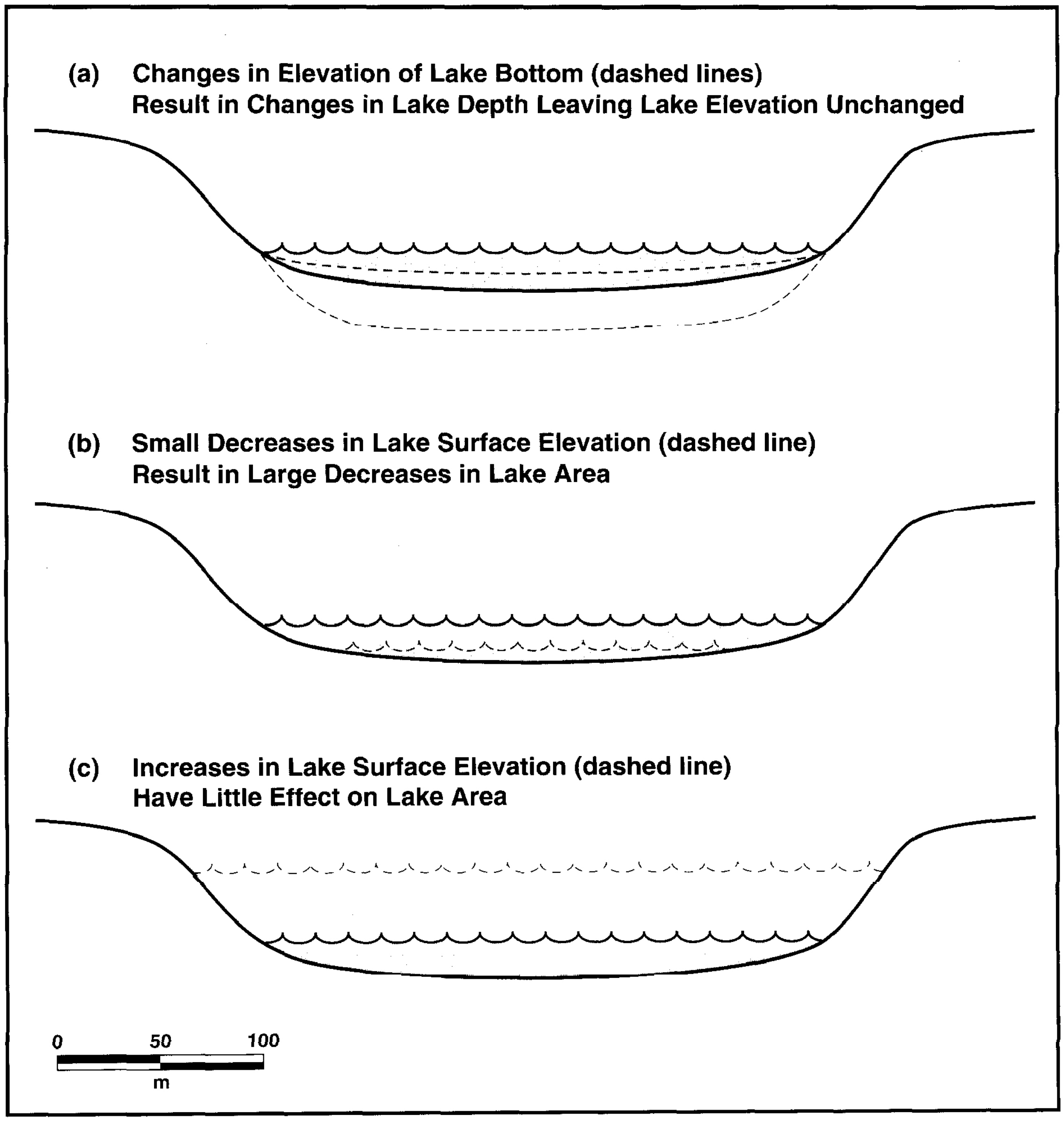
(a) Changes in Elevation of Lake Bottom (dashed lines) Result in Changes in Lake Depth Leaving Lake Elevation Unchanged

(b) Small Decreases in Lake Surface Elevation (dashed line) Result in Large Decreases in Lake Area

(c) Increases in Lake Surface Elevation (dashed line) Have Little Effect on Lake Area

FIGURE 4-5: Effects of Elevation Changes on the Lake. The present profiles are shown by solid lines. Vertical scale is exaggerated.

in BIOTRAC is distributed, but is truncated to 80 ha at its upper end because that is the approximate present area of Boggy Lake.

Because the discharge zone areas calculated by MOTIF assume a lake area equal to the present area of Boggy Lake, the calculated discharge areas may exceed the sampled lake area in some simulations. In these cases, part of the excess is interpreted as the arable land associated with terrestrial discharge (Section 4.2). Any remaining excess area is assumed to consist of wetland on the margins of the lake. Nuclides discharging to this wetland are assumed to contribute to the contaminant load in the lake proper, but otherwise are not accessed by the critical group and other organisms. Wetlands on the Canadian Shield tend to take the form of peat bogs, which are not heavily used by humans. Water from a bog would likely be considered unsuitable for drinking, swimming, irrigation or watering livestock. Moreover, bogs support little aquatic life that the critical group could directly include in its diet, although bogs are ecologically important for a variety of plant and animal species.

Wetlands may be drained for agricultural use, in which case the concentration in the residue would be similar to that in lake sediments, which we model explicitly. Wetlands may also be used as a source of peat for home heating. In BIOTRAC, however, we assume that peat is available from organic soils, which have a similar nuclide concentration in their lower layers as the organic material in a wetland. The exposure pathways unique to wetland discharge, which may occur when the discharge area exceeds the sampled lake area, are therefore implicitly incorporated into the model.

Changes in lake area may affect the location of the groundwater discharge zones if they are accompanied by a change in elevation of the lake surface. As noted in Section 4.2, the contaminant plume in fracture LD1 escapes the fracture and moves vertically upward at the point where the fracture passes under Boggy Lake. As the lake area decreases, this point will occur closer to the surface (Figure 4-6), and the plume will continue farther up the fracture before breaking away. The physical location of the Boggy Creek south discharge zone (Figure 4-4) would likely change, but its position relative to the shoreline would remain more or less the same. The locations of the other two discharge zones are likely to be unaffected because one occurs near the central channel of Boggy Lake and the other occurs outside of Boggy Lake altogether.

Hydraulic head distributions in the geosphere beneath soil-covered areas are determined to a limited extent by the amount of water infiltrating downward through the soil to the water table. Infiltration through soil-covered areas depends on water balance parameters (precipitation, runoff and evapotranspiration), soil type and soil depth, but the dependence is weak. Changes in precipitation, for example, would largely be counter balanced by runoff or evapotranspiration, and leave infiltration relatively unaffected. Furthermore, a uniform change in infiltration across a watershed would produce a uniform change in the elevation of the water table, but no change in hydraulic gradients. Accordingly, values for the infiltration parameters used in BIOTRAC may differ from those at the WRA without significantly affecting the hydraulic boundary conditions of the geosphere model. Values for soil type (Section 6.5.1.1), soil depth (Section 6.5.1.2) and the water balance parameters (Section 9.1.3) are therefore fully distributed in BIOTRAC.

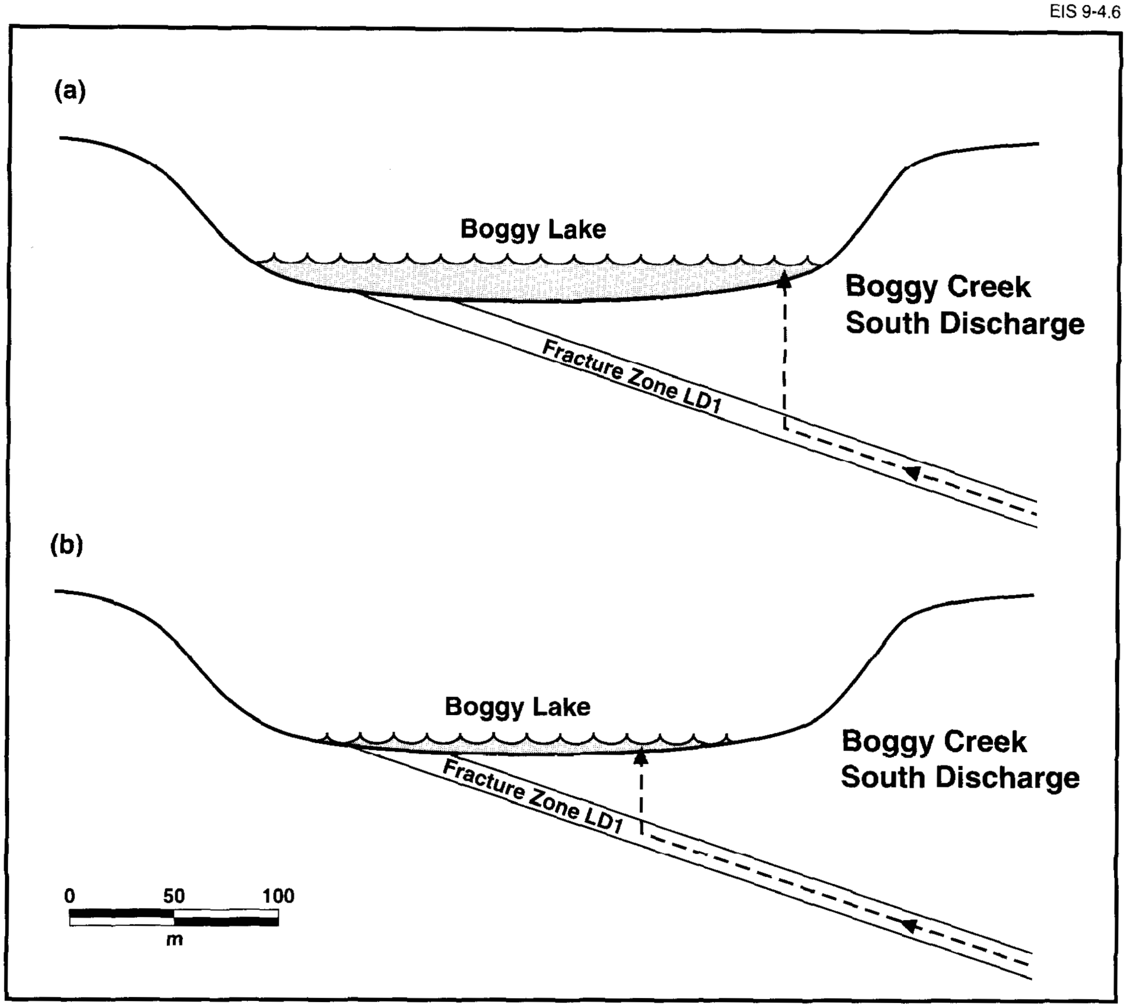

FIGURE 4-6: Effect of Lake Area on Discharge Location. (a) Present configuration of Boggy Lake in the Whiteshell Research Area. (b) Hypothetical discharge for a lake of smaller area. Vertical scale is exaggerated ten times.

For a well to intersect the contaminant plume, it must be located on the southeast side of Boggy Lake within a few hundred metres of the shoreline (Section 4.2). Although this position is dictated by the features of the WRA site, it presents no restrictions when viewed from a generic perspective. Fracture zone LD1 can supply up to 6 x 10^4 m^3 of water per year, which is more than enough to satisfy all the needs of the household making up the critical group at any given time (Section 9.1.1.4). The fracture geometry at the WRA does, however, place a constraint on well depth. Wells drilled to intersect LD1 at depths greater than 200 m would first encounter LD2 (Figure 4-3). Since LD2 is a reliable source of water, in practice drilling would stop there. Accordingly, wells in the model are restricted to depths less than 200 m. This constraint eliminates very few wells

(Davison et al. 1994b); deeper wells are rarely drilled because of their cost and the high mineral content of their water.

The generic interpretation of the discharge watershed used in BIOTRAC is shown in Figure 4-7. The watershed has the same area, shape and topography as the Boggy Creek watershed at the WRA to ensure consistency with the surface topography and hydrology of the geosphere model. It contains a lake of variable depth, and variable area less than 80 ha. Any nuclides that escape the geosphere discharge into this water body. All the discharge zones have an associated area of terrestrial discharge. When a well is present, it has a depth less than 200 m and is positioned so that it intersects the contaminated groundwater plume. The watershed is completely generic in all other respects.

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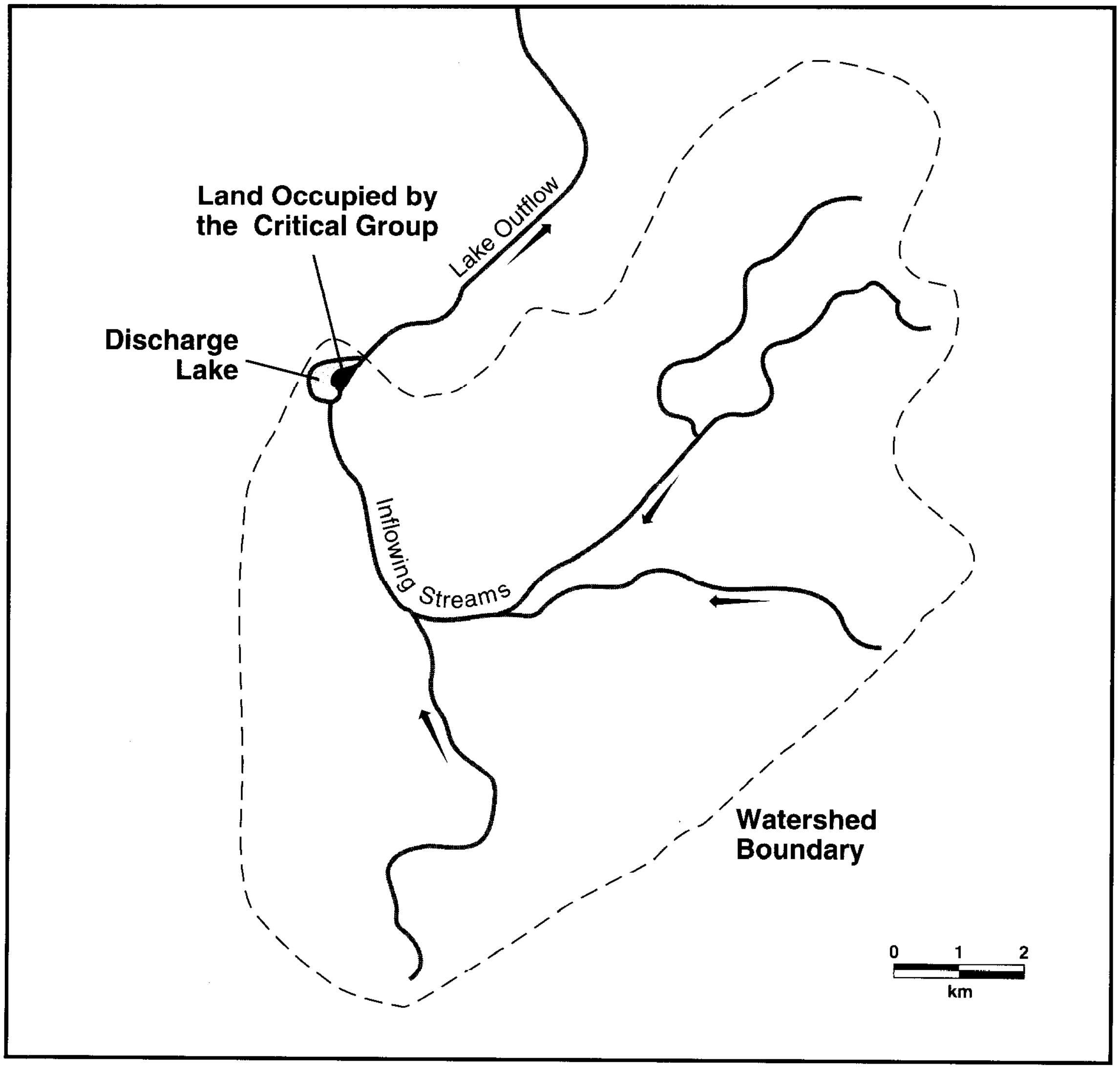


FIGURE 4-7: Generic Interpretation of the Boggy Creek Watershed

4.4 THE INTERFACE MODEL

As noted in Section 2.5.4, daughters in secular equilibrium and with half-lives greater than one day are separated from their precursors as they exit the geosphere, and are treated individually in the biosphere thereafter. Mass flow rates, χ^d (mol·a^{-1}), out of the geosphere for such secular-equilibrium daughters are calculated from (Davison et al. 1994b)

$$\chi^d = \chi^p \cdot (\lambda^p/\lambda^d) \cdot (RE^p/RE^d) \qquad (4.2)$$

where χ^p (mol·a^{-1}) is the mass flow rate of the precursor, λ^p and λ^d (a^{-1}) are the radioactive decay constants of the precursor and daughter, and RE^p and RE^d (unitless) are the corresponding retardation factors for the GEONET segment interfacing with the biosphere. If $RE^p/RE^d = 1$, the activity in the flow of the daughter equals the activity in the flow of the precursor. Where the sorptive behaviour of the precursor and daughter differ, the activities will not be equal. For example, if the precursor is more strongly bound than the daughter, the activity of the daughter in the aqueous phase will exceed that of the precursor, and χ^d will be greater than χ^p.

Models are required for the interfaces between the geosphere and sediment, surface water, soil and well. These models are discussed in turn below.

4.4.1 Geosphere/Sediment Interface

Sediment concentrations are required because sediments may serve as the substrate on which crops and other plant communities are growing (Section 3.3.2.1). Nuclides in the shallow, mixed sediment layer are assumed to come entirely from the water column. Concentrations in this layer are calculated using the mass balance approach described in Section 5.3.2. Concentrations in the compacted-sediment layer outside the zone of groundwater discharge are assumed to be the same as the mixed-sediment concentrations. Nuclides in the compacted layer within the region of groundwater discharge are assumed to arise entirely through sorption from upward-moving groundwater. Concentrations in this layer are calculated using the methods described below. Since compacted sediments are simply mixed sediments that have been buried, the compacted-sediment layer will contain some nuclides that originated in the mixed sediments. However, we do not consider this contribution in calculating the compacted-sediment concentration since in most cases it is expected to be very much less than the contribution from sorption from the upward-moving groundwater. A depth-weighted average of the concentrations in the two layers is taken to give an effective sediment concentration in the top 30 cm, which is assumed to be the depth of the root zone for terrestrial plants (Section 6.1). Concentrations at greater depths are not calculated because we assume that sediments below 30 cm are no longer accessed by the critical group and other biota.

Compacted-sediment concentrations for nuclide i are calculated for each of the three discharge zones in the following way. Pore-water concentrations, C^i_{pw} (mol·m^{-3} water), at the top of the compacted-sediment layer are first computed from

$$C^i_{pw}(t) = \chi^i_{ds}(t)/Fw_{ds} \quad . \qquad (4.3)$$

Here $\chi^i_{ds}(t)$ is the time-dependent mass flow (mol·a^{-1}) of nuclide i out of the compacted sediment layer and Fw_{ds} is the volumetric flow of water (m^3 water·a^{-1}) out of that layer. Both χ^i_{ds} and Fw_{ds} are GEONET outputs for each of the three discharge zones separately, as described in Davison et al. (1994b).

In Equation (4.3), we assume that the nuclide flow out of the geosphere is advection-dominated. This is likely to be the case only over relatively restricted areas of the total discharge zone, A_{DZ}, where groundwater velocities are high. In the remaining areas, the flow is diffusion-dominated and Equation (4.3) overestimates the pore-water concentrations. However, we assume conservatively that the flow is everywhere advection-dominated, and use Equation (4.3) to calculate pore-water concentrations over the entire discharge zone.

Nuclides will be present on the solid phase of the compacted sediments as well as in the liquid phase. We model the sorption process using the sediment solid/liquid partition coefficient, Kd^i_{ds}, defined as the ratio of nuclide concentration on compacted sediment solids to that in the pore water. The use of partition coefficients implies that sorption is a reversible process, and that an equilibrium between solid and liquid phases is reached instantaneously. From the definition of the partition coefficient,

$$C^i_{sol} = Kd^i_{ds} \cdot C^i_{pw} \tag{4.4}$$

where C^i_{sol} is the concentration of nuclide i on sediment solids (mol·kg^{-1} dry sediment), and

Kd^i_{ds} is the compacted sediment solid/liquid partition coefficient (m^3 water·kg^{-1} dry sediment) for nuclide i.

The total nuclide mass M^i_T (mol) in a compacted sediment layer of depth Z (m) and area A_{DZ} (m^2) is given by the sum of the masses in the solid and liquid phases:

$$M^i_T = (p_{ds} \cdot C^i_{pw} + \rho_{ds} \cdot C^i_{sol}) \cdot A_{DZ} \cdot Z \tag{4.5}$$

which, when combined with Equation (4.4), becomes

$$M^i_T = (p_{ds} + \rho_{ds} \cdot Kd^i_{ds}) \cdot A_{DZ} \cdot Z \cdot C_{pw} \quad . \tag{4.6}$$

Here, p_{ds} is the porosity of the compacted sediments (unitless), and ρ_{ds} is their bulk density (kg dry sediment·m^{-3} sediment). The total compacted-sediment concentration, C^i_{ds} (mol·kg^{-1} dry sediment), reflecting the contributions from both solid and liquid phases, is given by

$$C^i_{ds} = \frac{M^i_T}{\rho_{ds} \cdot A_{DZ} \cdot Z} = (p_{ds}/\rho_{ds} + Kd^i_{ds}) \cdot C^i_{pw} \quad . \tag{4.7}$$

Concentrations are calculated in this way for each of the three aquatic discharge locations, using the appropriate water flow rates and nuclide fluxes. The way in which sediment concentrations are used in BIOTRAC is discussed in Section 6.3.7.3.

The compacted-sediment concentration calculated using Equation (4.7) is an average over the area of discharge. In reality, the concentrations would vary substantially in space because groundwater flows to a given discharge zone are rarely uniform. However, to be used for farming, the sediments would have to be spread over a few hectares, which is the area needed to grow the crops to support a family and their livestock (Section 9.1.1.3). Average concentrations may then be appropriate to describe the mixing that would occur if the sediments were dredged, or the random positioning of the agricultural fields on exposed but undisturbed sediment.

Equation (4.7) provides sediment concentrations at the top of the compacted sediment layer. This concentration will be lower than at any other depth in the layer, because nuclides are lost through radioactive decay and sorption processes as the plume moves upward through the geosphere. C^i_{ds} will therefore underestimate the average concentration over the top 30 cm or so of compacted sediment. The underestimate will be largest for radionuclides with short half-lives or high Kd^i_{ds} values. But in no case will the underestimate be significant because the 30-cm depth is very small compared with the total travel distance through the geosphere.

4.4.2 Geosphere/Surface Water Interface

The input to the surface water model (Section 5.3) is the mass flow rate of nuclides discharging from the geosphere, χ^i ($mol \cdot a^{-1}$), which is the sum of nuclides discharging from the compacted sediments, χ^i_{ds} ($mol \cdot a^{-1}$), and the nuclides discharged via the well, χ^i_w ($mol \cdot a^{-1}$). Here we assume that nuclides pass through the mixed sediments without losses from sorption. Since χ^i_{ds} is one of the outputs of GEONET, no special interface model is required for groundwater discharge to the lake. The surface water model is driven by the total nuclide flow, including the flow through the well and through the three discharge zones (Section 5.2). Although some nuclides emerge into the Pinawa Channel and may bypass Boggy Lake (Figure 4-4), we make the conservative assumption that they all contribute to the nuclide load in Boggy Lake. We also assume that nuclides discharged via the well are simultaneously discharged to the lake. In this way, we implicitly account for runoff and erosion losses of nuclides initially deposited on soil with irrigation water, and for discharge of domestic water into the lake (Section 5.2).

4.4.3 Geosphere/Soil Interface

In each simulation, we assume that a terrestrial area is associated with each aquatic area of the three discharge zones, and that discharge to both types of areas occurs simultaneously. The soil model is driven by the nuclide concentration in the water that contacts the soil (Section 6.3.1.2). In the case of an underground source, the input to the model is the porewater concentration in the bottom or fourth soil layer (Section 6.1). The input is obtained for nuclide i at a given discharge zone through the use of a mass balance equation for the bottom soil layer:

$$\frac{dM^i(t)}{dt} = \chi^i_{ob}(t) + \lambda^{i-1} \cdot M^{i-1}(t) - Fw_{ob} \cdot C^i_{pw}(t) - \lambda^i \cdot M^i(t) \qquad (4.8)$$

where $M^i(t)$ is the mass (mol) of nuclide i in the bottom soil layer at time t (a),

$M^{i-1}(t)$ is the mass of the precursor to nuclide i in the bottom soil layer (mol),

$\chi^i_{ob}(t)$ is the mass flow rate of nuclide i out of the overburden ($mol \cdot a^{-1}$),

Fw_{ob} is the volumetric flow of water out of the overburden (m^3 water$\cdot a^{-1}$),

$C^i_{pw}(t)$ is the pore-water concentration in the bottom soil layer ($mol \cdot m^{-3}$ water), and

λ^i is the radioactive decay constant of nuclide i (a^{-1}).

Equation (4.8) states that the amount of nuclide in the bottom soil layer changes with time in response to the amount that enters the layer and the amount lost from it. The input term $\chi^i_{ob}(t)$ is an output of the geosphere model (Davison et al. 1994b). The second term on the right is also a source term that describes the ingrowth of nuclide i as a result of the radioactive decay of its precursor. Since radionuclide chains are treated member by member (Section 2.6), a value for $M^{i-1}(t)$ is available from previous calculations. This term would not appear in the equation if nuclide i headed a chain, or if it were the only member of a chain. The last two terms in Equation (4.8) describe the loss of nuclides from the layer by advection and radiological decay respectively. We assume that the water content of the layer remains constant so that the rate of advection loss is controlled by the rate of water flow into the layer, Fw_{ob}, which is also available as a geosphere output. In writing Equation (4.8), we assume uniform, instantaneous mixing of nuclides in the bottom layer.

The relationship between nuclide mass in the layer and pore-water concentration depends upon the sorptive behaviour of the nuclide. We model the sorption process in soil in the same way as in compacted sediments using solid/liquid partition coefficients. For an unsaturated medium, the total mass in the layer is given by

$$M^i = (\theta + \rho_s \cdot Kd^i) \cdot A_{TD} \cdot Z_4 \cdot C^i_{pw} \qquad (4.9)$$

where θ is the volumetric water content of the soil (m^3 water$\cdot m^{-3}$ soil),

ρ_s is the bulk density of the soil (kg dry soil$\cdot m^{-3}$ soil),

Kd^i is the soil solid/liquid partition coefficient for nuclide i (m^3 water$\cdot kg^{-1}$ dry soil),

A_{TD} is the area of the bottom soil layer subject to contamination (m^2), and

Z_4 is the depth of the bottom or fourth soil layer (m).

Combining Equations (4.8) and (4.9), we obtain

$$\frac{dC^i_{pw}(t)}{dt} = \frac{X^i_{ob}(t)}{\gamma^i \cdot V_{TD}} + \frac{\lambda^{i-1} \cdot \gamma^{i-1} \cdot C^{i-1}_{pw}(t)}{\gamma^i} - C^i_{pw}(t) \cdot \left[\frac{Fw_{ob}}{\gamma^i \cdot V_{TD}} + \lambda^i\right] \qquad (4.10)$$

where $\gamma^i = \theta + \rho_s \cdot Kd^i$ (4.11)

and $V_{TD} = A_{TD} \cdot Z_4$ (4.12)

which is the volume (m^3 soil) of the bottom soil layer subject to terrestrial discharge. Note that γ^i is a derived quantity. Equation (4.10) is subject to the initial condition $C^i_{pw}(0) = 0$. Using the theory developed in Section (2.6), we deduce that the impulse response function for the bottom or fourth soil layer is

$$RF^i_4(t) = \exp\left[-\left(\frac{Fw_{ob}}{\gamma^i \cdot V_{TD}} + \lambda^i\right) \cdot t\right] \qquad (4.13)$$

and that the time-dependent pore-water concentration is

$$C^i_{pw}(t) = \int_0^t \left[\frac{X^i_{ob}(t')}{\gamma^i \cdot V_{TD}} + \frac{\lambda^{i-1} \cdot \gamma^{i-1} \cdot C^{i-1}_{pw}(t')}{\gamma^i}\right] \cdot \exp\left[-\left(\frac{Fw_{ob}}{\gamma^i \cdot V_{TD}} + \lambda^i\right) \cdot (t - t')\right] dt' \quad . \qquad (4.14)$$

Equation (4.14) provides the pore-water concentrations required to drive the soil model for groundwater contamination. Pore-water concentrations are calculated for each of the three discharge zones separately, using the appropriate flow rates.

The concentration calculated in Equation (4.14) is based on the average nuclide flux over the entire discharge zone, A_{DZ}. As such, it will probably overestimate the concentration at the edge of the zone. Since this is where terrestrial discharge is most likely to occur, this approach yields a conservative result.

In deriving Equation (4.14), we assumed that the only water available for diluting the emerging nuclides is the water moving vertically upwards from the geosphere through the overburden. In fact, water draining downward through the soil profile or moving laterally as subsurface runoff may enhance the dilution. It is difficult to define volumes for these flows in a generic biosphere. Since they are expected to be small compared with

Fw_{ob}, they have been ignored in calculating C^i_{pw}. Groundwater nuclide concentrations will therefore be overestimated and the resulting soil nuclide concentrations will be conservatively high.

Equation (4.14) can be greatly simplified if the nuclide flow $\chi^i_{ob}(t)$ can be considered constant and if nuclide i has no precursor. Under these conditions the equation becomes

$$C^i_{pw}(t) = \frac{\chi^i_{ob}}{\gamma^i \cdot V_{TD}} \int_0^t \exp\left[-\left(\frac{Fw_{ob}}{\gamma^i \cdot V_{TD}} + \lambda^i\right) \cdot (t - t')\right] dt' \qquad (4.15)$$

which can be integrated directly to give

$$C^i_{pw}(t) = \frac{\chi^i_{ob}}{Fw_{ob} + \lambda^i \cdot \gamma^i \cdot V_{TD}} \cdot \left\{1 - \exp\left[-\left(\frac{Fw_{ob}}{\gamma^i \cdot V_{TD}} + \lambda^i\right) \cdot t\right]\right\} \quad . \qquad (4.16)$$

For sufficiently large values of t (~ 2 x 10^4 a for typical values of Fw_{ob}, γ^i and V_{TD}), the exponential term in Equation (4.16) becomes small compared with unity, and the steady-state pore-water concentration is given by

$$C^i_{pw} = \frac{\chi^i_{ob}}{Fw_{ob} + \lambda^i \cdot \gamma^i \cdot V_{TD}} \quad . \qquad (4.17)$$

Equation (4.17) provides a very simple, easily interpreted expression for the steady-state pore-water concentration in the bottom soil layer when χ^i_{ob} can be considered time-independent. Although the time to steady state is long relative to the assessment period of 10 000 a (AECB 1987), Equation (4.17) supplies an analytical solution that can be used to verify the numerical integration of Equation (4.14), which is the expression actually used in the model to calculate C^i_{pw}.

4.4.4 Geosphere/Well Interface

Domestic water supply wells are usually drilled only as deep as necessary to provide the amount and quality of water needed. If an overburden aquifer can meet the requirements, the well will not penetrate the bedrock. If there is insufficient quantity or quality of water in the overburden, the well may be extended into the bedrock until an adequate water supply is encountered. In practice, this means that wells in bedrock normally end in a fracture zone.

This practice is reflected in our well depth distribution, which is based on the depth, D_w (m), of wells on the Shield (Davison et al. 1994b). D_w is distributed lognormally with a GM of 37.2 m and a GSD of 2.2, and is truncated at a maximum depth of 200 m (Section 4.3). If a well is chosen as the water source for the critical group (Section 9.1.2), we determine its type by comparing the sampled value of D_w with the overburden depth, D_{ob} (m), at the Boggy Creek south discharge zone calculated from distributed parameters in the geosphere model (Davison et al. 1994b). If $D_w \leq D_{ob}$, we assume the well is an overburden well; if $D_w > D_{ob}$, we assume a bedrock well.

We have no explicit model to predict nuclide concentrations that might occur in water from an overburden well. Since overburden wells are supplied mostly by groundwater drawn from near-surface aquifers, we assume in this case that nuclide concentration in well water equals the concentration predicted for the lake (Section 5.3.1). On the other hand, concentrations in water from a bedrock well depend on the details of groundwater and contaminant transport in the geosphere.

This well water nuclide concentration, C^i_{ww} (mol·m^{-3} water), is calculated in BIOTRAC with the help of input from GEONET, which incorporates a detailed well model. The concentration is given by

$$C^i_{ww} = (\chi^i_w + C^i_l \cdot V_{ld})/W_w \qquad (4.18)$$

where χ^i_w is the mass flow rate of nuclide i out of the geosphere into the well (mol·a^{-1}),

C^i_l is the nuclide concentration in the lake water (mol·m^{-3} water),

V_{ld} is the volume of water from the lake drawn into the bedrock well (m^3 water·a^{-1}), and

W_w is the well demand (m^3 water·a^{-1}).

Here χ^i_w and V_{ld} are provided by GEONET (Davison et al. 1994b), C^i_l is calculated by the surface water model (Section 5.3.1) and W_w is calculated by BIOTRAC, as shown in Section 9.1.2. Note that V_{ld} is set to zero if W_w is less than a critical demand value, Q_{crt}, established by GEONET.

In the bedrock well model we assume that nuclides discharged from the geosphere, χ^i_w, are diluted by surface water. Potentially contaminated surface water may be drawn from the lake into the well, depending on the well demand, W_w. Thus, W_w is passed from BIOTRAC to GEONET to allow the calculation of V_{ld}. Nuclides are always assumed to be distributed uniformly in the water drawn from the well. GEONET establishes a maximum well capacity, Q_{cap}, which is used in BIOTRAC to establish the water sources (Section 9.1.2).

4.5 INTERFACE PARAMETERS

The parameters required in the interface models appear in Equations (4.1) to (4.18). GEONET calculates the nuclide and groundwater flows out of the geosphere, and the size of the discharge zones, and so these flows and discharge zones are not discussed further here. However, we have listed the names and symbols of the BIOTRAC parameters supplied by the geosphere model in Table 4-1 together with the corresponding parameter symbols from that model. Most of the properties of the compacted-sediment layer (retardation factors, porosity and solid/liquid partition coefficients) are reported by Bird et al. (1992) and summarized by Davison et al. (1994b), who also discusses the well depth and the volume of lake water drawn into the well. Section 9.1 documents the remaining well parameters, including demand and the probability of a well as the water source. Soil properties (water content, density, solid/ liquid partition coefficients and depth)

TABLE 4-1

GEOSPHERE/BIOSPHERE INTERFACE PARAMETER NAMES AND SYMBOLS FOR BIOTRAC AND THE CORRESPONDING SYMBOLS FOR THE GEOSPHERE MODEL

Interface Parameters	Biosphere Symbols	Geosphere Symbols
Retardation factor of parent	RE^p	R_q
Retardation factor of daughter	RE^d	R_q
Well capacity (m^3 water·a^{-1})	Q_{cap}	Q_{cap}
Critical well demand (m^3 water·a^{-1})	Q_{crt}	Q_{crt}
Well demand (m^3 water·a^{-1})	W_w	Q_{dem}
Flow of nuclide i into well (mol·a^{-1})	X_w^i	O_q
Surface water flow into well (m^3 water·a^{-1})	V_{ld}	Q_{sur}
Flow of nuclide i out of comp. sediment (mol·a^{-1})	X_{ds}^i	O_q
Water flow out of comp. sediment (m^3 water·a^{-1})	Fw_{ds}	Q_{dis}
Comp. sediment porosity (unitless)	P_{ds}	θ
Comp. sediment partition coefficient (m^3·kg^{-1})	Kd_{ds}^i	k_d
Flow of nuclide i from comp. sediment (mol·a^{-1})	X^i	O_q
Flow of nuclide i out of overburden (mol·a^{-1})	X_{ob}^i	O_q
Water flow out of overburden (m^3 water·a^{-1})	Fw_{ob}	Q_{dis}
Total area of discharge zone (m^2)	A_{DZ}	A'_{dis}

Note: In the geosphere model, all the nuclide flows are designated by O_q. Furthermore, Q_{dis} is equal to both Fw_{ds} and Fw_{ob} in the biosphere model because no water is gained or lost in the compacted sediment.

are discussed in Section 6.5. The only remaining parameters of the interface models are the fraction of the discharge zone underlying an arable terrestrial zone, δ, and compacted-sediment bulk density, ρ_{ds}, which are documented in the next sections.

4.5.1 Fraction of Discharge Zone Underlying an Arable Terrestrial Area, δ (unitless)

This parameter is used in Equation (4.1). There is little information available to define a generic value of the fraction of the discharge zone that underlies arable unsaturated soils. We assume that δ is uniformly distributed between 0.01 and 0.1, and that terrestrial discharge is associated with each of the three discharge zones in each model simulation. These assumptions likely overestimate the true frequency of occurrence of terrestrial discharge, but this means that terrestrial pathways, which are important in dose predictions, can be more fully considered.

4.5.2 Compacted-Sediment Bulk Density, ρ_{d_s} (kg dry sediment·m^{-3} sediment)

The bulk density of organic sediments varies little from lake to lake, and is not expected to change significantly over time. We have therefore adopted a fixed value of 1.25 x 10^2 kg dry sediment·m^{-3} sediment for ρ_{d_s}. This value is based on a particle density of 2.5 x 10^3 kg ·m^{-3} sediment (Evans J.E. et al. 1981, Durham and Joshi 1984), and a sediment porosity of 0.95 (El-Daoushy and Johansson 1983, Durham and Joshi 1984). Our ρ_{d_s} value is relatively small, but this is conservative because sorption is reduced in compacted sediment and the input of nuclides to the lake is enhanced (Sections 4.4.2 and 5.3.1).

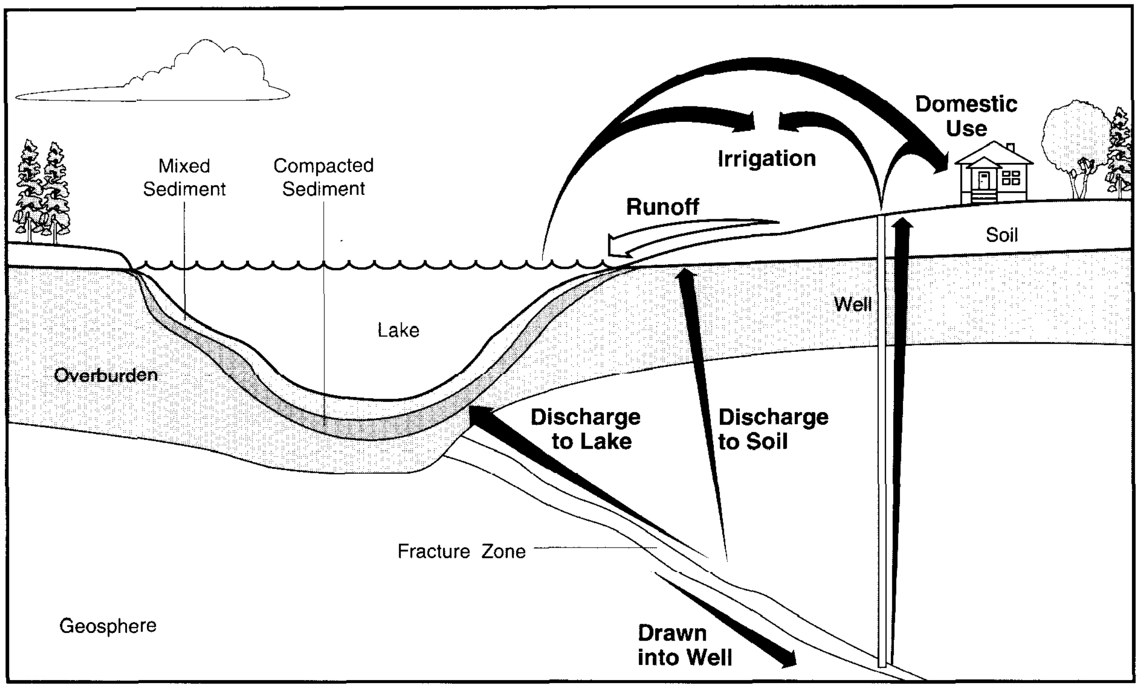
4.6 DISCUSSION AND ASSUMPTIONS

Our geosphere/biosphere interface models involve two closely related aspects: water flow and nuclide discharge (Figures 4-8 and 4-9). The water flows are used to calculate the concentrations of nuclides from the mass flows out of the geosphere. In Figure 4-8, we not only show the water flows associated with the geosphere/biosphere interface, but also those related to water usage because these flows are not independent (Section 4.4.4). We also indicate runoff, which is not modelled explicitly in BIOTRAC, but is assumed to occur instantaneously. This is why all the nuclides are allowed to discharge to the lake, unless they are sorbed on the way there (Figure 4-9). Thus, nuclides discharging to the well and soil can be thought of as being temporarily diverted from the lake.

Limited data are available on the discharge to the biosphere from deep groundwater flow systems. Few discharge zones have been identified or studied, and little quantitative information is available on their physical or chemical characteristics, or on discharge rates. However, methods have recently been developed for locating deep groundwater discharge zones (Lee 1985, Lee et al. 1991, Stephenson et al. 1992), and contaminant release through simulated zones is being studied in the laboratory.

With the limited observational data, we cannot demonstrate that our interface models are exact representations of the processes in question. Some of the assumptions that we have made are known to be approximations. For example, we do not explicitly model potential effects on nuclide behaviour and transport resulting from changes in redox conditions at the geosphere/biosphere interface. Nuclides released from the vault may pass from anaerobic geosphere to aerobic biosphere conditions as they reach the surface environment. This could lead to chemical precipitation and nuclide accumulation at the interface. Chemical precipitation is implicitly accounted for by the compacted-sediment solid/liquid partition coefficients, $Kd^i_{d_s}$, used in the geosphere model (Section 4.4.1). The coefficients measure the bulk partitioning of nuclides between soluble and solid phases, regardless of the mechanism leading to that partitioning. Thus, $Kd^i_{d_s}$ values implicitly include the processes of sorption and precipitation.

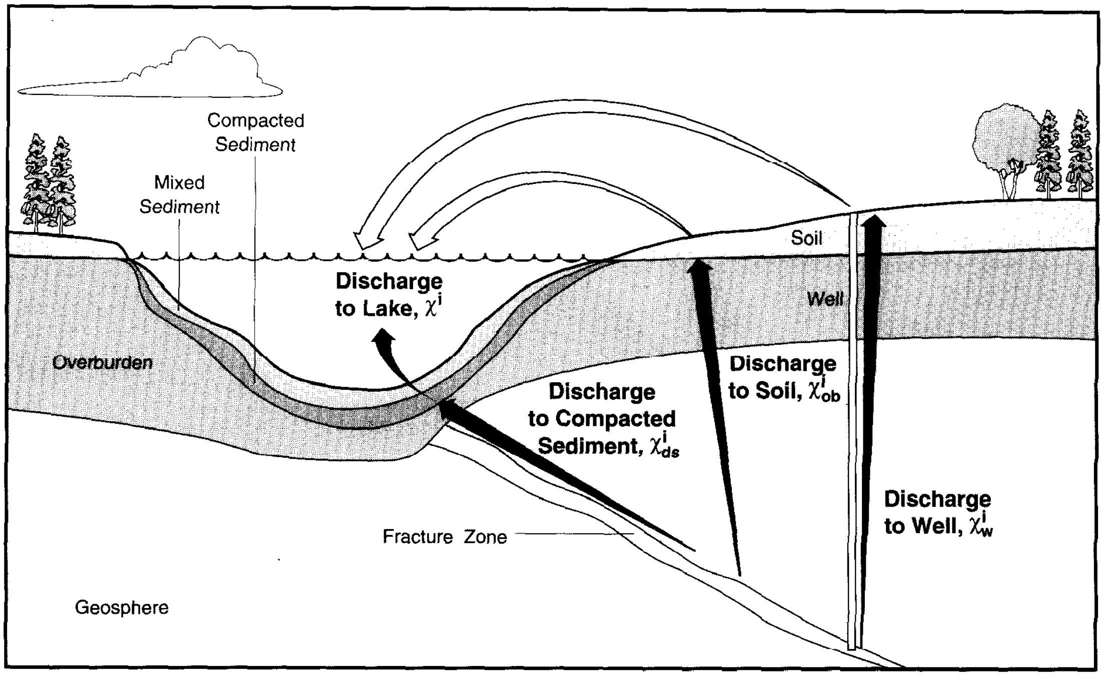
The most important radionuclides in the postclosure assessment are ^{14}C, ^{129}I and ^{99}Tc (Goodwin et al. 1994). None of them are expected to be subject to chemical precipitation. This is particularly true for ^{129}I and ^{99}Tc. Iodine will be present as reduced iodide (I$^-$) in the geosphere



EIS 9-4.8

FIGURE 4-8: Schematic Representation of Water Flows in the Biosphere Model and the Geosphere/Biosphere Interface. Water discharges from the geosphere to the lake through overburden and compacted and mixed sediments at three discharge zones (Figure 4-4). Water discharging to soil passes through overburden only. Water also passes into the bedrock well from the geosphere. Depending on demand, the well may draw in surface water directly from the lake. Domestic and irrigation needs may be satisfied by lake or well water. Surface and subsurface runoff is not modelled explicitly and is assumed to occur instantaneously, returning all the water used by the critical group to the lake. Closed arrows indicate explicit consideration, and open arrows indicate implicit consideration in BIOTRAC.

(Section 8.5.9.5). This form is relatively stable in the biosphere, and oxidation to iodate (IO_3^-) is slow; moreover, IO_3^- is only weakly retarded by geological and sedimentary materials. Technetium is strongly retarded in reduced geological or sedimentary environments, and this is reflected in our $Kd_{d_s}^{i}$ value. The most stable oxidized species of technetium is pertechnetate (TcO_4^-), which is highly soluble and unlikely to form precipitates. Carbon-14 is likely to be present as bicarbonate (HCO_3^-) (Section 8.5.10.4). Canadian Shield lake sediments generally contain few carbonate minerals because of the generally low dissolved inorganic carbon concentrations in the water. Thus, ^{14}C is unlikely to form precipitates at the geosphere/biosphere interface, although the inclusion of ^{14}C in trace calcium-carbonate or iron-carbonate mineralization at the interface cannot be ruled out. Some nuclides considered in the postclosure assessment (Table 1-1) may have the potential to form precipitates, but appreciable accumulations would be unlikely because these nuclides would reach the upper geosphere in



EIS 9-4.9

FIGURE 4-9: Schematic Representation of Geosphere/Biosphere Interface Showing Nuclide Discharges. Discharge to the well, χ^i_w, also goes to the lake. Discharge to soil, χ^i_{ob} is also included in the discharge to lake, χ^i_{ds}. The combined discharge to the lake, χ^i, includes well, soil and lake components for all three discharge zones shown in Figure 4-4. For concept assessment, only discharge from the Boggy Creek south zone is important. Closed arrows indicate explicit consideration, and open arrows indicate implicit consideration in BIOTRAC.

trace quantities only (Goodwin et al. 1994). For these reasons, our treatment of sorption and chemical precipitation at the geosphere/biosphere interface would likely not lead to underestimation of nuclide concentrations and doses in the biosphere.

Our model is not entirely conservative. The catchment area used lies toward the high end of the distribution of Canadian Shield catchments so that nuclides discharging to the lake are diluted by relatively large volumes of water. The calculated compacted-sediment concentrations apply to the top of the layer and underestimate the concentration at deeper points in the layer. Wells deeper than 200 m are not considered. However, all these assumptions are reasonable and, according to our own evaluations, will result in only small underestimates of environmental concentrations and doses.

On the other hand, many conservative assumptions have been built into the model to offset potential underestimations and uncertainties. We assume that sediments can be used for agricultural purposes. All nuclides discharging from the geosphere are assumed to enter a single lake. Part of each discharge zone is assumed to underlie a terrestrial area, thus contaminating the agricultural fields of the critical group and areas inhabited by terrestrial biota. When wells are used, they are located in a manner to intersect the maximum amount of nuclide that might be released from the vault. The consequences are overestimated in each computer simulation because of these assumptions.

On balance, we believe that our conservative assumptions more than compensate for any underestimates that could arise in the application of the interface models. We conclude that, given the conceptual model of the WRA geosphere and the predictions of GEONET, our models provide a reasonable description of nuclide behaviour at the interface, and result in an overestimate of environmental concentrations and doses to humans and other biota.

5. THE SURFACE WATER SUBMODEL

5.1 THE SURFACE WATER COMPARTMENT

Groundwater flow along fracture zones in plutonic rock typically reaches the earth's surface at topographic lows, which ordinarily are occupied by surface water bodies. A water body would therefore be the primary recipient of nuclides that have migrated through the geosphere from an underground disposal facility (Chapter 4). The subsequent behaviour of the nuclides in the water body will play a major role in determining the impact of the release on humans and the environment. The model developed to treat the surface water pathways for the postclosure assessment is summarized in this chapter. The information was extracted from the surface water submodel report (Bird et al. 1992), which contains additional details. The model and all its parameter values have also been published in the open literature (Bird et al. 1993).

In general, the discharge water body could be a lake, a river or a permanent wetland. For our assessment, we have chosen to model a lake. For a given drainage basin, concentrations of nuclides in water would be similar in both a river and a lake since the same volume of water would flow through each. We selected a lake because nuclides would be retained in lake sediments to a much greater degree than in river sediments, which are subject to scour, resuspension and downstream transport. A lake is the conservative choice because sediments may be used for agricultural purposes (Section 6.3.7.3). Similarly, a lake will likely result in higher doses than a permanent wetland because the number of potential exposure pathways is much higher for a lake (Section 4.3). Some of the exposure pathways unique to permanent wetlands (draining and peat use) are incorporated in BIOTRAC, and so our model includes such wetlands implicitly.

Because human population or collective dose estimates are not required for concept assessment, the surface water model can focus on the discharge lake alone. There is no need to consider the drainage system of which the lake is a part. The nuclides flushed to downstream lakes and rivers will be diluted by additional surface runoff. Individuals living downstream of the discharge lake will receive much lower doses than members of the critical group. This is also true for non-human organisms.

The lake considered in the surface water model is essentially generic. Because the surface water model interfaces directly with the site-specific geosphere model, the lake shows some features that are specific to the WRA (Section 4.3). In particular, the lake is located in a drainage basin with an area of 1.06 x 10^8 m^2, and the lake size is limited to less than 80 ha. In all other aspects the lake is generic, and has characteristics typical of Canadian Shield lakes. As noted in Section 2.1, Canadian Shield lakes are generally small, with a mean area of about 7 ha and a mean depth of about 5 m. Precipitation exceeds evapotranspiration, so that there is usually a discharge through the lake outlet. A small percentage of Canadian Shield lakes are seepage lakes having no surface outlet. Flushing rates are typically 0.1 to 10 a^{-1}. Processes such as productivity, water mixing and sedimentation vary strongly throughout the year.

The geochemistry of Canadian Shield lakes is more similar than their diverse geographic locations would suggest. Calcium, potassium and other chemical concentrations characterize these lakes as dilute systems with low alkalinity and little suspended particulate material in the water column (Armstrong and Schindler 1971, Nriagu et al. 1982). Suspended solids consist of both material produced within the lake and material washed in from the catchment. Approximately 50% of the particulate material is organic (Brunskill et al. 1971). The composition of the sediments reflects the nature of the material suspended in the overlying water. Usually sediment composition changes little with depth, although the high porosity (0.95 to 0.99) near the sediment/water interface decreases as the sediment is compacted by the continual particulate flux. The metazoan biota living in the water column and sediments comprise a small fraction (<10%) of the total organic material in the lake water (Birge and Juday 1927), and <1% of the organic material in the sediments (Wetzel 1975).

The similar characteristics of different Canadian Shield lakes and the relative stability of their properties over long periods of time (Robbins and Edgington 1975, Wetzel 1975) permit a generic lake to be defined in a meaningful way. Spatial and temporal variation as well as uncertainty in each model parameter can be combined in a single probability density function.

The objective of the surface water model is to estimate time-dependent nuclide concentrations in the water column of the discharge lake and in the mixed-sediment layer, given the flow of nuclides out of the geosphere. These concentrations are used directly to calculate internal doses to members of the critical group through water ingestion, and external doses arising from swimming, bathing, or exposure to beach sediments. The concentrations are also passed to the other three submodels where they are used as inputs to calculate nuclide concentrations in air, soil and the food chain. Furthermore, nuclide concentrations in lake water are also used to evaluate the radiological protection of the environment and the chemical protection of humans and the environment (Chapter 13, Amiro 1992a, Goodwin et al. 1994).

5.2 QUALITATIVE DESCRIPTION OF THE SURFACE WATER MODEL

A wide variety of models has been developed to simulate nuclide behaviour in lakes. These range from very simple, steady-state models that use bulk parameterizations to represent many processes (Cornett and Ophel 1986), to fully time-dependent formulations that treat a multitude of biogeochemical processes in detail (Onishi et al. 1976, 1981). The simple models are the most relevant for assessing the aquatic aspects of the concept for disposal of Canada's nuclear fuel waste. Complex models are difficult to generalize to different situations (Edgington 1981), and so are not appropriate for a generic application. Complex models require large amounts of computer time, and as such are not suitable for probabilistic assessments with repeated simulations. Moreover, the desired model outputs are annual average concentrations and doses, and so seasonal effects do not need to be modelled. Similarly, we do not explicitly model the evolution of the lake or the impact of human activities, such as major water diversions or hydroelectric impoundments, which are discussed by Goodwin et al. (in preparation) as part of the scenario analysis. The use of a simple lake model for postclosure assessment is therefore entirely appropriate. Comparisons of simple and complex models suggest that increased model complexity does not necessarily improve the accuracy or precision of the model predictions (Snodgrass and Dillon 1983).

Accordingly, the surface water model developed for postclosure assessment is a simple compartment model based on the mass balance equation. The model is made up of two compartments, one for water and one for recently deposited mixed sediment. This sediment compartment represents the top sediment layer, a few centimetres thick, which is continually mixed by the action of the water and the aquatic organisms. The deeper, compacted sediments are treated as part of the geosphere model (Section 4.4.1). In this form, the surface water model is similar to other models developed for similar purposes (e.g., Lerman and Taniguchi 1972, Wahlgren et al. 1980, Tracy and Prantl 1983). In particular, it is similar to models used internationally to assess the impact of nuclear fuel waste disposal (Bergström et al. 1982, Korhonen and Savolainen 1982, Lawson and Smith 1984).

The surface water model is driven by the nuclide flow out of the geosphere, which reaches the lake by a number of different routes (Figure 4-9). Part of the flow discharges directly to the lake; the remainder discharges initially to a terrestrial zone or to a well, but may eventually reach the lake indirectly via the soil and atmosphere. In the case of terrestrial discharge, the soil profile becomes contaminated by the upward movement of nuclides from the water table (Section 6.2.1). In reality, the nuclides may reach the lake in water that has run off over the contaminated soil. Alternatively, nuclides may be suspended from the soil to the atmosphere, and enter the lake from above via deposition. Similar pathways to the lake exist for nuclides discharged to the well since well water is used for aerial irrigation or is discarded following domestic use.

The indirect flow of nuclides to the lake from the soil and atmosphere is difficult to quantify. Runoff, suspension and deposition are complex, time-dependent processes that are strongly site-specific. They are difficult to model realistically when the topography of the site and the relative locations of the fields and the lake are unknown. Accordingly, we have chosen not to model these processes explicitly. Instead, we assume that all nuclides discharging to terrestrial zones and to the well reach the lake instantaneously (Figure 4-9). We also assume that nuclides removed from the lake with water used for irrigation or domestic purposes return to the lake instantaneously. This allows the transfer of nuclides from the soil and atmosphere to the lake to be modelled simply and conservatively. On the other hand, this assumption leads to an apparent generation of mass in the model and an overestimate of the nuclide load in the lake. The implications of this for the model predictions are discussed in Section 9.3.

Once in the water column, the nuclides are assumed to be uniformly and instantaneously mixed throughout the volume of the lake (Figure 5-1). They are diluted by the volume of water passing through the lake, and may subsequently be flushed out into downstream parts of the drainage system. Nuclides may also be lost from the water column in a variety of other ways. Volatile nuclides are allowed to degas to the atmosphere. Nuclides may sorb onto suspended material in the water column and be deposited on the lake bottom in the form of mixed sediment. Radioactive decay continuously removes radionuclides from the system, but some new nuclides are generated locally by the ingrowth of daughters.

A number of processes occurring within the lake are modelled only approximately or are omitted. The suspension of nuclides into the atmosphere as aerosols is not modelled when estimating nuclide concentrations in water.

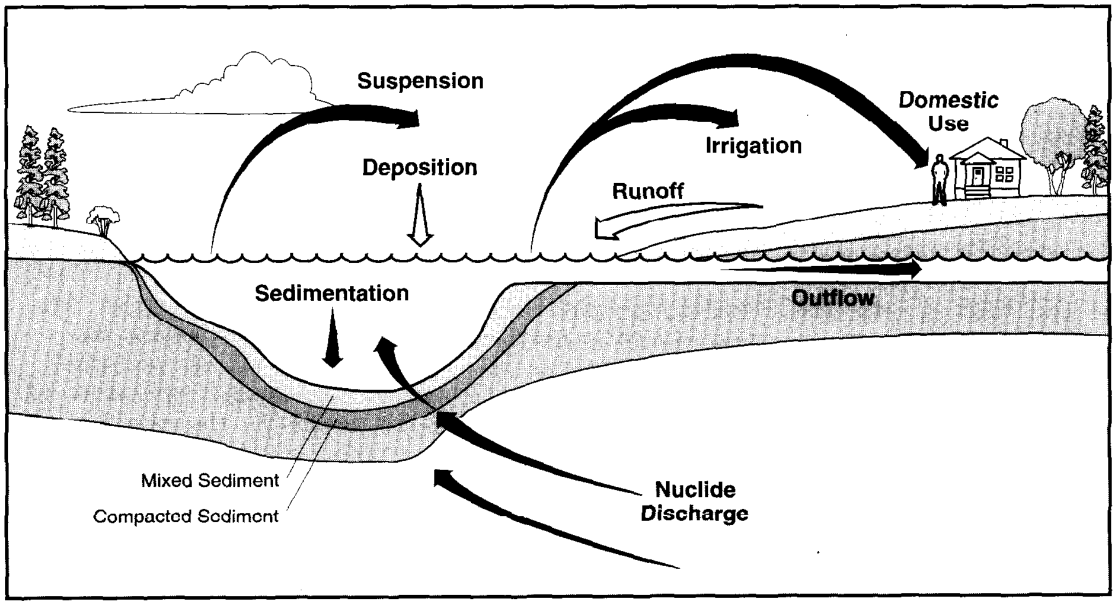


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FIGURE 5-1: Transport Processes in the Lake. Closed arrows indicate explicit consideration, and open arrows indicate implicit consideration in BIOTRAC.

Similarly, nuclide concentrations are calculated without allowing for losses to fish or aquatic plants. The chemical form of the nuclides is not considered explicitly, although a range of chemical behaviour is covered by the parameter distributions. The sediments are assumed to be immobile, and bioturbation and resuspension are considered only indirectly. As noted above, the lake is assumed not to evolve over time, and seasonal processes are not modelled. The effect of these assumptions on the model output is discussed in Section 5.7.

5.3 MATHEMATICAL FORMULATION OF THE SURFACE WATER MODEL

5.3.1 Nuclide Concentrations in Water

Because the transport processes in lake water and sediments occur relatively slowly, concentrations in these compartments also change slowly with time and must be calculated using a time-dependent model. Expressions for the water and sediment nuclide concentrations are therefore obtained using the response function/convolution approach discussed in Section 2.6.

The mass balance equation for the water compartment for a radionuclide i in a decay chain with precursor i-l is

$$\frac{dM_l^i(t)}{dt} = \chi^i(t) + \lambda^{i-1} \cdot M_l^{i-1}(t) - (f + \alpha^i + \eta_l^i + \lambda^i) \cdot M_l^i(t) \qquad (5.1)$$

where	$M_l^i(t)$	is the mass (mol) of nuclide i in the water column of the lake at time t (a),
	$\chi^i(t)$	is the total annual input of nuclide i to the lake ($mol \cdot a^{-1}$),
	λ^i	is the radioactive decay constant of nuclide i (a^{-1}),
	f	is the flushing rate of the lake (a^{-1}),
	α^i	is the rate constant describing the net rate of transfer of nuclide i from water to mixed sediments (a^{-1}), and
	η_l^i	is the rate constant describing the rate at which a volatile nuclide i is lost to the atmosphere by gaseous evasion (a^{-1}).

The input term $\chi^i(t)$ in Equation (5.1) is the output of the geosphere model. It is the sum of the nuclide flows over all three discharge zones and from the well, and implicitly includes fluxes from terrestrial areas to account for nuclides that reach the lake indirectly via runoff and atmospheric deposition. The second term on the right in Equation (5.1) is also a source term that describes the ingrowth of radionuclide i as a result of the decay of its precursor.

The four terms in parentheses on the right side of Equation (5.1) represent processes by which nuclides are lost from the lake water. In order, these are hydrological flushing through the lake outlet, deposition to the underlying mixed sediments, gaseous evasion (in the case of volatile nuclides) and radioactive decay. We assume that the water flow through the lake can be approximated by the product of the runoff, R (m water·a^{-1}), and the catchment area, A_d (m^2), so that the flushing rate is given by

$$f = R \cdot A_d / V_l \tag{5.2}$$

where V_l is the lake volume (m^3 water). The rate constant α^i in Equation (5.1) describes the net transfer of nuclides from the water to the mixed sediments, and so implicitly accounts for resuspension. The values assigned to the various parameters and to parameters appearing in subsequent equations, are discussed in Section 5.5.

Equation (5.1) is subject to the initial condition $M_l^i(0) = 0$. Comparing Equation (5.1) with Equation (2.2) in Section 2.6, and using Equations (2.4) and (5.2), we deduce that the impulse response function for the water column is

$$RF_l^i(t) = \exp\left[-\left(\frac{R \cdot A_d}{V_l} + \alpha^i + \eta_l^i + \lambda^i\right) \cdot t\right] \tag{5.3}$$

and that the time-dependent nuclide mass in the lake water is given by the convolution integral (e.g., Equation (2.3))

$$M_l^i(t) = \int_0^t \left[\chi^i(t') + \lambda^{i-1} \cdot M_l^{i-1}(t')\right] \cdot \exp\left[-\left(\frac{R \cdot A_d}{V_l} + \alpha^i + \eta_l^i + \lambda^i\right) \cdot (t - t')\right] dt' \quad . \tag{5.4}$$

If we assume that the nuclide load in the lake is uniformly and instantaneously mixed throughout the lake volume, then the nuclide concentration in the water column, $C_l^i(t)$ (mol·m^{-3} water), can be expressed as

$$C_l^i(t) = M_l^i(t)/V_l \quad . \tag{5.5}$$

Substituting Equation (5.5) into (5.4), and assuming the lake volume does not change with time, yields an equation for the time-dependent water concentration:

$$C_l^i(t) = \int_0^t \left(\frac{\chi^i(t')}{V_l} + \lambda^{i-1} \cdot C_l^{i-1}(t')\right) \cdot \exp\left[-\left(\frac{R \cdot A_d}{V_l} + \alpha^i + \eta_l^i + \lambda^i\right) \cdot (t - t')\right] dt' \quad . \tag{5.6}$$

This equation provides the concentration in water at a given time by considering the input function as an infinite sequence of impulse inputs at previous times and summing the contributions from each impulse.

5.3.2 Nuclide Concentrations in Mixed Sediment

Mixed-sediment concentrations are calculated in much the same way as water concentrations. The nuclide mass balance equation for the sediment compartment is

$$\frac{dM^i_{sed}(t)}{dt} = \alpha^i \cdot M^i_l(t) + \lambda^{i-1} \cdot M^{i-1}_{sed}(t) - \lambda^i \cdot M^i_{sed}(t) \tag{5.7}$$

where $M^i_{sed}(t)$ is the mass (mol) of nuclide in the sediments at time t (a), and all the other parameters are as defined above. We assume that the sediments become contaminated in two ways: through deposition from the water column (the first term on the right in Equation (5.7)), and through ingrowth following the decay of a precursor (the second term on the right). Nuclides in the upward-moving groundwater are assumed not to sorb to the mixed sediments. The only way in which nuclides are lost from the mixed sediments is through radioactive decay. Mixed sediments are not explicitly transformed into compacted sediments in BIOTRAC.

Using Equation (5.7) and the framework presented in Section 2.6, we deduce that the impulse response function for the mixed sediment compartment is

$$RF^i_{sed}(t) = \exp(-\lambda^i \cdot t) \tag{5.8}$$

and that the time-dependent mass of nuclide in the sediment is

$$M^i_{sed}(t) = \int_0^t \left[\alpha^i \cdot M^i_l(t') + \lambda^{i-1} \cdot M^{i-1}_{sed}(t')\right] \cdot \exp\left[-\lambda^i \cdot (t - t')\right] dt' \ . \tag{5.9}$$

For most nuclides, transfer from the water column to the mixed sediments is intimately linked to the process of sedimentation. Nuclides sorb to suspended particulate matter, which then settles out. Sedimentation is a continuous process that results in the gradual accumulation of sediments over time in Canadian Shield lakes. Nuclides deposited at early times will therefore be buried and compacted by subsequent deposits. The sediment compartment considered here is the mixed-sediment layer, the top few centimetres of material that is kept well-mixed by water and biological action. Let the depth of this layer be Z_{sed} (m). The time required (a) to accumulate sediments to this depth is

$$t_{sed} = Z_{sed} \cdot \rho_{sed} / S_{sed} \tag{5.10}$$

where ρ_{sed} is the bulk density of the sediments (kg dry sediment $\cdot m^{-3}$ sediment) and S_{sed} is the sedimentation rate (kg dry sediment$\cdot m^{-2} \cdot a^{-1}$). The total mass of sediment, SM (kg dry sediment), in the mixed-sediment compartment is then

$$SM = S_{sed} \cdot t_{sed} \cdot A_{sed} \tag{5.11}$$

where A_{sed}, the area of the lake bottom covered by sediments (m^2), is assumed to be equal to A_l, the area of the lake. The concentration in the mixed-sediment layer, $C^i_{sed}(t)$ ($mol \cdot kg^{-1}$ dry sediment), assuming uniform mixing, is then given by

$$C^i_{sed}(t) = \hat{M}^i_{sed}(t)/SM \quad . \tag{5.12}$$

Here, $\hat{M}^i_{sed}(t)$ is the nuclide mass (mol) in the mixed sediment accumulated over a period of t_{sed} (a), and is given by Equation (5.9) with a change in the lower limit of integration:

$$\hat{M}^i_{sed}(t) = \int_{t-t_{sed}}^{t} \left[\alpha^i \cdot M^i_l(t') + \lambda^{i-1} \cdot M^{i-1}_{sed}(t')\right] \cdot \exp\left[-\lambda^i \cdot (t - t')\right] dt' \quad . \tag{5.13}$$

Combining Equations (5.5), (5.11), (5.12) and (5.13), and noting that $V_l/A_{sed} = Z_l$, the mean depth of the lake (m), we arrive at the defining equation for the time-dependent mixed-sediment concentration, C^i_{sed} ($mol \cdot kg^{-1}$ dry sediment):

$$C^i_{sed}(t) = \int_{t-t_{sed}}^{t} \left[\frac{\alpha^i \cdot Z_l \cdot C^i_l(t')}{S_{sed} \cdot t_{sed}} + \lambda^{i-1} \cdot C^{i-1}_{sed}(t')\right] \cdot \exp\left[-\lambda \cdot (t - t')\right] dt' \quad . \tag{5.14}$$

Equations (5.6) and (5.14) are integrated numerically using standard techniques built into the SYVAC3 executive (Goodwin et al. 1994). Nuclide concentrations in water are calculated before sediment concentrations so that a time series for $C^i_l(t)$ is available when Equation (5.14) is evaluated. For times $t < t_{sed}$, the lower limit of integration in Equation (5.14) is set to zero.

Equation (5.14) provides estimates of the nuclide concentration in mixed sediments, of depth Z_{sed}. Since we assume that plant roots are distributed through the top 0.3 m of the soil profile (Section 6.1), we require an effective sediment concentration in the top 0.3 m of sediment when the sediments are used as soil. The effective concentration, $C^i_{se}(t)$ ($mol \cdot kg^{-1}$ dry sediment), is calculated as the depth-weighted average of the mixed- and compacted-sediment concentrations for each of the three discharge zones:

$$C^i_{se}(t) = \left[Z_{sed}\cdot C^i_{sed}(t) + (0.3 - Z_{sed})\cdot C^i_{ds}(t)\right]/0.3 \qquad (5.15)$$

where C^i_{ds} is the compacted-sediment concentration ($mol\cdot kg^{-1}$ dry sediment) calculated in Equation (4.7). In areas outside a discharge zone, the effective sediment nuclide concentration C^i_{se}, is the same as C^i_{sed} (Equation 5.14) because we assume that compacted sediments are contaminated to the same extent as mixed sediments (Section 4.4.1). In terms of Equation (5.15), C^i_{ds} would equal C^i_{sed}.

5.3.3 Special Solutions

Under special circumstances, Equations (5.6) and (5.14) can be integrated to give analytical solutions for the water and mixed-sediment concentrations. The time scale of transport processes in the lake is given approximately by $1/\beta^i_l$, where β^i_l is the loss rate (a^{-1}) of nuclide i from the water column for all the loss mechanisms combined, and is given by

$$\beta^i_l = R\cdot A_d/V_l + \alpha^i + \eta^i_l + \lambda^i \quad . \qquad (5.16)$$

Typically, $1/\beta^i_l$ varies from less than one year to a few hundred years. The presence of the term $[-(R\cdot A_d/V_l + \alpha^i + \eta^i_l + \lambda_i)\cdot(t - t')] = \exp[-\beta^i_l\cdot(t - t')]$ in Equation (5.6) means that there will be essentially no contribution to the integral for times prior to t_p, where t_p is defined by the equation $t - t_p \simeq 10/\beta^i_l$. The lower limit of integration in Equation (5.6) can then effectively be changed from 0 to $t - t_p$. Because the geosphere flow, χ^i, generally varies with a time scale that is very much longer than $1/\beta^i_l$, χ^i can be considered constant over the period $t - t_p$ to t. Then, for a nuclide i that has no precursor, Equation (5.6) becomes

$$C^i_l(t) = \frac{\chi^i(t)}{V_l}\int_{t-t_p}^{t} \exp\left[-\beta^i_l\cdot(t - t')\right]\, dt' \quad . \qquad (5.17)$$

This can be integrated directly to give

$$C^i_l(t) = \frac{\chi^i(t)}{V_l\cdot\beta^i_l}\cdot\left[1 - \exp(-\beta^i_l\cdot t_p)\right] \qquad (5.18)$$

which, following our definition of t_p, is approximately

$$C^i_l(t) = \chi^i(t)/(V_l\cdot\beta^i_l) \quad . \qquad (5.19)$$

Equation (5.14) for the mixed-sediment concentration can be similarly simplified. $C^i_l(t)$ will be approximately constant over the interval $t - t_{sed}$ to t, so that for a nuclide with no precursor, Equation (5.14) becomes

$$C^i_{sed}(t) = \frac{\alpha^i\cdot Z_l\cdot C^i_l(t)}{S_{sed}\cdot t_{sed}}\int_{t-t_{sed}}^{t} \exp[-\lambda^i\cdot(t - t')]\, dt' \quad . \qquad (5.20)$$

Carrying out the integration, and using Equation (5.19), we arrive at

$$C^i_{sed}(t) = \frac{\alpha^i \cdot Z_1 \cdot \chi^i(t)}{S_{sed} \cdot t_{sed} \cdot V_1 \cdot \beta^i_1 \cdot \lambda^i} \cdot \left[1 - \exp(-\lambda^i \cdot t_{sed})\right] \quad . \tag{5.21}$$

Now t_{sed} is typically only a few tens of years, so that $\lambda^i \cdot t_{sed}$ will be small for long-lived radionuclides, and $[1 - \exp(-\lambda^i \cdot t_{sed})]$ can be approximated by $\lambda^i \cdot t_{sed}$. Equation (5.21) then reduces to

$$C^i_{sed}(t) = \frac{\alpha^i \cdot Z_1 \cdot \chi^i(t)}{S_{sed} \cdot V_1 \cdot \beta^i_1} \quad . \tag{5.22}$$

Equations (5.19) and (5.21) provide very simple, accurate expressions for the water and mixed-sediment concentrations for nuclides with no precursor. An even simpler sediment relationship (Equation (5.22)) is available for long-lived nuclides. These expression are much easier to understand and to verify than the exact Equations (5.6) and (5.14) used in SYVAC3. Equations (5.19) and (5.22) are used in a sample BIOTRAC calculation in Appendix D.

5.3.4 Special Radionuclides

Water and mixed-sediment concentrations are calculated for all the nuclides in the vault inventory using the methods described in the previous sections, with the exception of the noble gas radionuclides ^{39}Ar, ^{81}Kr and ^{85}Kr. As noted in Section 2.5.5, these gases can impart a significant dose only through air immersion. Air concentrations were estimated conservatively by assuming that the argon and krypton fluxes to the atmosphere equal the predicted fluxes, χ^i_{as}, out of the geosphere. Accordingly, water or sediment concentrations are not required, and are not calculated.

Although the surface-water model includes the process of gaseous evasion, this feature is invoked only for ^{14}C. Too few data exist to define reliable evasion rates, η^i_1, for selenium or iodine, which are also potentially volatile. For radon, evasion is unimportant in determining water or sediment concentrations, which are controlled primarily by the very short half-life (3.8 d) of this radionuclide. Evasion rates are unnecessary for tritium, which is modelled using a specific activity approach (Section 2.5.1). Accordingly, from the point of view of the lake, we assume for now that tritium, selenium, iodine and radon do not degas to the atmosphere, and set their η^i_1 values to zero. This increases their sediment and water concentrations.

On the other hand, evasion must be taken into account in calculating air concentrations for the gaseous nuclides in order that these concentrations not be underestimated. The models and data used to predict the contribution of lake-derived volatile radionuclides to the air concentrations are discussed in Sections 7.3.4 and 7.5.1. They are necessarily fairly crude because of the shortage of information on gaseous evasion. This is one of the instances in which the inventory of a source compartment is not depleted when nuclides are transferred to a second compartment (Section 2.3.3).

This allows both the water and air concentrations to be calculated in a conservative manner even though the process being modelled is not understood in detail. The implications of this procedure for mass balance, and its effect on dose predictions, are discussed further in Section 9.3.

5.4 INTERFACES

The surface water submodel is driven by the nuclide mass flow out of the geosphere (Figure 4-9), including the flows through all aquatic and terrestrial discharge zones and through the well (Figure 5-2). These flows are the primary output of the geosphere model (Davison et al. 1994b). The surface water model is therefore intimately linked to the geosphere model.

The surface water submodel is also closely coupled to the other biosphere submodels. The output of the surface water model is the nuclide concentration in the water column and in the mixed-sediment layer of the discharge lake. The water concentrations are used directly by the food-chain and dose submodel to predict internal doses arising from water ingestion by humans (Section 8.3.1.8) and external doses from swimming or bathing (Section 8.3.2.2). The water nuclide concentrations are also used by the atmosphere submodel to estimate air concentrations arising from the suspension of aquatic particles and gases (Sections 7.3.3, 7.3.4 and 7.3.6). When the critical group is assumed to practise aerial irrigation with lake water, the soil submodel is driven by lake-water concentrations (Section 6.3.7.2). Furthermore, the food-chain and dose submodel uses the water concentrations to estimate nuclide uptake by plants following irrigation (Section 8.3.1.3), by terrestrial animals as a result of ingestion (Section 8.3.1.5), and by freshwater fish inhabiting the lake (Section 8.3.1.7). Finally, when lake sediments are used for farming, the mixed-sediment concentrations (combined with the compacted-sediment concentrations) contribute to the soil concentrations in all the exposure pathways involving the soil.

Water and soil nuclide concentrations based on sediments are also used to evaluate potential chemical toxic effects on humans, and to evaluate radiological and chemical toxic effects on the environment (Chapter 13, Goodwin et al. 1994). More specifically, water concentrations form the basis for calculating radiological doses to fish, and more indirectly, together with related soil concentrations, the radiological doses for various terrestrial biota.

5.5 SURFACE WATER PARAMETERS

The surface water model requires two groups of parameters, one to characterize the hydrological and morphometric properties of Canadian Shield lakes, and the other to describe the geochemical behaviour of nuclides in the lakes. Information on the hydrological properties is readily available from studies carried out over many years by meteorologists, engineers working in flood management and hydroelectric reservoir design, and biologists interested in fisheries productivity. The geochemical database is less well established. The values chosen for each model parameter are discussed in turn below. In the majority of cases, the values are distributed to reflect spatial and temporal variability, and uncertainty in the model formulation and the observational data. For each parameter, we show how

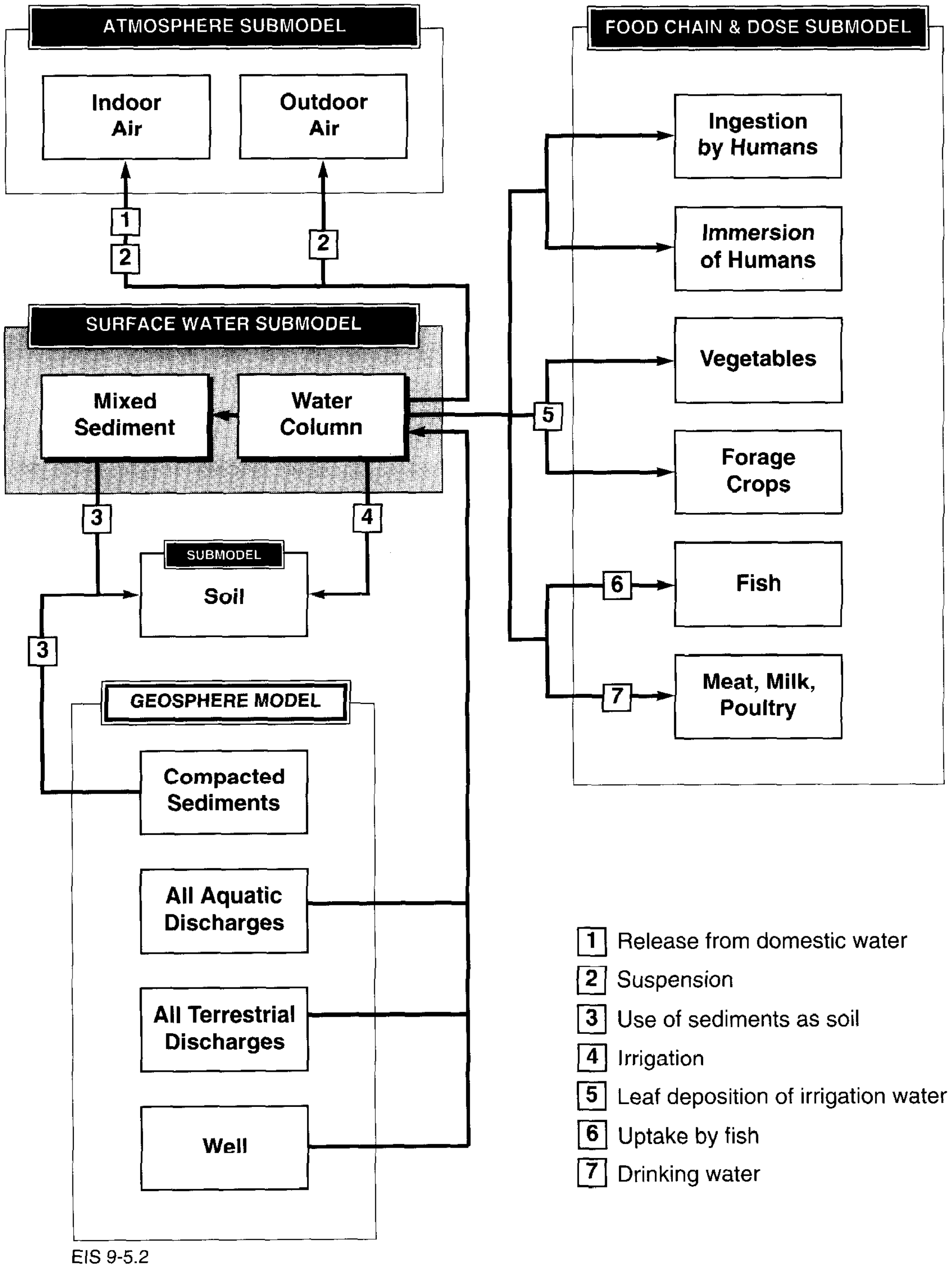


FIGURE 5-2: Interfaces (Related to Human Dose Prediction) Between the Surface Water Submodel (Shaded), the Geosphere Model, and the Other BIOTRAC Submodels

the available data have been used to construct a probability density function suitable for use in our waste management application.

5.5.1 Catchment Area, A_d (m^2)

As noted in Section 4.3, the catchment area used in the surface water model must equal the area of the Boggy Creek watershed to maintain consistency with the boundary conditions used in the geosphere model. Accordingly, a fixed value of 1.06×10^8 m^2 was adopted for A_d.

5.5.2 Lake Mean Depth, Z_l (m)

The only comprehensive published report of Canadian Shield lake depths is the Ontario Ministry of Natural Resources (MNR) Lake Inventory data set (Beals 1985a). This data set is oriented toward sport and commercial fish-producing lakes, and so is biased toward large lakes. It shows the depths to be lognormally distributed, with a GM of 4.6 m and a GSD of 2.0. To ensure that this distribution provides an accurate description of Shield lakes, we assembled depth data from a variety of sources, emphasizing surveys that used unbiased lake selection criteria (Bird et al. 1992). These data, plotted in Figure 5-3, support the assumption that the lake depth is distributed lognormally. They have a GM of 4.76 m and a GSD of 1.99, which are consistent with the MNR results. For postclosure assessment, we adopted the MNR values of GM = 4.6 m and GSD = 2.0, which are slightly lower and therefore predict slightly higher estimates of the nuclide concentrations in water and sediment.

The mean depth distribution is truncated at its lower end at a value of 1.0 m. Canadian Shield lakes are ice-covered to a depth of at least 0.5 m in the winter. Lakes shallower than 1 m would likely freeze to the bottom, and would not be able to supply water or fish to the critical group. An upper truncation limit of 60 m is imposed to eliminate unrealistically deep lakes in terms of the catchment and lake areas.

5.5.3 Lake Area, A_l (m^2)

Although lake area does not appear explicitly in our equations, it is used together with the mean lake depth to calculate the lake volume. We assume that the area, A_{sed}, covered by sediments on the lake bottom, equals A_l.

There are several data sets that report the areas of Canadian Shield lakes, including the MNR Lake Inventory (Beals 1985a), the MNR Counts and Measures (Beals 1985a), and the Canada Centre for Inland Waters database (Minns 1984). Each of these data sets has some advantages and limitations. As noted in the previous section, the MNR data are biased towards large lakes.

The Counts and Measures data were measured from topographic map sheets, but regional differences in map scales cast doubt on the overall quality of the sample. Furthermore, each map scale has a minimum measurable lake size, which biases the sample against small lakes. The data of Minns (1984) are

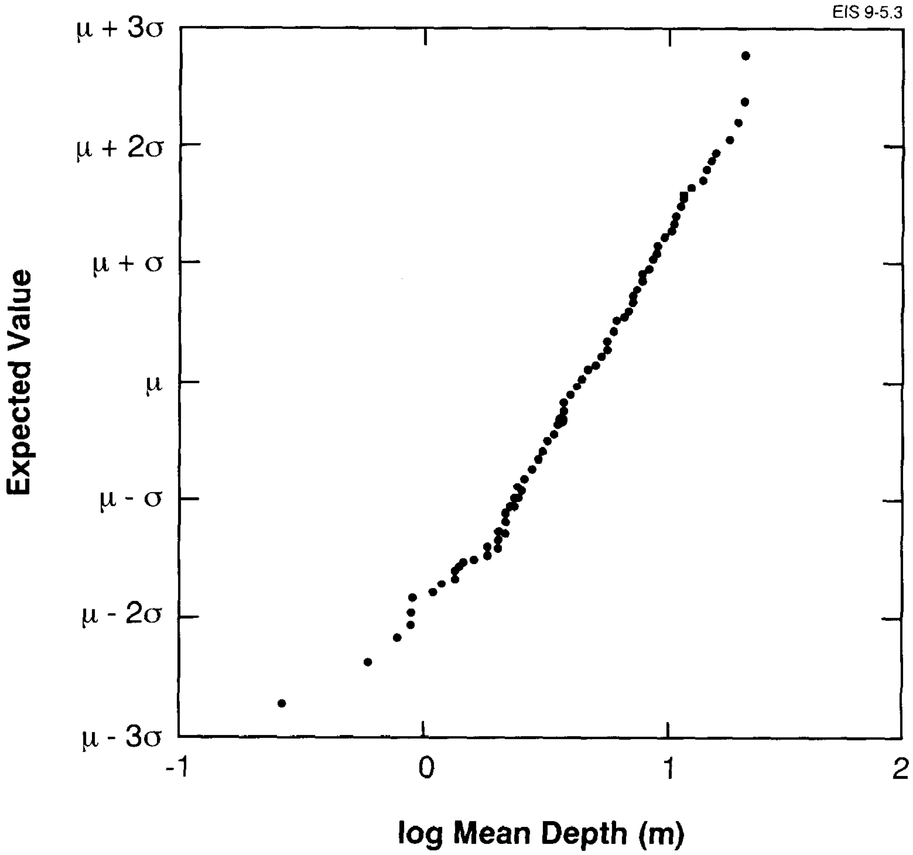


FIGURE 5-3: Probability Plot of the Log-Transformed Mean Depths of Canadian Shield Lakes. A straight line indicates a lognormal distribution. μ and σ are the mean and standard deviation of the log-transformed data.

the least affected by bias. The samples were drawn in an objective manner from 1:50 000 scale map sheets, with a minimum resolvable lake size of 0.2 ha. According to Beals (1985a), most sources indicate that lake areas are distributed lognormally.

We based our distribution for lake area for the postclosure assessment on the Minns (1984) data. In addition to its relatively high quality, this data set also has the smallest value for median lake size. Since nuclide dilution is low in small lakes, this results in conservative estimates for water and sediment nuclide concentrations. Accordingly, we assume that A_l is distributed lognormally with a GM of 7 ha and a GSD of 7.4. As noted in Section 4.3, lakes larger than Boggy Lake would be inconsistent with the boundary conditions assumed in the geosphere model. Accordingly, we truncate the A_l distribution at its upper end at 80 ha. This excludes only about 11% of the largest Canadian Shield lakes. This exclusion is conservative because large lakes have high dilution. The distribution is also truncated at its lower end at 1.0 ha to ensure that there is sufficient water in the lake to meet the demands of the critical group. Our value of

1 ha was established in the following way. We assume that the maximum annual water requirement is 10 000 $m^3 \cdot a^{-1}$ (Sections 4.4.4 and 9.1.1). Since the bulk of this demand would occur in the summer for irrigation purposes, the water must be supplied from the lake volume and not from runoff. Since the minimum mean depth of the lake is 1 m (Section 5.5.2), the lake area cannot be less than 1 ha.

The area and mean depth of Canadian Shield lakes appear to be moderately correlated. The data show a correlation coefficient, r, of 0.48 for the southern region of the Canadian Shield in Ontario (Beals 1985a). The coefficients are lower by about 0.2 to 0.3 for central and northern lakes, but the area data are strongly biased against small lakes in these regions. Our compilation of data from many regions of the Shield suggests that area and depth are correlated with a coefficient of 0.52 (Bird et al. 1992). We assume that lake area and mean depth are correlated with r = 0.5.

5.5.4 Runoff, R
(m $water \cdot a^{-1}$)

Runoff is one component of the water balance, which also involves precipitation, P, and evapotranspiration, ET. Both P and ET are used elsewhere in BIOTRAC, and because all three parameters are closely linked, they must be sampled in a consistent manner. The sampling scheme and the PDF adopted for P, R and ET are discussed in detail in Section 9.1.3. Runoff on the Canadian Shield can be described by a normal distribution with an arithmetic mean of 0.31 m $water \cdot a^{-1}$ and a SD of 0.08 m $water \cdot a^{-1}$. Runoff and precipitation are correlated, with a correlation coefficient, r, of 0.8. The distribution is truncated at a minimum value of 0.01 m $water \cdot a^{-1}$, a very low value found only in desert areas, to which our model and our concept of the critical group do not apply. With a catchment area of 1.06×10^8 m^2, this truncation value of R leads to a minimum flow through the lake of 1.06×10^6 m^3 $water \cdot a^{-1}$, which would likely exceed all the water demands of the critical group.

5.5.5 Sedimentation Rate, S_{sed}
(kg dry $sediment \cdot m^{-2} \cdot a^{-1}$)

The distribution for the sedimentation rate was obtained by synthesizing published results for individual sediment cores taken from Canadian Shield lakes exhibiting a wide range of geographic, physical and chemical properties (Bird et al. 1992). In all cases, the sedimentation values were determined using standard techniques (Appleby and Oldfield 1978, Robbins 1978). The data, plotted in Figure 5-4, are distributed approximately lognormally, with a GM of 0.16 kg dry $sediment \cdot m^{-2} \cdot a^{-1}$, and a GSD of 2.48. We have adopted these values for BIOTRAC.

5.5.6 Sediment Thickness, Z_{sed}
(m)

The active or mixed layer of sediments is quite thin. Sediment cores from Canadian Shield lakes commonly show mixed layers from 0.01 to 0.10 m thick (Robbins and Edgington 1975). For our assessment, we assume that Z_{sed} is distributed uniformly between 0.01 and 0.10 m.

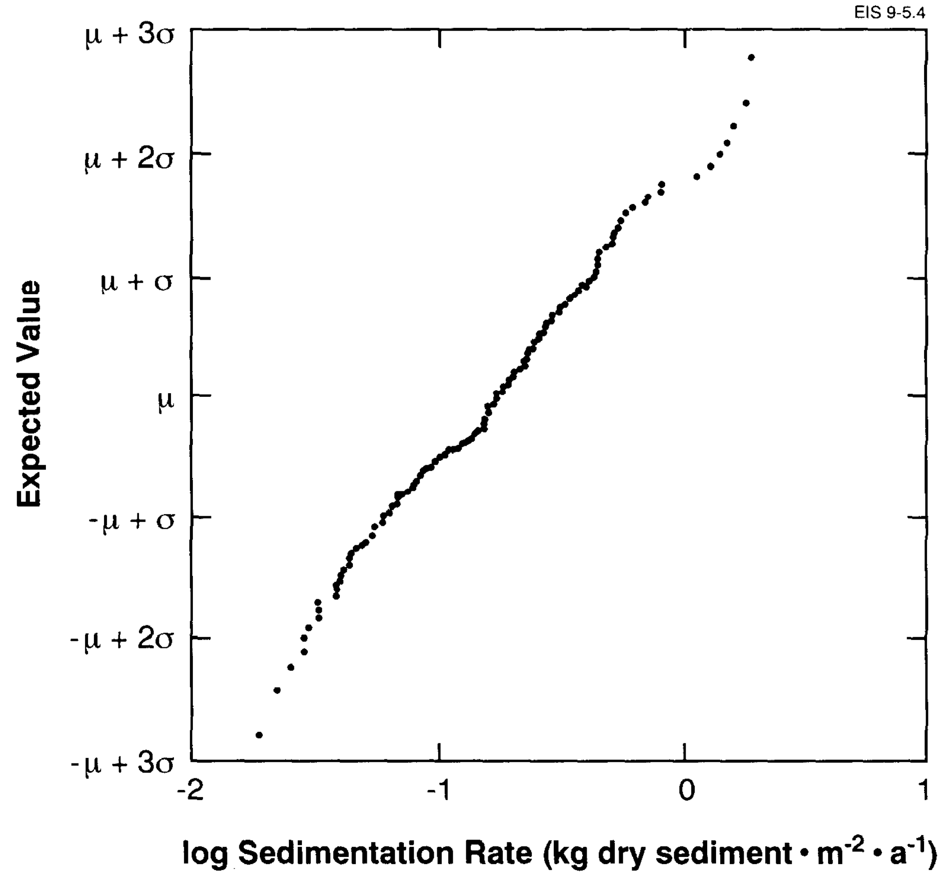


FIGURE 5-4: Probability Plot of the Log-Transformed Sedimentation Rates in Canadian Shield Lakes. A straight line indicates a lognormal distribution. μ and σ are the mean and standard deviation of the log-transformed data.

5.5.7 Mixed-Sediment Bulk Density, ρ_{sed}
(kg dry sediment·m^{-3} sediment)

As indicated in Section 4.5.2, the bulk density of organic sediments varies little from lake to lake, and is not expected to change significantly over time in a given lake. We have therefore adopted a fixed value of 1.25 x 10^2 kg dry sediment·m^{-3} sediment for ρ_{sed}, which is the same value as for ρ_{ds}.

5.5.8 Sediment Transfer Rate, α^i
(a^{-1})

The water/sediment transfer rate or rate constant describes the fraction of nuclide in the water column that is lost to the mixed sediments per unit time. As used to calculate water and sediment concentrations, it represents the net transfer to the mixed sediments, and so implicitly includes resuspension from the sediments back to the water column. It is a bulk parameter that accounts for the sorption of nuclides to suspended particulate material, sedimentation, mixing or diffusion of nuclides into the sediments, and diagenesis within the sediments.

Alpha values can be calculated in a number of different ways depending on the observational data available (Lerman 1979, Bergström et al. 1982, Santschi et al. 1986, Hesslein 1987). Mass balance studies in whole lakes over periods of at least one year provide the most reliable estimates of net transfer rates. However, short-term studies in lakes or lake enclosures (limnocorrals) using other methods can also yield useful data (Santschi et al. 1986). Alpha values are element-specific, and need to be distributed because they show substantial variability.

The data on which the α PDFs are based were drawn from diverse sources (Bird et al. 1992). Most of the information was obtained from Canadian Shield lakes. However, where such data were limited or unavailable, they were supplemented by information from other North American lakes, including the Great Lakes, and from European sources. Where possible, α values based on long-term mass balance studies in whole lakes were used, but the limited number of such studies made it necessary to consider a variety of data from short-term experiments in lakes or limnocorrals as well. Transfer rates measured during the summer, when sedimentation rates are high, may overestimate the annual values. Therefore, α values obtained from summer studies were adjusted downward by a factor of three to provide a better estimate of annual values (Cornett and Ophel 1986). The amount of data available for different elements varied considerably. The transport of elements such as calcium, cesium, phosphorus, lead and thorium has been well studied, whereas only single α values were available for bromine and molybdenum. For several other elements, no values have been measured. Values for these elements were assumed to be equal to those for elements with similar chemistry in the same periodic group. For some elements α values were found to be negative, indicating that the direction of net transfer is from the sediments to the water column. However, in most of these cases, the values were only slightly negative and could have resulted from imprecise measures of nuclide inventory in the water or sediment, or from unmeasured source terms such as sediment weathering or groundwater discharge. Furthermore, since we assume that mixed sediments become contaminated from the water column only, a positive α value is required to give nonzero sediment concentrations. Accordingly, all negative α values were replaced with a value of 0.001 a^{-1}.

Alpha values for those elements for which large amounts of data are available appear to be lognormally distributed (Figure 5-5). Accordingly, we adopted a lognormal PDF for all elements. The GMs and GSDs used for postclosure assessment are listed in Table 5-1. The values show considerable variation from element to element, with the GM ranging from 0.001 a^{-1} to 16.9 a^{-1}, and the GSD ranging from 1.2 to 31.7. Where the values are based on a single measurement only, the GSD was set equal to 6.9, the value for phosphorus, an element for which we have a good data set representing a variety of different lake types. The values given for tritium and radon were estimated by us because no data or suitable analogs were available for these elements.

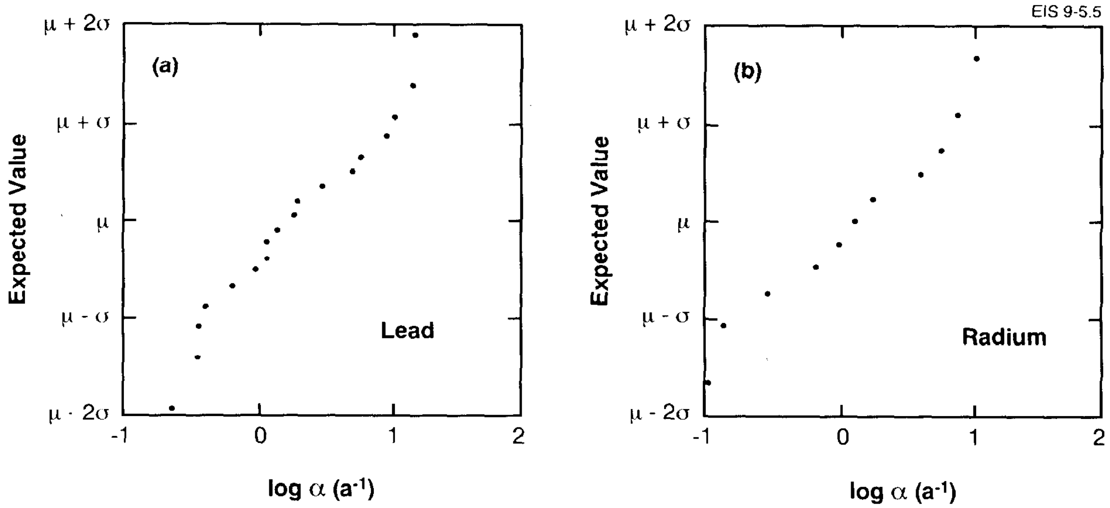


FIGURE 5-5: Probability Plots of the Log-Transformed Sediment Transfer Rates, α, for Canadian Shield Lakes for (a) Lead and (b) Radium. A straight line indicates a lognormal distribution. μ and σ are the mean and standard deviation of the log-transformed data.

5.5.9 <u>Gaseous Evasion Rate from Lake Water, η_l^i</u> (a^{-1})

As noted in Section 5.3.4, only ^{14}C is assumed to be lost from the water column via gaseous evasion to the atmosphere. The evasion rate for ^{14}C was calculated as the ratio

$$\eta_l^c = E/Ic \qquad (5.23)$$

where E is the evasive flux of CO_2 to the atmosphere ($mol \cdot m^{-2}$ lake surface$\cdot a^{-1}$), and Ic is the dissolved inorganic carbon inventory in the lake ($mol \cdot m^{-2}$ lake surface). The evasive flux itself depends on the molecular diffusion coefficient of CO_2 in water, the dissolved CO_2 concentration in the lake, and the boundary-layer thickness at the air/water interface (Hesslein et al. 1980). Similarly, the inventory, Ic, depends on lake depth and the concentrations of CO_2, HCO_3^- and $CO_3^=$ in the lake. We specified PDFs for each of these parameters (Bird et al. 1992), and 1000 values of η_l^c were calculated from Equation (5.23) using 1000 sets of values drawn randomly from these PDFs. The η_l^c values were found to be distributed lognormally, with a GM of 0.92 a^{-1} and a GSD of 5.43.

The distribution of η_l^c depends on the mean lake depth, Z_l, which is sampled independently in each BIOTRAC simulation. To ensure that the sampled values of η_l^c and Z_l are consistent in each simulation, the evasion rate was correlated with lake depth. The correlation coefficient, r, was set equal to -0.77, the value obtained when the values of η_l^c were correlated with the values of Z_l in the simulation we used to calculate them. In this way, shallow lakes, which have a low dissolved carbon inventory per unit area,

TABLE 5-1

ELEMENT-SPECIFIC WATER/SEDIMENT TRANSFER RATES, α^i

Element	Number of Observations	GM (a^{-1})	GSD
Ac (Th)[*]		4.9	3.3
Am (Pu)		0.9	3.0
Be (Pb)		1.9	3.7
Bi (Sb)		0.1	7.2
Br	1	0.2	6.9
C	**	2.2	4.7
Ca	16	0.005	13.1
Cd (Zn)	(21)	1.0	4.7
Cr	3	2.0	16.1
Cs	15	1.8	2.6
H[+]		0.001	6.9
Hf (Np,U)		0.5	3.0
I	1	2.3	6.9
K	14	0.003	11.0
Mo	1	1.0	6.9
Nb (Mo)		1.0	6.9
Ni	3	0.3	1.5
Np (U)		0.5	3.0
P	71	1.5	6.9
Pa (U)		0.5	3.0
Pb	18	1.9	3.7
Pd (Ni)		0.3	1.5
Po	2	1.1	6.2
Pu	11	0.9	3.0
Ra	11	1.3	4.7
Rb (K)		0.003	11.0
Re (Tc,Mn)	(10)	1.7	9.9
Rn[+]		0.001	6.9
Sb	3	0.1	7.2
Se	5	3.1	6.6
Si	4	0.1	1.2
Sm (Pu)		0.9	3.0
Sn	2	16.9	1.2
Sr	5	0.1	31.7
Ta (V)	(1)	12.2	6.9
Tc (Mn)	(10)	1.7	9.9
Te (Se)		3.1	6.6
Th	12	4.9	3.3
U	9	0.5	3.0
Y (La)	(1)	0.3	6.9
Zr (Nb,Mo)		1.0	6.9

[*] Elements in brackets were used as analogs in estimating values for elements for which no data were available.

[**] Values for ^{14}C were calculated (Bird et al. 1992).

[+] Estimated on the basis of expert opinion since no data were available.

are predicted to have large gaseous ^{14}C evasion rates, whereas deep lakes are predicted to have lower evasion rates.

The evasion rate of carbon also appears as a parameter in the atmosphere submodel (Section 7.3.4.3), where it is used in units of s^{-1}. In these units, η_I^c is distributed lognormally, with a GM of 2.9×10^{-8} s^{-1} and a GSD of 5.43.

5.5.10 Radioactive Decay Constant, λ^i (a^{-1})

Values for the radioactive decay constants used in the surface water model and in other parts of BIOTRAC are listed in Table 1-1. Because these values are well known and show no spatial or temporal variations, they are represented by constants.

5.6 MODEL VALIDATION

Validation is the process of demonstrating that a model adequately represents the system it is meant to describe. In the case of the surface water model, validation means showing that the processes responsible for nuclide transport in lakes are properly simulated, and that the predicted nuclide concentrations in water and sediment are realistic.

Validation is normally achieved by showing that model predictions agree with experimental observations when the uncertainties in each are taken into account. In this chapter, and in the following three chapters on the other BIOTRAC submodels, we will restrict our discussion of validation to this aspect. Additional confidence in the model can be gained through activities such as peer review, model and data evaluation, model intercomparisons, sensitivity analysis and code verification, and through the use of natural analogs, conservatism and a formal quality assurance program. These aspects have been treated in a similar manner in all four submodels, and are discussed from an overall perspective in Chapter 11.

The surface water model was experimentally validated in eight separate tests.

1. Predicted and observed water and sediment calcium concentrations were compared for a relatively large, deep lake in northwestern Ontario that was subject to a natural influx of calcium (Schindler et al. 1976). The comparison was done for a single point in time when the concentrations had reached steady state.

2. Predicted and observed water and sediment cadmium concentrations were compared for a lake of average size and depth near Sault Ste. Marie that was contaminated with cadmium through atmospheric deposition (Johnson M.G. et al. 1986, Chan et al. 1986). The comparison was done for a single point in time when the concentrations were assumed to be steady state.

3. Predicted and observed water and sediment ^{60}Co concentrations were compared for a relatively large, shallow lake in the Ottawa Valley that was subject to a time-varying input of ^{60}Co (Cornett

and Ophel 1986). Annual average water concentrations were compared over an 11-a period, and sediment concentrations were compared for two years.

4. Predicted and observed phosphorus concentrations in water were compared for 13 lakes with widely ranging characteristics in the Haliburton-Kawartha area (Dillon and Rigler 1974). The comparison was done for a single point in time when the concentrations were assumed to be at steady state.

5. Model predictions of steady-state concentrations of phosphorus in water and sediments were compared with observations for five wetland areas (Devito et al. 1989). This validation tested the ability of our model to simulate nuclide transport in a variety of permanent wetlands, including beaver ponds.

6. Predicted and observed water concentrations in a Canadian Shield lake in northwestern Ontario treated with ^{60}Co, ^{134}Cs and ^{3}H were compared. The lake was of average depth and above-average area. The comparison was done for a single point in time one year after the nuclides were added to the lake.

7. Model predictions of cadmium concentrations in water and sediments were compared with observations for a Canadian Shield lake of average depth and above-average area in northwestern Ontario. The cadmium was added in an experiment to the lake water during the summer months in four consecutive years. Water and sediment concentrations were sampled in the spring before the cadmium was added for that year. Water concentrations were available for comparison with predictions for three years, and sediment concentrations for one year.

In each of the above seven tests, site-specific data were used wherever possible as input to our model.

8. Predicted and observed water and sediment concentrations were compared on a stochastic basis for 64 lakes in central Ontario that were contaminated by cadmium through atmospheric deposition (Evans H.E. et al. 1983, Dillon and Smith 1984, Johnson M.G. et al. 1986, Stephenson and Mackie 1988). Our model was run numerous times using input parameter values drawn randomly from the PDFs described in Section 5.5. The distributions of predicted concentrations were compared with the observed distributions for a single point in time when the concentrations had reached steady state.

These eight validation studies tested the surface water model under a wide range of conditions. The comparisons involved permanent wetlands and lakes substantially different in area, depth and geographic location. They treated six contaminants with very different geochemical properties. They tested the model in both its deterministic and stochastic formulations, and in both its time-dependent mode and its steady-state limit. In all cases, the model predictions were consistent with observations when uncertainties in the observed data were taken into account, as discussed in detail by

Bird et al. (1992). We conclude that the surface water model adequately simulates contaminant behaviour in lakes characteristic of the Canadian Shield, and that it produces realistic water and sediment nuclide concentrations.

5.7 MODEL DISCUSSION

5.7.1 Assumptions

A number of assumptions were made in deriving the surface water model. In this section, we restate, explain and review the main assumptions, and discuss their effects on the predicted water and mixed-sediment concentrations. In this section, and in similar ones for the other submodels, the model assumptions are stated in italics.

1. *The water body is modelled as a lake rather than as a river or a permanent wetland.* As noted in Section 5.1, water concentrations would be similar in either a lake or river, but sediment concentrations would be higher in a lake. Because permanent wetlands, such as bogs, are not heavily used by humans, the number of potential exposure pathways is higher for a lake than a wetland. Therefore, a lake is the choice that maximizes concentrations and doses. The exposure pathways unique to wetlands are incorporated into BIOTRAC so that no exposure pathways are ignored.

2. *All the nuclides discharging to terrestrial zones and to wells reach the lake instantaneously.* This is obviously a crude assumption, but it allows the transfer of nuclides from the soil and the atmosphere to the lake to be modelled very simply (Section 2.3.3). It leads to an overestimate of the nuclide load in the lake, and a corresponding overestimate of water and sediment concentrations.

3. *The lake drains a catchment area of fixed size.* The value of 1.06×10^8 m^2 lies toward the upper end of the distribution of Canadian Shield catchment areas (Minns 1984). It will therefore lead to relatively high flushing rates, and relatively low water and sediment concentrations. However, it is an appropriate value for the WRA watershed considered in the geosphere model. In interpreting the assessment results, catchment area can be treated simply as one of the many site-specific geosphere parameters. The effects of lower flushing rates are accounted for to some extent through variations in runoff, which is allowed to take on very low values (Section 5.5.4).

4. *Nuclides emerging from the geosphere do not sorb to mixed sediments, but discharge directly into the water column.* The mixed sediments become contaminated only through the deposition of nuclides from the water compartment. Through this assumption, water concentrations are increased, but mixed-sediment concentrations may be underestimated because sorption might occur in the real situation (Johansen et al. 1985). The overall impact would likely be an overestimate of the consequences because the critical group and other biota access the water in the discharge lake far more frequently than they access the sediments. Also, when the sediments are accessed,

an effective sediment concentration, made up of the depth-weighted average of the compacted- and mixed-sediment concentrations, is used in consequence calculations (Equation (5-15)). The compacted sediments usually make up about 75% of the effective sediment depth, and are allowed to sorb nuclides discharging from the geosphere (Section 4.4.1). The effective sediment concentration would therefore not be significantly underestimated by neglecting the contribution of sorption to the mixed-sediment concentration.

5. *Diffusion of nuclides across the sediment/water interface has not been modelled explicitly.* Upward diffusion can be disregarded because all the nuclides are assumed to be advected directly into the water column. Downward diffusion (Nyffeler et al. 1986, Santschi et al. 1986) is unlikely to occur at the point of discharge when the source is underground. Moreover, the water/sediment rate constant, α^i, accounts implicitly for diffusional transport across the interface (Section 5.5.8).

6. *Sediments are deposited permanently and are not transported downstream.* Mass balance calculations of particulate fluxes through lakes suggest that sediment loss through the outflow is very small (Risto et al. 1987), although it increases in small or shallow lakes (Rausch and Schreiber 1981). Resuspended sediment is usually recycled internally within the lake. This assumption is therefore justified. It is also conservative because it overestimates the retention of nuclides within the lake system.

7. *The lake water is a single continuously and instantaneously mixed compartment, with a uniform nuclide concentration throughout.* In reality, nuclides emerging into the lake from a localized discharge zone will be concentrated near the source, and more diluted farther away as a result of diffusion and mixing. In assuming that the lake is well mixed, we effectively assume that water is drawn randomly from the lake relative to the discharge location. A water intake located in the immediate vicinity of the discharge zone would essentially duplicate the well scenario, and would be overly conservative.

   The assumption of a well-mixed compartment also implies that vertical and horizontal stratifications of the lake have been ignored. During the ice-free season, this is a good assumption for shallow lakes, which make up a large portion of Canadian Shield lakes (Section 5.5.2). Tracer studies in such lakes and in the surface layers of deeper lakes have shown that uniform concentrations are achieved within hours (Hesslein et al. 1980, Quay et al. 1980, Hesslein 1987). In deeper lakes, heating in the summer and ice cover in the winter result in the vertical thermal stratification of the water into two or more layers. In this case, the nuclide concentration would be relatively high in the bottom layer receiving the discharge, and relatively low in the upper layers because of reduced mixing. However, most of the stratified lakes mix completely in spring and fall. Our assumption of uniform mixing implies that users randomly access the various layers of the lake.

8. *With the exception of volatile forms of ^{14}C, water concentrations are not depleted in the model when nuclides, whether particles or gases, are lost to the atmosphere.* Too few data exist for now to model suspension processes reliably from the point of view of the lake. This results in an overestimate of the nuclide inventory in the lake, and in conservative water and sediment concentrations.

9. *The net rate of nuclide transport from water to sediments is proportional to the concentration of the nuclide in the lake water.* Short-term experiments (Hesslein et al. 1980, Santschi et al. 1986, Chant and Cornett 1988) and multiyear simulations (Carlsson 1978, Wahlgren et al. 1980, Cornett and Ophel 1986) support the use of first-order kinetics to approximate this transfer. We assume that the water/sediment rate constant, α^i, adequately parameterizes the effects of the various physical, chemical and biological processes (including those responsible for resuspension) that control transfer from water to sediments (Section 5.5.8). This is justified because most of the transfer rates were measured in whole lake ecosystems with these various processes active. Many of the α^i values were derived from short-term studies and may tend to be overestimates. This will result in conservative mixed-sediment concentrations, but an underprediction of water concentrations. However, the underprediction will not be significant because water concentrations are controlled primarily by flushing.

10. *Aquatic biota do not represent a significant sink for nuclides in the water body.* The fraction of the water and sediment nuclide inventory contained in biota is generally less than 1% (Section 5.1 and 9.3.7, Wahlgren et al. 1980, Cornett and Ophel 1986). Therefore, it is not necessary to include biotic activity in the mass balance equations. The uptake of activity by biota is considered in the food-chain and dose model (Zach and Sheppard 1992).

11. *The model does not account for the effects of variations in parameter values throughout the year.* In reality, runoff, sedimentation rate, water-to-sediment transfer rates, evasion rates and other lake properties may all vary considerably throughout the year in response to winter ice cover, spring snowmelt and periods of excess evapotranspiration. However, fluctuations on these time scales have little effect on annual average concentrations. Because the surface water model is linear, annual concentrations calculated using annual averages of the parameter values are equal to the annual average concentration calculated using finer time scales (Section 3.2). Seasonal effects can therefore be neglected in the model.

12. *The long-term evolution of the lake system is not modelled explicitly.* In particular, the lake volume is assumed to remain constant over time, even though observed sediment accumulation rates (Section 5.5.5) suggest that Canadian Shield lakes fill in at the rate of approximately 1×10^{-3} $m \cdot a^{-1}$. However, by setting a high probability on the frequency of occurrence of organic soils, we do model the end point of the infilling process in an approximate way. As

noted in Section 3.3.2.1, it is at this stage of the evolutionary process that impacts on predicted concentrations and doses are the greatest. Other long-term changes, such as eutrophication, are also omitted from consideration. Such changes are expected to have relatively small effects on nuclide concentrations, and are accounted for, to some extent, through the use of distributed parameter values.

Mixed sediments are assumed to evolve into compacted sediments after a period of time, t_{sed} (Equation 5.10), when a mixed-sediment layer of thickness Z_{sed} (Section 5.5.6) has accumulated. We do not explicitly model this conversion because in most cases it would not significantly affect the compacted-sediment concentrations calculated on the basis of upward-moving groundwater (Section 4.4.1). This approach is conservative because it assumes that the compacted sediments achieve an equilibrium level of contamination instantaneously. In areas outside discharge zones, we assume that compacted sediments have the same nuclide concentration as mixed sediments (Section 4.4.1). This is obviously conservative early on when the compacted sediments would be uncontaminated. It may not be conservative at times far in the future when nuclide fluxes from the geosphere are declining. Because the decline would likely be slow relative to the accumulation rate of mixed sediments, the underestimation of nuclide concentrations in these sediments would be small.

5.7.2 Evaluation

The body of knowledge on the physical, chemical and biological processes responsible for contaminant transport and accumulation in lakes is quite extensive. We have used this information to formulate a model of nuclide behaviour in lakes using a simple mass balance approach. Similar models have been shown to perform well in predicting water and sediment concentrations over multiyear periods (Lerman and Taniguchi 1972, Wahlgren et al. 1980, Tracy and Prantl 1983, Cornett and Ophel 1986). The majority of the assumptions inherent in the model lead to overestimates of water and sediment concentrations (Section 5.7.1); where an assumption is non-conservative, concentrations are not significantly underpredicted. Model predictions were successfully validated against observations over a wide range of conditions, including permanent wetlands (Section 5.6). This suggests that the model contains all the important processes, that the processes are adequately simulated and that the model predictions are realistic.

The model was specifically designed for the postclosure assessment. It was formulated to provide annual average concentrations for a generic Canadian Shield lake. Its simplified, efficient structure, coupled with the distributed nature of its input parameters, make it suitable for a probabilistic assessment. It interfaces smoothly with the geosphere model and the other three submodels of BIOTRAC (Figure 5.2).

The parameter values and distributions required by the surface water model were derived from the most appropriate available observational data from Canadian Shield lakes (Section 5.5). Most of the information was extracted from the literature, although some was supplied by our own studies (Cornett et al. 1984, Cornett and Ophel 1986). The database is quite complete, except for some nuclides for which information on water/sediment transfer rates is limited. In these cases, values were determined through expert opinion, or by analogy with another nuclide with similar chemical behaviour. Conservative values were selected for all parameters to compensate for uncertainty.

We conclude that the surface water submodel and its associated database provide a reasonable description of nuclide behaviour in Canadian Shield lakes, and that it will not underestimate concentration and doses when used for the postclosure assessment of the concept for disposal of Canada's nuclear fuel waste.

6. THE SOIL SUBMODEL

6.1 THE SOIL COMPARTMENT

Nuclides may reach the soil directly with contaminated groundwater that discharges to a terrestrial zone, or indirectly via irrigation water or deposition from the atmosphere. Regardless of the source, nuclides may accumulate gradually in the soil over time, and enter crops and natural vegetation. Because the soil interfaces directly with the geosphere and is pivotal in the primary production of agricultural and natural ecosystems, it is not surprising that it is an important determinant of radiological consequences (Chapter 10). The model developed to treat the soil pathways for the postclosure assessment is summarized in this chapter. The information was extracted from the soil submodel report (Sheppard M.I. 1992), which contains additional details. The model has also been published in the open literature by Davis et al. (1992).

The soil compartment comprises the unsaturated layers of soil and overburden above the water table, and it includes temporary, or seasonal, wetlands. The compartment consists of a matrix of mineral and organic particles with water and gases in the pore spaces. It exhibits distinct layers, or horizons, as a result of weathering and pedogenic processes, and is able to support plant life.

Soil properties on the Canadian Shield vary considerably on all spatial scales. Because nuclide behaviour depends strongly on soil texture and mineralogy, it is not possible to treat the soil compartment in a completely generic way. Instead, we recognize four distinct soil types based on soil texture: sand, loam, clay and organic. The latter includes peat. In each model simulation, we choose one of these soil types and assign to it representative generic properties. Sandy soils are most common on the Ontario portion of the Canadian Shield, and loams least common (Section 6.5.1.1). Clays are largely restricted to the northern parts of the region, whereas loams are generally confined to the south. The proportion of organic soils increases in lowland areas.

Soil depth is treated as a probabilistic or distributed parameter in BIOTRAC (Section 6.5.1.2). Soils less than or equal to 0.5 m deep are modelled as a single, well-mixed compartment. For deeper soils, we model the soil profile by dividing it into four layers. The topmost layer is a litter layer composed mainly of decomposed plant residues, and has the characteristics of an organic soil. This layer is the same for all four soil types and has a fixed depth of 0.1 m. The other three layers are soil-type-specific, and have the same physical, chemical and hydraulic properties; these layers define the overall soil type for the profile. The second layer has a depth of 0.2 m, and together with the surface organic layer makes up the 0.3-m-deep root zone. The bottom layer is also 0.2 m deep. This helps to ensure that the pore-water concentration, C^i_{pw}, in this layer is the same as the concentration of the contaminating groundwater from the geosphere below (Section 4.4.3). The depth of the third layer is adjusted to make up the full depth of the soil profile, which is treated as a generic parameter (Figure 6-1).

In each simulation of BIOTRAC, we calculate nuclide concentrations in three different fields: a garden that supplies all of the plant food eaten by the critical group; a forage field that provides the feed or fodder required by their livestock and wild animals; and a woodlot that supplies the wood

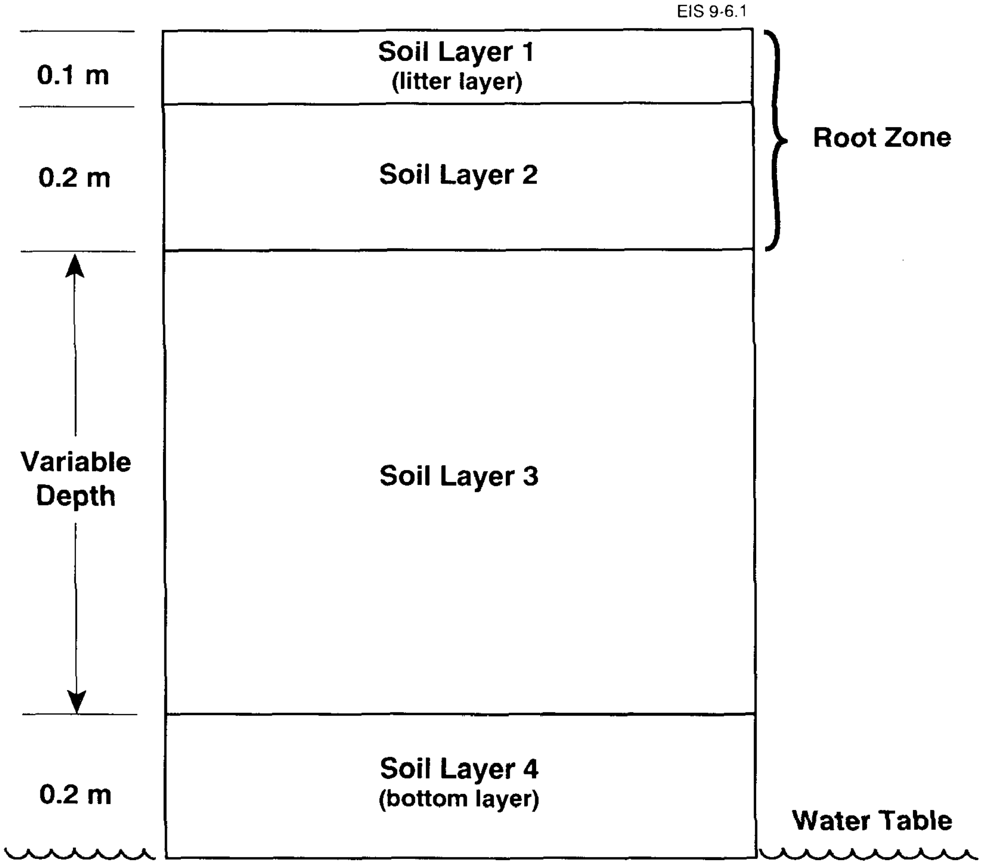


FIGURE 6-1: Soil Profile Used in the Soil Model for Deep Soils Showing the Breakdown into Four Layers. Nuclides may enter the soil through layer 4 from below or through layer 1 from above.

needed to build and heat their home. When the soil type is chosen to be organic, and when the critical group, represented by a household of variable size, is assumed to heat its home with peat (Section 7.3.5.2), we calculate nuclide concentrations in a fourth field with the characteristics of a peat bog. Each of the three or four fields treated in a given simulation has the same soil type, and the same physical, chemical and hydraulic properties. However, in general, each field will have a different nuclide concentration because the distribution of terrestrial discharges and irrigation practices differ from field to field (Sections 6.3.7.1 and 6.3.7.2). Differences among field concentrations may also arise in those simulations in which lake sediments are used for agricultural purposes because of the way in which the sediments are distributed among the fields (Section 6.3.7.3). As discussed in Section 1.5.4, the four fields are also relevant for evaluating potential effects on non-human biota.

In each simulation, the food, wood and peat production needs of the household belonging to the critical group are used to calculate the areas of the four fields (Section 9.1.1.3). Although soil properties vary strongly in space, we assume that soils are homogeneous over the required areas, with properties representative of the soil type sampled by SYVAC3. The geometry and physical layout of the fields do not need to be specified from the point of view of the soil submodel, although some simple assumptions regarding spatial aspects are made in the atmosphere submodel to allow dispersion factors to be calculated (Section 7.3.1).

The objective of the soil submodel is to calculate time-dependent nuclide concentrations for each field used by the critical group and other biota. The model provides the total concentration in the root zone resulting from the terrestrial discharge of contaminated groundwater, the application of contaminated irrigation water, and the deposition of nuclides from the atmosphere. The predictions are used directly to calculate human internal doses from soil ingestion, and external doses from exposure to contaminated soil and inorganic building materials. The predictions are passed to the atmosphere submodel to aid in calculating nuclide concentrations in air, and to the food-chain and dose submodel for estimating concentrations in plants and animals, and internal doses to man. Furthermore, soil concentrations are also used to ensure radiological protection of the environment (non-human biota), and chemical protection of humans and the environment (Chapter 13, Amiro 1992a, Goodwin et al. 1994).

6.2 QUALITATIVE DESCRIPTION OF THE SOIL MODEL

6.2.1 Transport Processes in the Soil

Nuclides in an unsaturated soil are subject to a number of transport mechanisms: the mass flow of dissolved nuclides with the movement of soil water, vapour diffusion of gaseous nuclides through both the vapour and liquid phases of the soil, molecular diffusion along a concentration gradient, diffusion along potential gradients induced by temperature differences in the soil, movement in association with fine particles, microbes or colloids, and mechanical mixing through processes such as ploughing and bioturbation. Since the predominant mechanism for nuclide redistribution in temperate regions is mass flow (Richter 1987), this is the only transport process considered explicitly in our soil model. However, since nuclide concentrations are assumed to be uniform in each soil layer (Section 6.2.2), we implicitly account for mechanical mixing.

Dissolved nuclides moving with the soil water interact chemically with the soil solids. This sorption process is reversible because nuclides may also move from the solid phase back to the aqueous phase. Sorption retards nuclide migration through the soil profile, and allows concentrations to build up to levels that depend on the chemical properties of the nuclide and the soil. We assume that sorption is substantially greater in the surface organic layer of the soil than in the underlying mineral layers. Because of sorption, the soil can act as a barrier to nuclide movement from layer to layer. Strongly sorbing nuclides have long residence times in the soil, and may continue to accumulate over hundreds or thousands of years. The soil model must therefore be time-dependent.

The internal movement of water within an unsaturated soil is determined by the gradient in the hydraulic head or by the water potential, ψ, the binding energy with which water is held in the soil. The potential is made up of two parts: a matric potential, ψ_m, that describes the pressure head in the soil; and a gravitational potential, ψ_g, that represents the head resulting from vertical position in the profile. Under steady-state conditions, a unique relationship exists between ψ_m and the volumetric water content of the soil, θ. This relationship is called the moisture characteristic curve. If a soil is perturbed, for example, by adding water to the upper layer through precipitation, potential gradients are set up. These gradients induce internal flows that reestablish the relationship between ψ and θ at some new point on the characteristic curve. The potentials are such that water movement in soils is primarily vertical. Water drains down through the profile during and immediately after precipitation events, and moves upward by capillary rise during periods of high evapotranspiration.

Since water flow controls nuclide transport, nuclides can move upward or downward through the soil profile (Figure 6-2). For nuclides deposited on the soil surface from the air or with irrigation water, root-zone concentrations are determined primarily by downward leaching and sorption. For nuclides reaching the bottom of the soil profile with groundwater, capillary rise also plays a role. These nuclides move upward through the profile during dry periods, and some sorb to the soil solids. A portion is leached back down following the next rainfall, but the remainder continues to rise to the surface in subsequent dry periods. In this way, nuclides can reach the root zone from the water table even in areas where the net flow of water through the soil is downward.

A number of processes act to deplete nuclide concentrations in soil (Figure 6-3). Nuclides may flow out of the bottom of the profile with drainage water and be lost to the regional groundwater system. During wet weather, some may escape the soil with surface runoff. Nuclides may be lost to the atmosphere through suspension of contaminated particulate matter, or through gaseous evasion. Finally, nuclides may be taken up by plants through their roots, although a portion of these may return to the soil when the plant dies and decays.

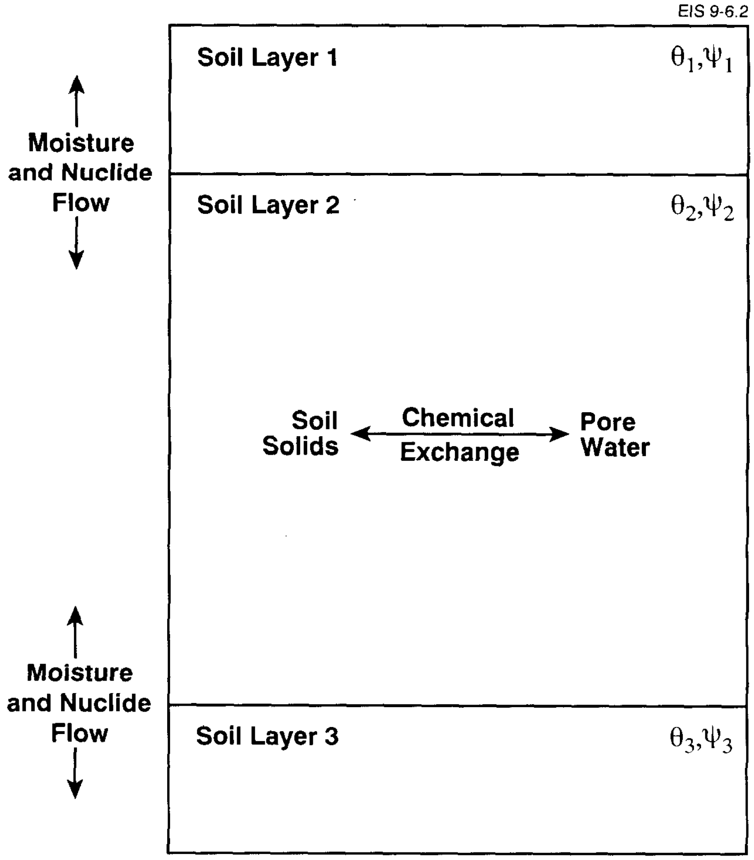


FIGURE 6-2: Water and Nuclide Fluxes in Soil Layer 2. θ_i and ψ_i are the volumetric water content and water potential respectively of the layers.

6.2.2 Modelling Approaches

Soil processes have been studied for many years, and a variety of models have been developed to simulate contaminant behaviour in soils (Broyd et al. 1983). Since contaminants usually reach the soil via the atmosphere, most soil models used for environmental assessments have been designed to treat an atmospheric source. These models are typically formulated as simple compartment models in which the contaminant input is mixed instantaneously and uniformly throughout the surface soil layer (CSA 1987).

Such models are not suited to waste management assessments, which allow the soil to become contaminated from below. In this case, the upward movement of contaminants can be very slow and is not well-described by a single compartment. A more detailed, mechanistic model capable of predicting soil concentrations as a function of depth over long time spans is required. Few such models are available in the literature. We have chosen to base the soil model for the postclosure assessment on the water flow and chemical exchange subroutines of the Terrestrial Ecosystem Hydrology Model (TEHM)

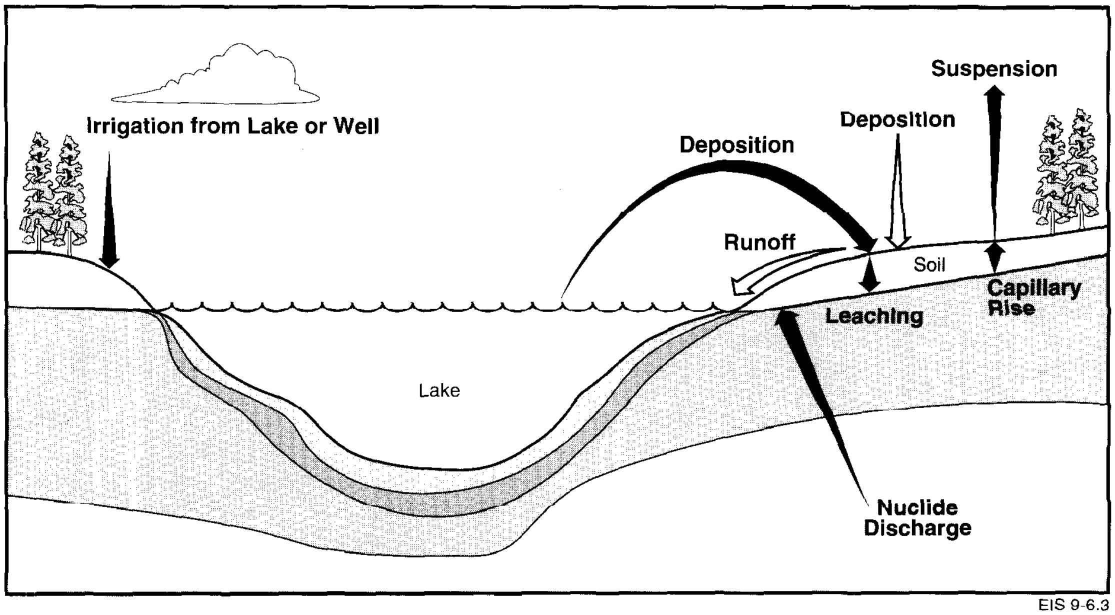


FIGURE 6-3: Transport Processes in the Soil. Closed arrows indicate explicit consideration, and open arrows indicate implicit consideration in BIOTRAC.

developed at Oak Ridge National Laboratory (Huff et al. 1977). TEHM models contaminant transport in a watershed and has been validated experimentally.

Our version of the TEHM code is called SCEMR1 (Soil Chemical Exchange and Migration of Radionuclides - Revision 1 (Bera and Sheppard 1984)). It is a one-dimensional model that considers vertical flow through four soil layers. Each layer is treated mathematically as a compartment; nuclides entering the layer are assumed to be uniformly and instantaneously mixed. SCEMR1 first calculates water flows between the layers. It then redistributes nuclides reaching the soil from above or below throughout the profile, and partitions them between the solid and liquid phases. The use of four layers allows realistic modelling of the slow migration of nuclides through the soil, and of the development of a depth-dependent concentration profile.

Because we assume that water advection controls nuclide transport in soils, the water flow subroutines lie at the heart of SCEMR1. The water content of the four soil layers and flows between the layers are calculated over and over on a daily basis for a standard year's weather using detailed meteorological inputs. The water content is estimated through a water balance approach that takes into account irrigation, precipitation, the fraction of precipitation intercepted by plants, evapotranspiration and advection. The water potentials inferred by these water contents are deduced from the characteristic curve, and are used in the Darcy flow equation and the equation of continuity to estimate the amount of water flowing between layers.

For irrigation, SCEMR1 uses an internal process to determine the times in the standard year at which water is applied, and the amounts applied. During the irrigation season (assumed to be May to September, inclusive), the model checks the soil water content of the top two layers for each day. If the content is below the field moisture capacity (the content at which internal drainage ceases), enough water is added to bring the content up to capacity. The total water volume, I_w (m^3 water$\cdot m^{-2}$ soil$\cdot a^{-1}$), added annually through aerial irrigation is found by summing the daily values predicted by SCEMR1. I_w was found to depend on the soil type, the soil depth Z_s (m), and the annual difference, Pe (m^3 water$\cdot m^{-2}$ soil$\cdot a^{-1}$), between precipitation and surface runoff. We used SCEMR1 to generate numerous values of I_w for a large range of values of Z_s and Pe for each of the four soil types. This information was used to derive a regression equation relating I_w to Z_s and Pe for each soil type. This equation has the form

$$I_w = a + b \cdot Pe + c \cdot Z_s + d \cdot Pe \cdot Z_s \quad . \tag{6.1}$$

Values for the regression constants a, b, c and d are listed by soil type in Table 6-1. Values assigned to Pe, Z_s and other soil model parameters are discussed in Section 6.5. Equation (6.1) allows the annual irrigation volumes predicted by SCEMR1 to be readily calculated for various soil and meteorological conditions.

Nuclides enter the soil either at the surface with irrigation water or atmospheric deposition, or at the water table with groundwater (Section 4.4.3). The nuclide concentration in the receiving layer is calculated using a mass balance equation, taking into account sorption and advection into and out of the layer, and assuming instantaneous, uniform mixing within the layer. The sorption process is modelled using the soil solid/liquid partition coefficient, Kd^i, defined as the ratio of the concentration of nuclide i on soil solids to that in the pore-water. The use of partition coefficients implies that sorption is a reversible process, and that an equilibrium between solid and liquid phases is reached instantaneously. As discussed in Section 4.4.1, partition coefficients are also used to account for sorption in compacted sediments.

TABLE 6-1

VALUES OF THE REGRESSION CONSTANTS FOR CALCULATING IRRIGATION WATER AMOUNTS BY SOIL TYPE

Constant	Sand	Loam	Clay	Organic
a	0.7618	0.7819	0.7861	0.3050
b	-0.4912	-0.4282	-0.3855	-0.2249
c	0.03295	0.1297	0.5734	1.108
d	0.05054	-0.001977	-0.01589	-0.1190

Note: Constants apply to Equation (6.1).

Using the framework developed in Section 4.4.3, the total nuclide mass, M_T^i (mol), in a soil layer of depth Z_s(m) and area A (m^2) is given by

$$M_T^i = (\theta + \rho_s \cdot Kd^i) \cdot A \cdot Z_s \cdot C_{pw}^i \qquad (6.2)$$

where θ is the volumetric water content of the soil (m^3 water$\cdot m^{-3}$ soil),

ρ_s is the bulk density of the soil (kg dry soil$\cdot m^{-3}$ soil),

Kd^i is the soil solid/liquid partition coefficient (m^3 water$\cdot kg^{-1}$ dry soil), and

C_{pw}^i is the concentration of nuclide i in the pore-water (mol$\cdot m^{-3}$ water).

The total soil concentration reflecting the concentration in both solid and liquid phases is given by

$$C_s^i = \frac{M_T^i}{\rho_s \cdot Z_s \cdot A} = (\theta/\rho_s + Kd^i) \cdot C_{pw}^i \quad . \qquad (6.3)$$

Nuclides are transported from the source layers to other layers in the soil profile using the water flows calculated previously. Concentrations in these layers are also calculated using a mass balance approach, assuming that the water entering the layer has a pore-water nuclide concentration equal to that in the layer from which it originated.

SCEMR1 is a detailed research model that can predict time-dependent nuclide concentrations throughout the soil profile. However, it requires a considerable amount of computer time and is not practical for use in a long-term probabilistic assessment. Accordingly, instead of incorporating SCEMR1 itself into BIOTRAC, we incorporated an assessment model based on SCEMR1 predictions. SCEMR1 was run systematically a large number of times outside BIOTRAC and its results were compiled. These results were used to generate regression equations that allow the most important SCEMR1 predictions (steady-state soil concentration and time required to reach this concentration) to be reconstructed very simply from an analytical expression involving the values of the most important input parameters. The regression equations are thus based on results from SCEMR1. The regression equations were incorporated into BIOTRAC in response function form (Section 2.6).

SCEMR1 does not allow nuclides to be lost from the soil profile by any process other than drainage. However, we have incorporated other loss mechanisms into the regression form of the model. Radionuclides may be lost through radioactive decay, through gaseous evasion to the atmosphere (in the case of volatile nuclides), and through root uptake by plants. The amount or rate of nuclide lost from a given layer for each of these processes is assumed to be linearly proportional to the nuclide mass in the layer. Only a small fraction of the material taken up by plants is permanently removed from the soil; the remainder is assumed to be returned to the soil when the plant dies or is harvested. In this way, we implicitly model the recycling of plant material and the nuclides contained in it.

A number of other loss mechanisms are not modelled explicitly. Soil nuclide concentrations are not reduced by the lateral transport of nuclides in runoff water or through eroding soil material. These processes are difficult to model at a generic site where the local topography and the physical layout of the various fields are unknown. Losses via lateral transport are expected to be small, and their omission results in an overestimate of soil concentrations. The contribution of lateral transport through runoff to the nuclide load in the lake is accounted for implicitly since the entire nuclide flow out of the geosphere, including the flow to terrestrial areas and the well, is used to calculate lake water nuclide concentrations (Sections 4.4.2 and 5.2).

Soil nuclide concentrations are also not reduced in the model when contaminated particles are suspended into the atmosphere because losses via this mechanism are expected to be small. This means that atmospheric particulate material derived from the soil need not be considered when calculating soil concentrations from atmospheric deposition. The same is true for nuclides that evade the soil as gases, but in this case we model the net transfer between the soil and the air so that deposition is accounted for implicitly. Similarly, few of the nuclides taken up by plants are permanently removed from the soil in BIOTRAC. Accordingly, there is no need to redeposit nuclides that enter the atmosphere as a result of fires. Since air concentrations are made up of nuclides suspended from the soil, from fires and from the lake (Section 7.3), the only components that need to be considered when calculating soil concentrations resulting from deposition are particles and gases derived from the lake.

The above procedures are used to calculate the time-dependent root-zone soil concentrations for each of three pathways (groundwater discharge, atmospheric deposition, aerial irrigation) that could contribute to the nuclide load in each of the fields used by the critical group and other biota. The total soil concentration for each field is found by summing over the three pathways. All nuclides in the vault inventory, except ^{39}Ar, ^{3}H, ^{81}Kr, ^{85}Kr and ^{222}Rn, are treated in this way. Significant doses from argon and krypton radionuclides arise only through air immersion. Since air concentrations for them can be calculated without knowing the soil concentrations (Section 7.3.2.2), the latter are not computed. Similarly, doses from tritium are calculated using a specific activity approach (Section 8.3.1.13), and soil concentrations are not required. Radon moves essentially in concert with its precursor, ^{226}Ra. Accordingly, we do not track radon explicitly through the soil profile, but deduce its concentration in the root zone from the predicted ^{226}Ra concentrations.

6.3 MATHEMATICAL FORMULATION OF THE SOIL MODEL

6.3.1 SCEMR1 Model

The mathematical basis for the SCEMR1 code is complex. It has been discussed in detail by Goldstein et al. (1974) and Sheppard M.I. (1992), and will only be summarized here. It will be described in two parts, the first dealing with water flow and the second with the calculation of nuclide concentrations.

6.3.1.1 Calculation of Water Contents and Flows

Water flow in the soil profile is predicted by the Darcy flow equation and the equation of continuity (Hillel 1971). Darcy's law defines the flow through a time-invariant system subject to a gradient in the water potential. In one dimension

$$q_z = -K \frac{d\psi}{dz} \tag{6.4}$$

where q_z is the volume flux density of water in the z direction ($m \cdot s^{-1}$),

ψ is the total potential of the soil water (m water),

K is the hydraulic conductivity ($m\ water \cdot s^{-1}$), and

z is the elevation of a space coordinate measured vertically upward from the ground surface (m).

The hydraulic conductivity, K, is a function of the volumetric soil water content, θ, and a number of other soil properties. The values of K and ψ determine how quickly water moves through the soil profile.

Time-dependent flow is treated by introducing the equation of continuity, which is simply a statement of the conservation of mass. In one dimension,

$$\frac{\partial \theta}{\partial t} = -\frac{\partial q_z}{\partial z} \tag{6.5}$$

where θ is the volumetric water content of the soil ($m^3\ water \cdot m^{-3}\ soil$). Combining Equations (6.4) and (6.5) yields

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}\left[K \frac{\partial \psi}{\partial z}\right] \tag{6.6}$$

which provides a general equation for predicting water flow through the soil profile. The potential ψ can be written

$$\psi = \psi_m + \psi_g \tag{6.7}$$

where ψ_m is the matrix potential (m water) and ψ_g is the gravitational potential (m water) of the soil water.

Equation (6.6) is used in an iterative technique to calculate the water content of the four soil layers and the water flows between them on a daily basis. The water content of each layer is initially set equal to the field moisture capacity for the soil in question:

$$\theta_j^o = \theta_{FC} \tag{6.8}$$

where θ_j^o is the volumetric water content of layer j at t = 0 (m^3 water$\cdot m^{-3}$ soil), and

θ_{FC} is the field capacity moisture (m^3 water$\cdot m^{-3}$ soil).

Initial values of the matric potential, ψ_{mj}^o (m water), consistent with the θ_j^o values, are determined from the characteristic curve for the soil. The total potential for each layer, ψ_j^o (m water), is then found by adding ψ_{mj}^o to the gravitational potential, ψ_{gj}^o (m water), for each layer.

This initial hydrological state of the soil will fluctuate in response to precipitation, the application of irrigation water, and evapotranspiration. The increase in water content following a precipitation event depends on the amount of water, P_p (m water$\cdot a^{-1}$), that is able to percolate down into the soil; P_p equals the precipitation less the surface runoff, less the amount of water intercepted by plants, P_I (m water$\cdot a^{-1}$), and less the amount of water evaporated from surface pools, E_D (m water$\cdot a^{-1}$) (Section 9.1.3). E_D is a small term, and P_I is calculated by SCEMR1 from the characteristics of the plant canopy overlying the soil. The relevant input to SCEMR1 is therefore an effective precipitation, Pe (m water$\cdot a^{-1}$), defined as the difference between precipitation and surface runoff. Pe is treated as a distributed parameter (Section 9.1.3).

The amount of irrigation water applied to the garden and forage field is calculated by SCEMR1 as the amount needed to maintain the soil moisture at field capacity (Section 6.2.2). The amount of water lost through evapotranspiration, ET (m water$\cdot a^{-1}$), is also calculated internally by SCEMR1. Daily averaged values of net solar radiation, wind speed, vapour pressure and air temperature are used to predict ET from an energy balance/aerodynamic approach (Cowan and Milthorpe 1968, Tanner 1968) that takes account of the properties of the plant canopy. We assume a canopy typical of the Canadian Shield and characterize it using generic values for average leaf length, leaf area index, root cross-sectional area, root distribution with depth, and surface resistance to water flow (Sheppard M.I. 1992). The canopy geometry is reflected in the values assigned to the albedo. The effects of the canopy on ET are strongly seasonal since the model accounts for leaves appearing in the spring and for leaves falling in the autumn.

The water content and flows on the first day of the simulation are derived as follows. The volumetric water content of the top layer, θ_1^1, is calculated using a water balance equation that takes into account any precipitation or evapotranspiration that occurred on that day. The water balance should also include any flow between layers 1 and 2, but this flow is as yet unknown. It is included in subsequent iterations. θ_1^1 is used to deduce a value of K_1^1 and of the matric potential, ψ_{m1}^1, through the characteristic curve. The total potential, ψ_1^1, is then calculated by adding ψ_{m1}^1 to the gravitational potential ψ_{g1}^1 for the layer. The values of ψ_1^1 and ψ_2^0 are used to predict the flow between layers 1 and 2 using Equation (6.6). This flow is used together with the predicted evapotranspiration to formulate a water balance for layer 2, from which the water content, θ_2^1, is deduced. In subsequent iterations, the flow between layers 2 and 3 is also included in this water balance. The procedures described for layer 1 are used to calculate K_2^1, ψ_2^1, and the flow between layers 2 and 3. Layers 3 and 4 are then treated in a like manner, subject to the restriction that the water content of the bottom layer remains constant. This restriction is consistent with the assumptions made in formulating the nuclide mass balance equation for this layer (Section 4.4.3).

Water conditions throughout the soil profile may not be internally consistent following this first iteration since flows between layers have not been included in the water balance equations. The entire procedure is therefore repeated, taking these flows into account, until the water contents and potentials in all four layers have converged to consistent values. The calculations then proceed to day 2 of the simulation.

During and shortly after precipitation events, water may drain through the bottom of the fourth soil layer and contribute to the groundwater. We assume that lateral flow in the saturated overburden carries this drainage volume away quickly enough to prevent changes in the level of the water table. Since the drainage volume is generally small, we assume further that it does not dilute the nuclide concentrations in the rising groundwater (Section 4.4.3).

The meteorological data used to drive SCEMR1 are observed daily-average values for each day of a year that are typical of the long-term average climate of a specific site on the Canadian Shield in Ontario (Section 6.5.2.1). We assume that the climate does not change with time, so that this sequence of daily values repeats from year to year within a simulation. The model is run until the time series of water fluxes calculated for any one year is equal to the time series predicted for the previous year. These steady-state fluxes are then assumed to persist for all time, and the water flow calculations are subsequently bypassed. This saves a substantial amount of computer time and allows the calculation of nuclide concentrations to be carried out over long time scales.

6.3.1.2 Calculation of Nuclide Soil Concentrations

Nuclide transport in SCEMR1 is driven by the pore-water concentration in the soil layer that receives the nuclide input. Pore-water nuclide concentrations are therefore required for the bottom layer when the soil becomes contaminated by groundwater discharge, and for the root zone (the top two layers) when the contaminant source is irrigation water or the atmosphere. In the groundwater case, concentrations in the bottom layer are derived using the mass balance equation discussed in Section 4.4.3. For irrigation, concentrations in the surface layers are also calculated using a mass balance approach. Using Equation (6.2), the nuclide mass, M_p^i (mol), present in the root zone at time t can be expressed as

$$M_p^i = (\theta + \rho_s \cdot Kd^i) \cdot A \cdot Z_r \cdot C_{pw}^i(t) \tag{6.9}$$

where Z_r is the depth of the root zone of 0.3 m,

A is the area of the field (m^2), and

$C_{pw}^i(t)$ is the root-zone pore-water concentration ($mol \cdot m^{-3}$ water).

In a time interval Δt, the nuclide mass, M_I^i (mol), added to the root zone with irrigation water is given by

$$M_I^i = C_I^i \cdot V_I \cdot A \cdot \Delta t \tag{6.10}$$

where C_I^i is the nuclide concentration in the irrigation water ($mol \cdot m^{-3}$ water),

V_I is the volume of irrigation water applied per unit area and time (m^3 water$\cdot m^{-2}$ soil$\cdot d^{-1}$), and

Δt is the time interval (d).

In practice, $\Delta t = 1$ d because nuclide transport is modelled using the same time steps as the water flow calculations. The total nuclide mass, M_T^i (mol), in the layer at time $t + \Delta t$ is given by the sum of M_P^i and M_I^i:

$$M_T^i = (\theta + \rho_s \cdot Kd^i) \cdot A \cdot Z_r \cdot C_{pw}^i(t) + C_I^i \cdot V_I \cdot A \cdot \Delta t \quad . \tag{6.11}$$

The ingrowth of daughter nuclides and losses through radioactive decay, gaseous evasion, soil leaching and cropping are not included in this equation because they are treated in the response function formulation of the SCEMR1 results (Section 6.3.3). We assume that the nuclide mass is uniformly and instantaneously mixed throughout the root zone, and partition it between solid and liquid phases. M_T^i can then be expressed in terms of the pore-water concentration at time $t + \Delta t$, $C_{pw}^i(t+\Delta t)$ ($mol \cdot m^{-3}$ water), using

$$M_T^i = (\theta + \rho_s \cdot Kd^i) \cdot A \cdot Z_r \cdot C_{pw}^i(t+\Delta t) \quad . \tag{6.12}$$

The daily root-zone pore-water concentrations required to drive SCEMR1 in the irrigation case are found by combining Equations (6.11) and (6.12):

$$C_{pw}^i(t+\Delta t) = C_{pw}^i(t) + \frac{C_I^i \cdot V_I \cdot \Delta t}{(\theta + \rho_s \cdot Kd^i) \cdot Z_r} \quad . \tag{6.13}$$

Equation (6.13) applies only to days on which irrigation water is added to the soil. For other days (when precipitation brings the root-zone water content above field capacity, or between October to April when irrigation is not practised), $V_I = 0$, and nuclides may be lost from the root zone by leaching. The total nuclide mass in the root zone for these days is given by the total mass on the previous day (Equation (6.9)) less the amount lost through leaching:

$$M_T^i = [\theta(t) + \rho_s \cdot Kd^i] \cdot A \cdot Z_r \cdot C_{pw}^i(t) - Fw_r \cdot C_{pw}^i(t) \cdot A \cdot \Delta t \tag{6.14}$$

where Fw_r (m^3 water$\cdot m^{-2}$ soil$\cdot d^{-1}$) is the flux of water out of the root zone. The pore-water concentration at time $t + \Delta t$ is then found by equating (6.12) and (6.14):

$$C^i_{pw}(t+\Delta t) = C^i_{pw}(t)\cdot\left\{\frac{[\theta(t) + \rho_s\cdot Kd^i]\cdot Z_r - Fw_r\cdot\Delta t}{[\theta(t+\Delta t) + \rho_s\cdot Kd^i]\cdot Z_r}\right\} . \qquad (6.15)$$

Equations (6.13) and (6.15) together provide the root-zone pore-water concentrations required to drive SCEMR1 in its aerial irrigation mode.

Because both aerial irrigation and atmospheric deposition may contaminate the surface soil layer, we model deposition in the same way as irrigation. This means that a single set of SCEMR1 simulations suffices to treat these surface sources. The assessment model is also simplified because the same regression equations and response functions can be used for the two cases. To treat the deposition source, we assume that nuclides reaching the soil from the atmosphere each day do so in a volume of water equal to V_I. The concentration in this volume, C^i_D (mol·m⁻³ water), is calculated in such a way that the nuclide flux reaching the soil with the water equals the flux that would have been deposited from the air over the course of the year. This latter flux is derived in Section 7.3.8, and is given by

$$D^i_s = \left[(C^i_a)_{AP} + (C^i_a)_{AG}\right]\cdot\left[Vd + P\cdot Wr\right] \qquad (6.16)$$

where D^i_s is the rate of deposition of nuclide i from the atmosphere to the soil (mol·m⁻² soil·a⁻¹),

$(C^i_a)_{AP}$ and $(C^i_a)_{AG}$ are the air concentrations of particles and gases suspended from the lake (mol·m⁻³ air),

Vd is the dry deposition velocity (m·a⁻¹),

P is the precipitation rate (m water·a⁻¹), and

Wr is the washout ratio (unitless).

To achieve this flux through the annual application of a volume of water equal to I_W, the concentration in the water must be given by

$$C^i_D = D^i_s/I_W$$

$$= \left[(C^i_a)_{AP} + (C^i_a)_{AG}\right]\cdot\left[Vd + P\cdot Wr\right]/I_W \quad . \qquad (6.17)$$

The pore-water concentrations required to drive the soil model in the deposition case are then calculated from an equation identical to (6.13), but with C^i_I replaced by C^i_D. The implications of using an effective water concentration to model the deposition flux are discussed in Section 6.7.1.

Once pore-water concentrations in the two source layers have been defined, nuclide concentrations throughout the soil profile can be calculated. We assume that the nuclides are advected with the water flow calculated by the water flow subroutines of SCEMR1. The concentration in any layer at any time is found through a mass balance calculation. Consider the case in which water flow on a given day is downward through the profile. Let the mass of nuclide i present in layer j at time t be M_p^i (mol); M_p^i is given by

$$M_p^i = (\theta_j + \rho_s \cdot Kd^i) \cdot A \cdot Z_j \cdot Cpw_j^i(t) \qquad (6.18)$$

where $Cpw_j^i(t)$ is the concentration of nuclide i in the pore-water of layer j (mol·m^{-3} water) and Z_j is the depth of the layer (m).

During the day, a mass M_{in}^i (mol) will enter layer j from above, where

$$M_{in}^i = F_{j-1} \cdot Cpw_{j-1}^i(t) \cdot A \cdot \Delta t \quad . \qquad (6.19)$$

Here F_{j-1} (m^3 water·m^{-2} soil·d^{-1}) is the water flow between layers j-1 and j. At the same time, a mass M_{out}^i (mol) will drain out of the bottom of layer j with a water flow F_j:

$$M_{out}^i = F_j \cdot Cpw_j^i(t) \cdot A \cdot \Delta t \quad . \qquad (6.20)$$

The total mass M_T^i (mol) in layer j at time t + Δt is given by the sum of M_p^i and M_{in}, less M_{out}^i. Following Equation (6.3), the total soil concentration Cs_j^i (mol·kg^{-1} dry soil) in the layer is given by $M_T^i/(\rho_s \cdot A \cdot Z_j)$, or

$$Cs_j^i(t+\Delta t) = \Big[(\theta_j + \rho_s \cdot Kd^i) \cdot Z_j \cdot Cpw_j^i(t) + F_{j-1} \cdot Cpw_{j-1}^i(t) \cdot \Delta t - F_j \cdot Cpw_j^i(t) \cdot \Delta t\Big] \cdot (\rho_s \cdot Z_j)^{-1} \quad . \qquad (6.21)$$

Pore-water concentrations for use in the subsequent time step are found by partitioning M_T^i into solid and liquid phases:

$$Cpw_j^i(t+\Delta t) = Cpw_j^i(t) + \Delta t \cdot \Big[F_{j-1} \cdot Cpw_{j-1}^i(t) - F_j \cdot Cpw_j^i(t)\Big] \cdot \Big[(\theta_j + \rho_s \cdot Kd^i) \cdot Z_j\Big] \quad . \qquad (6.22)$$

The linear relationship between nuclide concentrations in the solid and liquid phases does not hold at high concentrations. The concentration on the solid phase, C_{sol} (mol·kg^{-1} dry soil), cannot exceed a limit imposed by the cation exchange capacity, CEC (mol·kg^{-1} dry soil), of the soil. If C_{sol} as predicted by SCEMR1 exceeds CEC, C_{sol} is reset to CEC, and the excess nuclides are assumed to remain in solution. The CEC limit is rarely reached in practice.

The final output of SCEMR1 is a daily time series of total nuclide concentration in the soil root zone, considering both solid and liquid contributions for each of the three contamination sources (groundwater discharge, irrigation and atmospheric deposition). The root-zone concentration in each case is found by taking a depth-weighted average of the total concentrations predicted for the top two soil layers.

6.3.1.3 Sample SCEMR1 Results

Figure 6-4 shows the time-dependent root-zone concentrations predicted by SCEMR1 for soil contamination via both groundwater discharge and irrigation. In both cases, the soil was assumed to be sand 1.5 m deep subject to an effective precipitation of 0.4 m·a^{-1}, and contaminated with ^{129}I

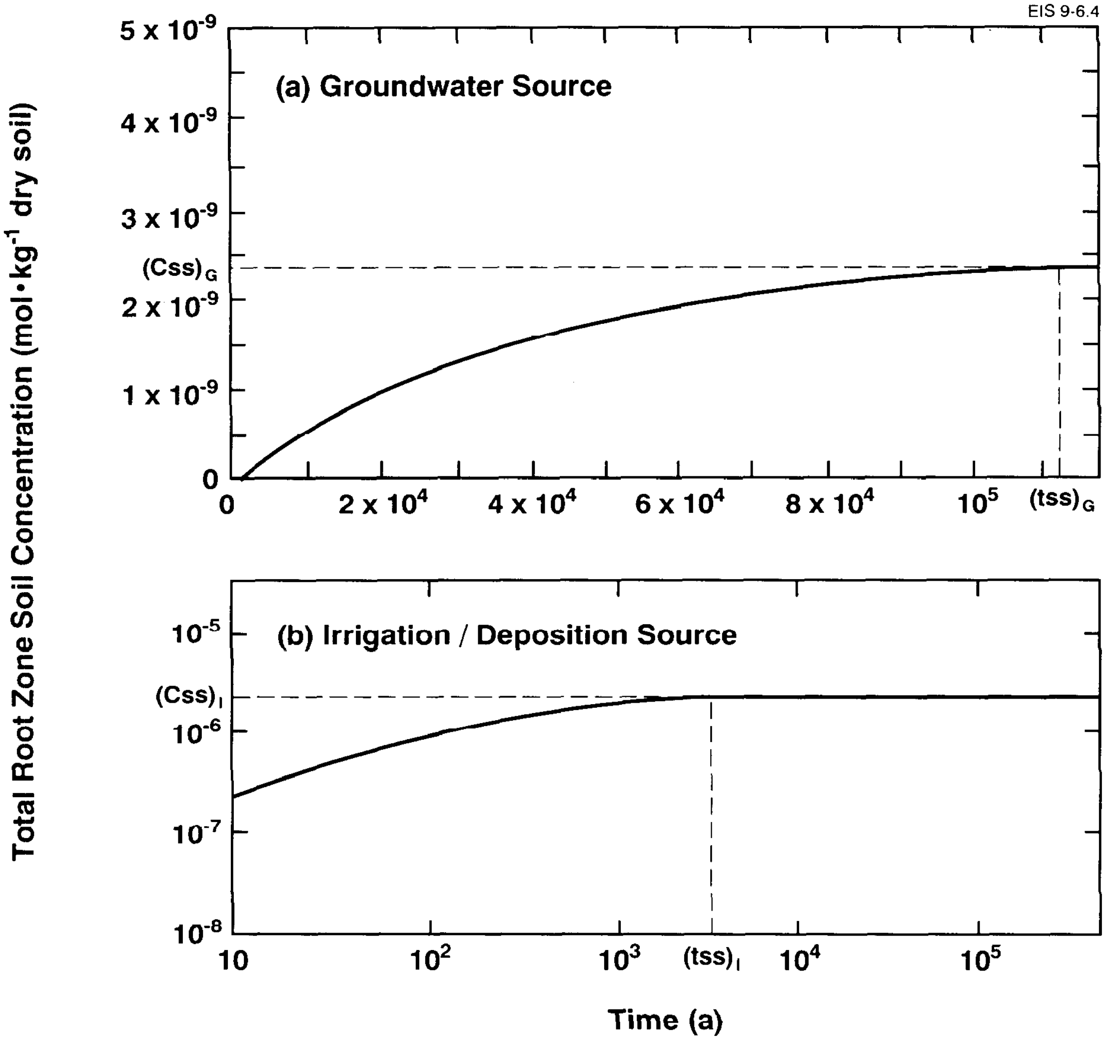


FIGURE 6-4: Time-Dependent Root-Zone Soil Concentrations Predicted by SCEMR1 for a Sand Soil with a Soil Depth, Z_s, of 1.5 m, an Effective Precipitation, Pe, of 0.4 m·a^{-1}, a Solid/Liquid Partition Coefficient, Kd, of 10^{-3} m^3 Water·kg^{-1} Dry Soil and a Source Concentration of 1 μmol·m^{-3} Water. (a) $(Css)_G$ and $(tss)_G$ are the steady-state concentration and the time to reach this concentration for groundwater contamination. (b) $(Css)_I$ and $(tss)_I$ are the corresponding values for irrigation. Note that the scales are linear in (a) but logarithmic in (b).

($Kd^i = 10^{-3}$ m^3 water$\cdot$$kg^{-1}$ dry soil). In the groundwater case, the model was driven by a constant iodine concentration of 1 $\mu mol\cdot m^{-3}$ in the porewater in the bottom soil layer; in the irrigation case, the source term was a constant iodine concentration of 1 $\mu mol\cdot m^{-3}$ in the irrigation water. This normalizes the curves in Figure 6-4.

In the irrigation case (Figure 6-4b), the root zone becomes contaminated as soon as the source is turned on. The concentration increases steadily, and eventually levels off to a steady-state value, $(Css)_I$, at time $(tss)_I$ as the loss of nuclide through leaching balances the gain from additional irrigation. Concentrations resulting from atmospheric deposition would show similar behaviour. In the groundwater case (Figure 6-4a), the root zone does not become contaminated immediately because time is required for the ^{129}I to rise through the soil profile. Following this delay, the concentration builds up to a steady state value, $(Css)_G$, at time $(tss)_G$. The concentration at steady state is higher in the irrigation case than in the groundwater case, and is achieved earlier because the nuclide is applied directly to the root zone.

6.3.2 Regression Model

SCEMR1 requires too much computer time to be of practical use in a probabilistic, long-term assessment model such as BIOTRAC. Instead, we ran SCEMR1 outside of BIOTRAC, and calculated root-zone soil concentrations for a wide range of input parameter values. These databases were then used to generate regression equations that were used together with a simple analytical expression in BIOTRAC to reconstruct the root-zone concentrations from the values of the most important input parameters.

The first step in deriving the regression equations was to identify the SCEMR1 parameters for which distributions, rather than fixed values, are required. This was accomplished through a sensitivity analysis (Sheppard M.I. and Bera 1984). Soil concentrations were predicted using different values of each input parameter in turn, and compared with the results of a standard simulation. The analysis revealed that SCEMR1 predictions are relatively sensitive to variation in four of its parameters: soil type, soil depth, Z_s, effective precipitation, Pe, and soil solid/liquid partition coefficient, Kd^i. Variations in the remaining parameters had little influence on the soil concentrations. Thus, only the four sensitive parameters need to be varied in the regression model for BIOTRAC.

The effective precipitation is defined in SCEMR1 not by a single value, but by a sequence of 365 daily values. Thus there exists a very large number of possible precipitation regimes, and a rigorous treatment of the variability becomes very difficult. Instead, we assume that soil concentrations are sensitive only to variations in the total annual effective precipitation, Pe, and not to variations in its pattern of values throughout the year. We then account for variability in Pe in the usual way by distributing its annual values.

The next step in the procedure was to use SCEMR1 to generate root-zone soil concentrations for a wide range of values of the four sensitive parameters to provide the regression databases. Concentrations were calculated for three soil depths, five effective annual precipitations, and five Kd values for each of the four soil types (sand, loam, clay and organic), for both the groundwater and irrigation/deposition cases. The parameter values used covered the ranges expected to occur on the Canadian Shield in Ontario, given the nuclide inventory in the vault. In each simulation, the daily effective precipitation values, $(Pe)_d$, required to drive SCEMR1 were determined by multiplying the annual value $(Pe)_a$ by the ratio $(P)_{ds}/(P)_{as}$, where $(P)_{ds}$ and $(P)_{as}$ are the daily and total annual precipitations at a representative site on the Canadian Shield (Section 6.5.2).

For these calculations, the model was driven by a constant source term of 1 $mol \cdot L^{-1}$ applied to the irrigation water in the irrigation/deposition case and to the pore-water concentration in the bottom soil layer in the groundwater contamination case. The simulations were carried out to 100 000 a, at which time the root-zone soil concentrations in all irrigation simulations and most groundwater simulations had reached steady state. Steady state was assumed to be achieved when

$$C_s^i(t + \tau_s) - C_s^i(t) < \upsilon \qquad (6.23)$$

where $C_s^i(t)$ is the total root-zone concentration of nuclide i ($mol \cdot kg^{-1}$ dry soil) on a given day of the year in the t^{th} year of the simulation,

$C_s^i(t + \tau_s)$ is the total root-zone concentration ($mol \cdot kg^{-1}$ dry soil) on the same day of the year, τ_s years later,

τ_s is the time interval of SCEMR1 output (a), and

υ is a small nuclide concentration increment ($mol \cdot kg^{-1}$ dry soil).

The results showed essentially no dependence on the day of the year chosen for the comparison. In practice, we used the concentrations predicted for December 31.

The values of both τ_s and υ varied depending on how rapidly steady state was reached (Sheppard M.I. 1992). The steady-state soil concentration, $(Css)^i$ ($mol \cdot kg^{-1}$ dry soil/$mol \cdot L^{-1}$ water), was set equal to $C_s^i(t + \tau_s)$, and was assumed to occur at time $(tss)^i$ (a) = $t + \tau_s$. The $(Css)^i$ and $(tss)^i$ values were stored in a database associated with the parameter values used to generate them. Only the simulations in which steady state had been achieved within 10^5 a were included in the database.

Regression surfaces describing the relationship between $(Css)^i$ and $(tss)^i$ and the input parameters were fitted to the values in the database using standard statistical techniques (SAS 1985). The functional relationship between $(Css)^i$ and the input parameters for the groundwater case was

$$(Css)_G^i = \exp\Big[a_G + b_G \cdot Z_s + c_G \cdot Pe + d_G \cdot \ln Kd^i + e_G \cdot Z_s \cdot \ln Kd^i + f_G \cdot Z_s \cdot Pe + g_G \cdot Pe \cdot \ln Kd^i\Big] \qquad (6.24)$$

where $(Css)^i_G$ is the normalized steady-state root-zone soil concentration of nuclide i resulting from contaminated groundwater ($mol \cdot kg^{-1}$ dry soil/($mol \cdot L^{-1}$ water)),

Z_s is the soil depth (m),

Pe is the annual effective precipitation ($m\ water \cdot a^{-1}$),

Kd^i is the soil solid/liquid partition coefficient for nuclide i ($L\ water \cdot kg^{-1}$ dry soil), and

a_G, b_G, c_G, d_G, e_G, f_G and g_G are regression constants that depend on the soil type.

The values of the regression constants for predicting $(Css)^i_G$ are given in Table 6-2.

The regression equation relating the time to steady state in the groundwater case, $(tss)^i_G$ (a), to the input parameters has the same functional form as Equation (6.24), but involves different regression constants:

$$(tss)^i_G = \exp\left[h_G + l_G \cdot Z_s + m_G \cdot Pe + n_G \cdot \ln Kd^i + q_G \cdot Z_s \cdot \ln Kd^i + r_G \cdot Z_s \cdot Pe + s_G \cdot Pe \cdot \ln Kd^i\right] \quad . \tag{6.25}$$

TABLE 6-2

VALUES OF THE REGRESSION CONSTANTS FOR CALCULATING NORMALIZED STEADY-STATE ROOT-ZONE SOIL CONCENTRATIONS, $(Css)^i_G$, BY SOIL TYPE FOR GROUNDWATER CASE

Constant	Sand	Loam	Clay	Organic
a_G	1.469	0.8901	0.4039	-1.709
b_G	1.507	-0.1329	-1.738	0.6909
c_G	-0.3460	-1.788	-0.1837	3.387
d_G	0.8650	0.5615	0.5632	0.3632
e_G	-0.09265	-0.04214	-0.2248	-0.03611
f_G	-3.542	-0.8678	1.238	-1.883
g_G	-0.1285	0.1098	0.4052	0.1566

Note: Constants apply to Equation (6.24).

Values for the regression constants appearing in Equation (6.25) are listed in Table 6-3.

The regression equation for the normalized steady-state soil concentration in the irrigation/deposition case, $(Css)^{i}_{ID}$ ($\mathrm{mol\cdot kg^{-1}}$ dry soil/($\mathrm{mol\cdot L^{-1}}$ water)), is somewhat more complex than for the groundwater case:

$$\begin{aligned}(Css)^{i}_{ID} = \exp\Big[&a_{ID} + b_{ID}\cdot Z_s + c_{ID}\cdot Pe + d_{ID}\cdot(\ln Kd^{i})^2 + e_{ID}\cdot \ln Kd^{i}\\ &+ f_{ID}\cdot Z_s\cdot \ln Kd^{i} + g_{ID}\cdot Z_s\cdot(\ln Kd^{i})^2 + h_{ID}\cdot Z_s\cdot Pe\\ &+ l_{ID}\cdot Pe\cdot \ln Kd^{i} + m_{ID}\cdot Pe\cdot(\ln Kd^{i})^2\Big] \;. \qquad (6.26)\end{aligned}$$

The equation for the time to steady state in the irrigation/deposition case, $(tss)^{i}_{ID}$ (a), has the same functional form as Equation (6.26), but involves different regression constants:

$$\begin{aligned}(tss)^{i}_{ID} = \exp\Big[&n_{ID} + q_{ID}\cdot Z_s + r_{ID}\cdot Pe + s_{ID}\cdot(\ln Kd^{i})^2 + t_{ID}\cdot \ln Kd^{i}\\ &+ u_{ID}\cdot Z_s\cdot \ln Kd^{i} + v_{ID}\cdot Z_s\cdot(\ln Kd^{i})^2 + w_{ID}\cdot Z_s\cdot Pe\\ &+ x_{ID}\cdot Pe\cdot \ln Kd^{i} + y_{ID}\cdot Pe\cdot(\ln Kd^{i})^2\Big] \;. \qquad (6.27)\end{aligned}$$

TABLE 6-3

VALUES OF THE REGRESSION CONSTANTS FOR CALCULATING TIME TO STEADY STATE OF NORMALIZED ROOT-ZONE SOIL CONCENTRATIONS, $(tss)^{i}_{G}$, BY SOIL TYPE FOR GROUNDWATER CASE

Constant	Sand	Loam	Clay	Organic
h_G	8.653	9.651	10.01	8.533
l_G	2.542	1.282	0.9084	1.614
m_G	-1.029	-2.300	-1.713	-1.548
n_G	0.3621	0.4502	0.3058	0.1616
q_G	-0.06803	-0.07706	-0.06961	-0.03711
r_G	-2.466	-0.6424	-0.3975	-0.8703
s_G	0.3611	0.1438	0.2547	0.2495

Note: Constants apply to Equation (6.25).

Values for the regression constants appearing in Equations (6.26) and (6.27) are listed in Tables 6-4 and 6-5 respectively.

TABLE 6-4

VALUES OF THE REGRESSION CONSTANTS FOR CALCULATING NORMALIZED STEADY-STATE ROOT-ZONE SOIL CONCENTRATIONS, $(Css)^i_{ID}$, BY SOIL TYPE FOR IRRIGATION/DEPOSITION CASE

Constant	Sand	Loam	Clay	Organic
a_{ID}	11.55	6.535	7.078	7.874
b_{ID}	-1.593	0.3149	0.2696	0.4996
c_{ID}	-6.585	-1.003	-1.772	-1.208
d_{ID}	0.09320	0.06383	0.06606	0.07152
e_{ID}	0.5972	0.6659	0.6460	0.4694
f_{ID}	-0.004256	-0.001183	0.007551	0.02331
g_{ID}	-0.006437	0.0001378	-0.0007243	-0.000005273
h_{ID}	1.851	-0.2346	0.4413	0.1503
l_{ID}	0.05587	-0.02532	-0.002168	0.01530
m_{ID}	-0.01808	0.002815	0.0003853	0.0007918

Note: Constants apply to Equation (6.26).

Equations (6.24) to (6.27) allow the steady-state soil concentrations predicted by SCEMR1 to be reproduced efficiently, but additional information is needed to reconstruct the time-dependent concentrations. Based on the shape of the curves shown in Figure 6-4, we postulated simple analytical expressions to describe the soil concentrations as a function of time. For the groundwater case

$$(C^i_s(t))_G = \kappa \cdot C^i_{pw} \cdot (Css)^i_G \cdot \left[1 - \exp(-t/(tr^i)_G\right] \qquad (6.28)$$

where $(C^i_s(t))_G$ is the root-zone concentration of nuclide i at time t resulting from groundwater contamination (mol·kg^{-1} dry soil),

C^i_{pw} is the pore-water concentration in the bottom soil layer (mol·m^{-3} water),

TABLE 6-5

VALUES OF THE REGRESSION CONSTANTS FOR CALCULATING TIME TO STEADY STATE OF NORMALIZED ROOT-ZONE SOIL CONCENTRATIONS, $(tss)^i_{ID}$, BY SOIL TYPE FOR IRRIGATION/DEPOSITION CASE

Constant	Sand	Loam	Clay	Organic
n_{ID}	11.85	7.665	7.653	6.597
q_{ID}	-1.157	0.2745	-0.07020	0.01937
r_{ID}	-6.555	-2.133	-1.747	-0.5435
s_{ID}	0.07153	0.06884	0.06395	0.06858
t_{ID}	0.7790	0.5983	0.6048	0.3601
u_{ID}	-0.01129	-0.009102	0.005474	0.03996
v_{ID}	-0.01589	-0.005360	-0.004422	-0.0006362
w_{ID}	1.740	0.2456	0.5255	-0.2195
x_{ID}	-0.1021	0.02882	-0.02456	0.001428
y_{ID}	0.003770	-0.003250	0.007149	0.008422

Note: Constants apply to Equation (6.27).

κ is a conversion factor equal to 10^{-3} $m^3 \cdot L^{-1}$ water, and

$(tr^i)_G$ is a time scale related to $(tss)^i_G$ (a).

The factor κ converts the pore-water concentrations, which are available from the interface model (Section 4.4.3) in units of $mol \cdot m^{-3}$ water, to $mol \cdot L^{-1}$ water, the units used in SCEMR1 to develop the regression equations.

For the irrigation case

$$(C^i_s(t))_I = \kappa \cdot C^i_I \cdot (Css)^i_{ID} \cdot \{1 - \exp[-t/(tr^i)_{ID}]\} \qquad (6.29)$$

where $(C^i_s(t))_I$ is the root-zone concentration of nuclide i at time t as a result of irrigation water contamination ($mol \cdot kg^{-1}$ dry soil),

C^i_I is the nuclide concentration in the irrigation water ($mol \cdot m^{-3}$ water), and

$(tr^i)_{ID}$ is a time scale related to $(tss)^i_{ID}$ (a).

For the atmospheric deposition case

$$(C_s^i(t))_D = \kappa \cdot C_D^i \cdot (Css)_{ID}^i \cdot \{1 - \exp[-t/(tr^i)_{ID}]\} \qquad (6.30)$$

where $(C_s^i(t))_D$ is the root-zone concentration of nuclide i at time t from deposition contamination ($mol \cdot kg^{-1}$ dry soil), and

C_D^i is the effective deposition water concentration defined in Equation (6.17) ($mol \cdot m^{-3}$ water).

To complete the regression formulation of SCEMR1 results, it is necessary to derive the relationship between the time scales $(tr^i)_G$ and $(tr^i)_{ID}$, and the respective times to steady state, $(tss)_G^i$ and $(tss)_{ID}^i$. For the groundwater case, we substitute Equation (6.28) into Equation (6.23) and rearrange the terms to obtain

$$\frac{(tss)_G^i}{(tr^i)_G} > \ln\left\{\frac{\exp[\tau_s/(tr^i)_G] - 1}{\upsilon/[\kappa \cdot C_{pw}^i \cdot (Css)_G^i]}\right\} \quad . \qquad (6.31)$$

Because the times to steady state were determined in simulations driven by a source term of 1 $mol \cdot L^{-1}$ water, we must set $\kappa \cdot C_{pw}^i = 1$ $mol \cdot L^{-1}$ water in Equation (6.31).

Once $(tss)_G^i$, $(Css)_G^i$, τ_s and υ are known, Equation (6.31) can be solved by standard numerical techniques to yield an upper limit for $(tr^i)_G$. Because the soil concentration at any given time increases with increasing values of $(tr^i)_G$ (Equation (6.28)), use of this upper limit is conservative. It was found that, when the empirical quantity

$$\xi = \upsilon \cdot (tss)_G^i / (\tau_s \cdot \kappa \cdot C_{pw}^i \cdot (Css)_G^i) \geq 0.35 \text{ (with } \kappa \cdot C_{pw}^i = 1 \ mol \cdot L^{-1} \text{ water),}$$

Equation (6.31) is satisfied for all values of $(tr^i)_G$ and cannot be solved. In this case, $(tr^i)_G$ was set equal to 1.6, the value it takes when $\xi = 0.35$. This is conservative since $(tr^i)_G$ decreases for increased values of ξ.

Values for $(tr^i)_{ID}$ are obtained in an identical way using Equations (6.23) and (6.29) or (6.30).

In writing Equations (6.28), (6.29) and (6.30), we assumed that the SCEMR1 predictions, which were obtained with a nuclide concentration of unity in pore water or groundwater, scale linearly with the source terms C_{pw}^i, C_I^i and C_D^i. This assumption was verified in numerous SCEMR1 simulations involving both soluble and insoluble nuclides over a wide range of input concentrations (Sheppard M.I. 1992). Thus the SCEMR1 results that form the basis of the regression equations can be readily used in Equations (6.28) (6.29) and (6.30) to predict soil concentrations for any values of C_{pw}^i, C_I^i and C_D^i calculated by the geosphere and surface water models.

To summarize, the regression model allows SCEMR1 results to be reconstructed through a number of simple steps. A soil type is first chosen, together with values for soil depth, effective precipitation, and Kd appropriate to the nuclide in question. These values are used in regression

Equations (6.24) and (6.25) (for a groundwater source) or Equations (6.26) and (6.27) (for an irrigation/deposition source) to calculate a steady-state concentration, (Css), and the time, (tss), to reach steady state. The time scale, (tr^i), is then calculated from Equation (6.31) (or an equivalent equation in the irrigation/deposition case). Finally, root-zone soil concentrations at any time t are calculated using Equations (6.28), (6.29) or (6.30), with the source concentrations, C_{pw}, C_I or C_D, provided by the geosphere model, or the surface water or atmosphere submodels of BIOTRAC. A comparison of soil concentrations predicted by SCEMR1 and by the regression model is shown in Figure 6-5. For this one case, the steady-state concentrations predicted by the two models agree to within a factor of two for both groundwater contamination and irrigation. A more detailed discussion of the ability of the regression model to duplicate SCEMR1 results is given in Section 6.6.

6.3.3 Response Function Formulation

In the framework developed to this point, we have assumed that the source concentrations driving the soil model are constant in time. In addition, we have not allowed for the buildup of daughter nuclides following decay of the precursor, or for losses resulting from radioactive decay, gaseous evasion or cropping. In this section we develop a fully time-dependent model that treats these additional processes. We achieve this by formulating a mass balance equation for the root-zone compartment, which is solved using the response function/convolution approach discussed in Section 2.6.

We begin by considering the soil concentrations that result from groundwater contamination. A mass balance equation for this case (for a unit soil mass) is obtained by differentiating Equation (6.28) with respect to time:

$$\frac{d(C_s^i(t))_G}{dt} = \kappa \cdot C_{pw}^i \cdot (Css)_G^i \cdot \exp[-t/(tr^i)_G]/(tr^i)_G \quad . \tag{6.32}$$

This can be expressed as

$$\frac{d(C_s^i(t))_G}{dt} = \kappa \cdot C_{pw}^i \cdot (Css)_G^i/(tr^i)_G - (C_s^i(t))_G/(tr^i)_G \quad . \tag{6.33}$$

The first term on the right in Equation (6.33) is the source term resulting from nuclide discharge from the geosphere (Section 4.4.3), and the second is a loss term resulting from leaching. We now generalize Equation (6.33) by allowing C_{pw}^i to be time-dependent, and by adding terms to account for ingrowth, radioactive decay, gaseous evasion and cropping losses:

$$\frac{d(C_s^i(t))_G}{dt} = \kappa \cdot C_{pw}^i(t) \cdot (Css)_G^i/(tr^i)_G + \lambda^{i-1} \cdot (C_s^{i-1}(t))_G - (C_s^i(t))_G \cdot \left[1/(tr^i)_G + \lambda^i + \eta_s^i + \zeta^i\right] \quad . \tag{6.34}$$

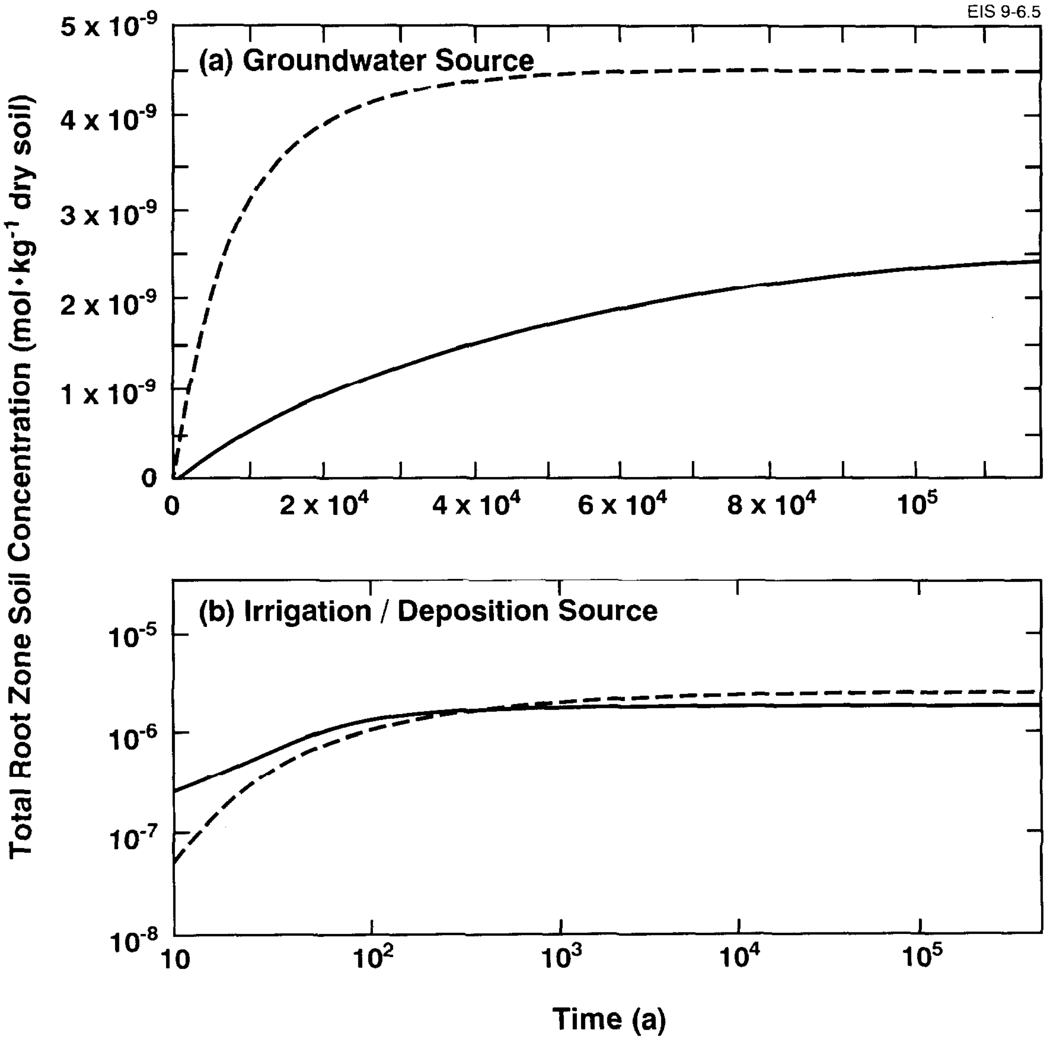


FIGURE 6-5: Comparison of Root-Zone Soil Concentrations Predicted by SCEMR1 (Solid Lines) and the Regression Model (Dashed Lines), for a Sand Soil with a Soil Depth, Z_s, of 1.5 m, an Effective Precipitation, Pe, of 0.4 m Water$\cdot$a^{-1}, a Solid/Liquid Partition Coefficient, Kd, of 10^{-3} m^3 Water$\cdot$kg^{-1} Dry Soil and a Source Concentration of 1 μmol$\cdot$m^{-3} Water. (a) Groundwater source and (b) irrigation/deposition source. Note that the scales are linear in (a) but logarithmic in (b).

In writing Equation (6.34), we have assumed that the sorptive properties of precursor and daughter nuclides are identical. This assumption is reasonable for the actinides (Sheppard M.I. et al. 1985). It has no implications for ^{129}I, ^{14}C and ^{99}Tc, the most important dose contributors (Chapter 10), because these radionuclides have no daughters or precursors.

Each of the loss terms in Equation (6.34) is described by a first-order rate constant expressed in units of a^{-1}: λ^i for decay, η_s^i for gaseous evasion, and ς^i for cropping. Equation (6.34) is subject to the initial condition $(C^i_s(t=0))_G = 0$.

Following the framework presented in Section 2.6, the response function for Equation (6.34) is

$$(RF^i_s(t))_G = \exp\{-[1/(tr^i)_G + \lambda^i + \eta^i_s + \varsigma^i]\cdot t\} \quad . \qquad (6.35)$$

The time-dependent root-zone soil concentration from groundwater contamination is then given by

$$(C^i_s(t))_G = \int_0^t \left[\kappa\cdot C^i_{pw}(t')\cdot(Css)^i_G/(tr^i)_G + \lambda^{i-1}\cdot(C^{i-1}_s(t'))_G\right] \cdot \exp\left\{-[1/(tr^i)_G + \lambda^i + \eta^i_s + \varsigma^i]\cdot(t - t')\right\} dt' \quad . \qquad (6.36)$$

Time-dependent concentrations from irrigation and atmospheric deposition can be derived in a similar manner, beginning with Equations (6.29) and (6.30) respectively. The response function for a surface source becomes

$$(RF^i_s(t))_{ID} = \exp\{-[1/(tr^i)_{ID} + \lambda^i + \eta^i_s + \varsigma^i]\cdot t\} \quad . \qquad (6.37)$$

The time-dependent root-zone soil concentration for an atmospheric deposition source is given by

$$(C^i_s(t))_D = \int_0^t \left[\kappa\cdot C^i_D(t')\cdot(Css)^i_{ID}/(tr^i)_{ID} + \lambda^{i-1}\cdot(C^{i-1}_s(t'))_D\right] \cdot \exp\left\{-[1/(tr^i)_{ID} + \lambda^i + \eta^i_s + \varsigma^i]\cdot(t - t')\right\} dt' \quad . \qquad (6.38)$$

It is unreasonable to assume that a given field can be irrigated indefinitely. The minerals introduced with the irrigation water will accumulate in the soil and eventually make the field unsuitable for agriculture. Accordingly, we assume that irrigation occurs only over a period of time t_{irr} (a). The time-dependent root-zone soil concentration for an irrigation source is then given by

$$(C^i_s(t))_I = \int_{t-t_{irr}}^t \left[\kappa\cdot C^i_I(t')\cdot(Css)^i_{ID}/(tr^i)_{ID} + \lambda^{i-1}\cdot(C^{i-1}_s(t'))_I\right] \cdot \exp\left\{-[1/(tr^i)_{ID} + \lambda^i + \eta^i_s + \varsigma^i]\cdot(t - t')\right\} dt' \quad . \qquad (6.39)$$

If $t - t_{irr} < 0$, the lower limit of integration in Equation (6.39) is set to 0.

Note that the field is irrigated over the preceding period, t_{irr}, for each time t. Therefore, in the case of irrigation, we effectively model a series of fields over time, each of which is irrigated only over t_{irr} before the time of interest. Aerial irrigation may involve either lake or well water (Equations 4.18 and 5.6).

6.3.4 Rate Constant for Cropping Losses

The rate constants λ^i (for radioactive decay) and η_s^i (for gaseous evasion) are fundamental parameters, and can be defined using experimental data (Table 1-1, Section 6.5). On the other hand, ζ^i, the rate constant for cropping losses, is a derived parameter that depends on a number of soil and plant properties. An expression for ζ^i can be derived as follows. The fractional rate of removal of nuclide i from the soil by root uptake, F_r^i (a^{-1}), is given by the ratio

$$F_r^i = U_N^i / I_N^i \qquad (6.40)$$

where I_N^i is the nuclide inventory per unit area of the soil root zone ($mol \cdot m^{-2}$ soil), and

U_N^i is the annual mass of nuclide taken up per unit soil area by the crop ($mol \cdot m^{-2}$ $soil \cdot a^{-1}$).

I_N^i is given by

$$I_N^i = Z_r \cdot \rho_s \cdot C_s^i \qquad (6.41)$$

where Z_r is the depth of the soil root zone of 0.3 m,

ρ_s is the soil bulk density (kg dry $soil \cdot m^{-3}$ soil), and

C_s^i is the root-zone soil concentration of nuclide i ($mol \cdot kg^{-1}$ dry soil).

U_N^i in turn is given by

$$U_N^i = Bv^i \cdot Y_j \cdot C_s^i \qquad (6.42)$$

where Bv^i is the plant/soil concentration ratio for nuclide i (($mol \cdot kg^{-1}$ wet biomass)/($mol \cdot kg^{-1}$ dry soil)), and

Y_j is the annual crop yield (kg wet $biomass \cdot m^{-2}$ $soil \cdot a^{-1}$).

Substituting Equations (6.41) and (6.42) into (6.40), we obtain

$$F_r^i = Bv^i \cdot Y_j / (Z_r \cdot \rho_s) \quad . \qquad (6.43)$$

We assume that only a small fraction, ϵ (unitless), of F^i_r is permanently removed from the soil as a result of cropping, so that

$$\zeta^i = \epsilon \cdot Bv^i \cdot Y_j / (Z_r \cdot \rho_s) \quad . \tag{6.44}$$

The remaining fraction of the nuclides returns to the soil when the plant dies. In this way we model recycling implicitly.

Cropping losses are accounted for in calculating soil concentrations for all the fields except the peat bog, from which no live crop is taken. Different ζ^i values are used for each of the three remaining fields because each field supports a different crop with a different yield, Y_j. Three different types of crops are grown on the forage field to supply feed or fodder to the dairy cows, beef cattle and poultry raised by the critical group (Section 8.5.8.1). The smallest yield value among these crops is used to calculate ζ^i for the forage field to ensure conservatively high predictions.

Cropping is a cultural activity and would not occur unless humans occupied the site and used it for farming. Accordingly, we allow for cropping losses only over the irrigation period, t_{irr}, assuming that a field that is irrigated will be cropped. In cases where irrigation is not practised, we apply the cropping loss over a 50-a period corresponding to the nominal lifetime of members of the critical group (Section 1.2.3). In practice, this restriction was implemented by setting ζ^i equal to the value predicted by Equation (6.44) in the period defined by t_{irr} (or 50 a) prior to the time of interest, and to zero for earlier times.

6.3.5 Treatment of Shallow Soils

Soils less than about 0.5 m deep simulate temporary wetland conditions because they may be subject to seasonal flooding with groundwater as a result of fluctuations in the water table. For the groundwater contamination case, we model these soils as a single well-mixed compartment, with nuclide concentrations controlled by advection. Accordingly, the porewater concentration in the compartment is given by (Equation (4.14)):

$$C^i_{pw}(t) = \int_0^t \left[\frac{\chi^i_{ob}(t')}{\gamma^i \cdot V} + \lambda^{i-1} \cdot (\gamma^{i-1}/\gamma^i) \cdot C^{i-1}_{pw}(t') \right] \cdot \exp\left\{ -[Fw_{ob}/(\gamma^i \cdot V) + \lambda^i] \cdot (t - t') \right\} dt' \tag{6.45}$$

where all the parameters have the meanings given in Section 4.4.3, except for the layer volume, V (m^3), which is based on the total depth of the shallow soil layer:

$$V = Z_s \cdot A_{TD} \quad . \tag{6.46}$$

Here the soil depth, Z_s (m), and the area of shallow soil, A_{TD} (m^2), are associated with a given terrestrial discharge. The total concentration in the layer is then found using Equation (6.3):

$$C_s^i = (\theta/\rho_s + Kd^i)\cdot C_{pw}^i \quad . \tag{6.47}$$

Equation (6.47) provides the root-zone soil concentrations directly without the need to implement SCEMR1 or the regression equations. The advective flow associated with seasonal flooding is far more important in setting the concentration than any capillary rise that may occur throughout the remainder of the year. Similarly, we do not consider cropping losses or gaseous evasion for shallow soils. These mechanisms would be able to act over a period of less than one year before the soil was reflooded; losses over this short period of time would be small.

Shallow soils are subject to contamination by atmospheric deposition in the same way as deep soils; the concentration resulting from deposition is calculated using Equation (6.38). Although shallow soils are sometimes irrigated to prevent salt buildup, we assume that they are not irrigated because plants can draw all the water they need directly from the nearby water table.

6.3.6 Special Solutions

Under special circumstances, Equations (6.36), (6.38) and (6.39) can be integrated to give analytical solutions for calculating the root-zone soil concentrations. If the source concentration, C_{pw}^i, can be considered constant, and we treat a nuclide with no precursor, Equation (6.36) becomes

$$(C_s^i(t))_G = \frac{\kappa\cdot C_{pw}^i\cdot(Css)_G^i}{(tr^i)_G}\int_0^t \exp\left[-(\beta_s^i)_G\cdot(t - t')\right]\,dt' \tag{6.48}$$

where $$(\beta_s^i)_G = 1/(tr^i)_G + \lambda^i + \eta_s^i + \zeta^i \quad . \tag{6.49}$$

Equation (6.48) can be integrated directly to give

$$(C_s^i(t))_G = \kappa\cdot C_{pw}^i\cdot(Css)_G^i/[(tr^i)_G\cdot(\beta_s^i)_G] \quad . \tag{6.50}$$

Under the same conditions, Equation (6.38) for the concentration resulting from deposition becomes

$$(C_s^i(t))_D = \kappa\cdot C_D^i\cdot(Css)_{ID}^i/[(tr^i)_{ID}\cdot(\beta_s^i)_{ID}] \tag{6.51}$$

where $$(\beta_s^i)_{ID} = 1/(tr^i)_{ID} + \lambda^i + \eta_s^i + \zeta^i \quad . \tag{6.52}$$

The analogous equation for an irrigation case is

$$(C_s^i(t))_I = \frac{\kappa\cdot C_I^i\cdot(Css)_{ID}^i}{(tr^i)_{ID}\cdot(\beta_s^i)_{ID}}\cdot\left\{1 - \exp[-(\beta_s^i)_{ID}\cdot t_{irr}]\right\} \quad . \tag{6.53}$$

Equations (6.50), (6.51) and (6.53) provide simple, accurate expressions for the soil concentrations, which, when appropriate, are much easier to interpret and verify than the full equations. They are used in the sample BIOTRAC calculation presented in Appendix D.

6.3.7 Field Concentrations

Equations (6.36), (6.38) and (6.39) provide soil nuclide concentrations in a field subject to contamination by groundwater, irrigation and atmospheric deposition. Because of the uses to which they are put, the fields used by the critical group and other biota (the garden, the forage field, the woodlot and the peat bog) will not necessarily become contaminated via all three pathways. In this section, we discuss the contribution of each pathway in turn to the soil concentration in each field.

6.3.7.1 Groundwater Contamination

Nuclides from the hypothetical vault beneath the WRA are predicted to discharge to the biosphere at three distinct zones in the Boggy Creek watershed (Section 4.2). We assume that a small fraction of each discharge zone underlies arable terrestrial soil. Given the relative sizes of the terrestrial discharge areas, and of the fields required by the critical group, all four fields will not, in general, be subject to groundwater contamination. The available areas of terrestrial discharge are distributed among the fields for each nuclide in the following way. At each time t (a), we use Equation (4.14) to calculate the pore-water concentration in the lowest soil layer resulting from groundwater discharge from each discharge zone in turn. We then rank these concentrations in order of magnitude. Let Cpw_1 be the largest concentration, Cpw_2 be next, and Cpw_3 the smallest. Let the areas associated with these concentrations be A_{TD1}, A_{TD2} and A_{TD3} respectively (Figure 6-6).

We proceed by assuming that the fields become contaminated in the order peat bog, garden, forage field and woodlot (fields 1, 2, 3 and 4 respectively). Let the areas of these fields be A_{F1}, A_{F2}, A_{F3} and A_{F4} respectively. We position the fields over the discharge zones in such a way that the highest concentrations occur below field 1, the next highest below field 2, and so on. In practice, this means that the peat bog is initially assumed to overlie the discharge zone with the largest pore-water concentration, Cpw_1. If $A_{F1} > A_{TD1}$, part of the bog is assumed to overlie discharge zone 2 as well (and perhaps discharge zone 3, depending on the relationship among the areas). On the other hand, if $A_{F1} < A_{TD1}$, the remaining area from discharge zone 1 is available to underlie field 2 (and perhaps fields 3 and 4 as well). This procedure continues until the entire terrestrial discharge area, $A_{TD1} + A_{TD2} + A_{TD3}$, has been assigned to one or the other of the four fields, or until a pore-water concentration has been assigned to the total area under all four fields. Any fields, or portions of fields, that do not overlie a discharge zone will have uncontaminated pore water. In general, different parts of the same field will experience different pore-water concentrations. An effective concentration is found by taking an area-weighted average. This effective concentration is used in Equation (6.36) to calculate average root-zone soil concentrations for that field as a result of groundwater contamination.

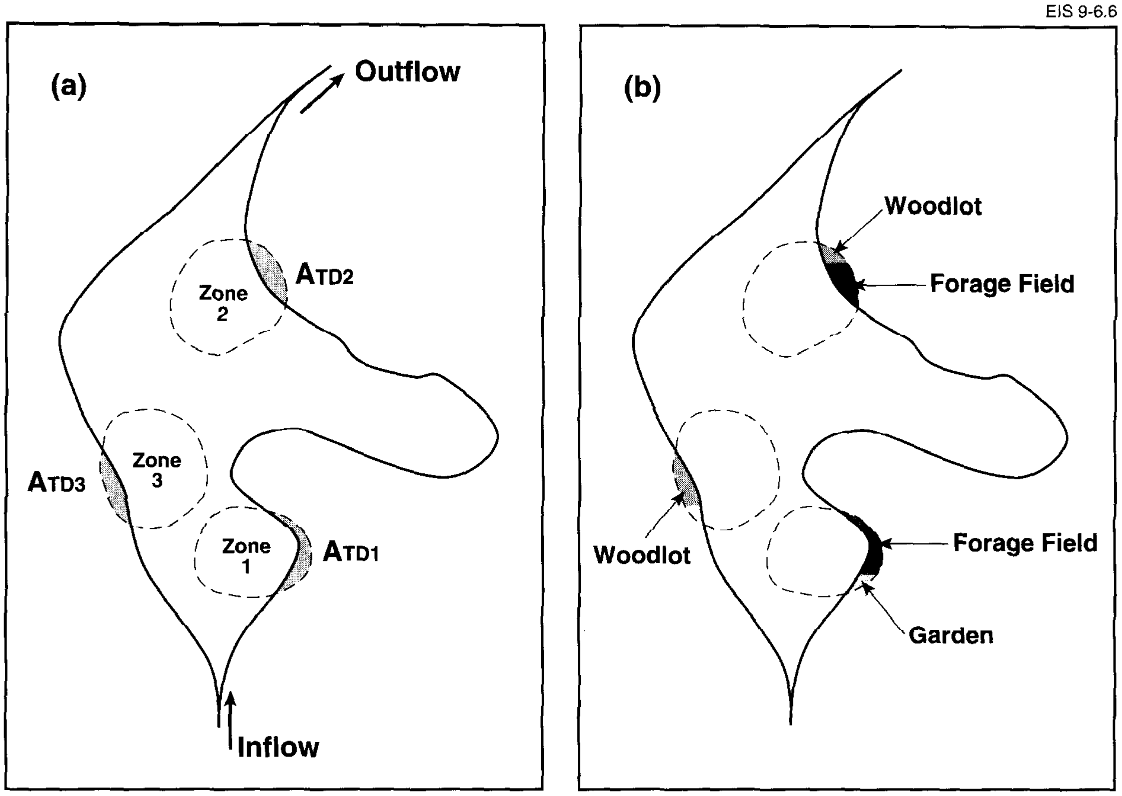


FIGURE 6-6: Schematic Representation of Groundwater Contamination of Fields in the Generic Watershed. (a) Arable terrestrial discharge areas (A_{TD} 1 to 3) associated with discharge zones 1 to 3. Pore-water concentrations, C^i_{pw}, in zone 1 exceed those in zone 2, which exceed those in zone 3. (b) Location of agricultural fields with respect to terrestrial discharge zones, assuming the critical group does not access the peat bog.

Terrestrial discharge occurs infrequently. When it does happen, it is likely to occur to wetlands near the margins of lakes where the soil is commonly organic. It is for this reason that we direct the discharge initially to the peat bog. There is no physical reason for the ordering of the remaining fields; rather, the order was chosen to reflect the importance of the fields in contributing to the dose to the critical group. In general, the highest doses are those associated with the garden, followed by the forage field and then the woodlot. When the critical group does not heat its home with peat (Section 7.5.2.7), the peat bog is not modelled and the remaining fields are moved up one notch each in the assignment procedure.

This approach to terrestrial discharge is conservative in a number of ways. First, we assume that terrestrial discharge always occurs. Secondly, we assume that the agricultural fields are always located over the discharge zones, even though the discharges are widely separated in space and are likely to be wetlands unsuitable for farming. Finally, we assume that the most sensitive fields in terms of dose contributions are located over the most highly contaminated discharges for each nuclide.

6.3.7.2 Irrigation and Atmospheric Deposition

We assume that only two fields are subject to aerial irrigation: the garden and the forage field. The garden is irrigated in most simulations to reflect the common practice of watering gardens (Section 6.5.5.2). On the other hand, the forage field is rarely irrigated, either in reality or in BIOTRAC. When the garden is irrigated, the water originates from the well or the lake with equal probability (Section 9.1.2). The water used to irrigate the forage field is always drawn from the lake because a well is unlikely to be able to meet the high demand. Shallow soils (Section 6.3.5) are not irrigated; in this case, we assume that plants can get all the water they require directly from the water table.

Atmospheric deposition contributes to the soil concentration in all the fields.

6.3.7.3 Sediment Used as Soil

In a small percentage of model simulations, the fields are assumed to be composed of fresh lake sediment (Section 3.3.2.1). This simulates the situation in which the sediments are dredged and applied to the fields, or in which the lake is drained and farmed. The soil concentrations used in these simulations are the effective sediment concentrations discussed in Section 5.3.2, which are calculated as the depth-weighted average concentration through the top 0.3 m of mixed and compacted sediment. These concentrations are used as predicted, without accounting for subsequent losses resulting from radioactive decay, leaching, cropping, gaseous evasion or conditioning of the sediments, or gains resulting from irrigation, deposition or ingrowth of daughters. Some of these processes would increase the concentrations and others would decrease them; it is not clear what the net effect would be. The time interval over which these processes should be allowed to act is also not clear. In the dredging case, a continuous supply of fresh sediments is assumed to be available each year, and concentrations would be maintained at fresh sediment levels regardless of the various processes that might influence this concentration once the sediments are used as soil. On the other hand, if the sediments are subject to these processes over long periods of time, they would take on the characteristics of organic soils. In using the fresh sediment concentrations as predicted, we capture the distinct and important aspects of sediment use as soil.

As with terrestrial discharge, we assume that the fraction of the field area consisting of sediments is limited by the sediment area in the lake (Section 5.5.3), and assign the highest concentrations to the fields having the greatest potential for producing large doses (Figure 6-7). Distinct sediment concentrations are found in four areas of the lake: at each of the three discharge zones and in the remainder of the lake where sediment contamination occurs from the water column only as mixed sediment (Section 4.4.1). The areas of the discharge zones, which are calculated in the

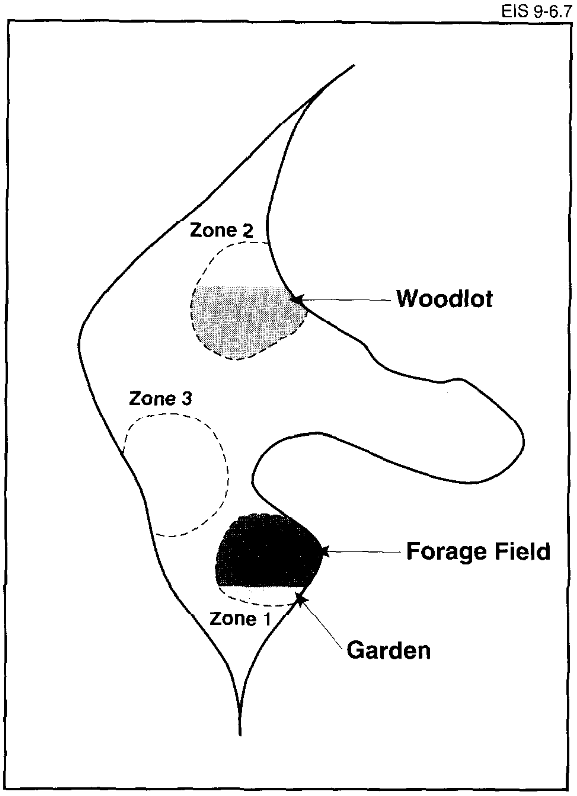


FIGURE 6-7: Schematic Representation of the Location of Agricultural Fields with Respect to the Regions of Contaminated Sediment in the Generic Discharge Lake. Sediment concentrations, $C^i_{s\bullet}$, in zone 1 exceed those in zone 2, which exceed those in zone 3.

geosphere model for present conditions at the WRA, may in some simulations exceed the lake area, which is a sampled parameter in BIOTRAC (Sections 4.3 and 5.5.3). We interpret these simulations as drainage cases in which an originally large lake or wetland area has been partially drained, making available a sediment area surrounding a new smaller lake. In these cases, we assume that the original lake area is equal to the total discharge area so that there is no part of the lake where sediment contamination results only from deposition from the water column.

The distribution of sediment concentrations among the various fields follows a procedure similar to that developed for terrestrial discharge (Section 6.3.7.1). First, the sediment concentrations in the four sediment areas are ranked. The largest concentrations are assigned to the garden, the next largest to the forage field, the next to the woodlot, and any remainder to the peat bog, taking the available areas into account. The ordering of the fields here is somewhat different than for terrestrial discharge. Dredging and draining require considerable effort and would not be undertaken without large benefits being gained. They are most likely to be practised to improve soil conditions in a small field of high value, such as the garden. The remaining fields were ordered to reflect their value to the critical group and their potential contribution to dose.

In some simulations, the sum of the four field areas may exceed the total sediment area. Soil concentrations in the excess area, which would belong to one or more fields, are then predicted using the soil model described in this chapter, taking into account the processes that would normally occur on each field. In general, soil concentrations in different parts of the same field may reflect sediment concentrations from different parts of the lake, as well as true soil concentrations. An effective concentration for each field is found by taking an area-weighted average, which is used as the final soil concentration for that field in BIOTRAC.

6.3.7.4 Contributions to Soil Concentrations in Each Field

Table 6-6 summarizes the processes accounted for in calculating the soil concentration in each field, and the pathways by which the fields become contaminated. The different processes and pathways considered for each field define the only ways in which the fields differ in a given simulation; they are identical in all other respects, including soil properties and the meteorological conditions to which they are exposed. The transport processes considered for a given field are active for all pathways contributing to the soil concentration in that field. The total soil nuclide concentration for each field is found by summing the contributions from each relevant pathway.

6.4 INTERFACES

The primary inputs to the soil submodel are the nuclide concentrations in the water that enters the soil profile from above and below (Figure 6-8). For groundwater discharge, the model is driven by the pore-water concentration in the lowest soil layer, which is available from the interface model (Section 4.4.3). For aerial irrigation, the input is the nuclide concentration in the irrigation water, which is predicted by either the well model or surface water submodel, depending on the water source. For atmospheric deposition, the soil submodel is driven by an effective water concentration based on the air concentrations of lake-derived nuclides as calculated by the atmosphere submodel (Sections 7.3.3.2 and 7.3.4). SCEMR1 requires daily meteorological data to calculate the water contents and flows responsible for redistributing the nuclides within the soil profile.

The size of the areas of the groundwater discharge zones and the agricultural fields used by the critical group must be known in order to determine the contribution of each discharge to the concentration in each field. The areas of the discharge zones are available from the geosphere model (Section 4.2), and field areas can be calculated from the size of the critical group and from its dietary and heating requirements (Section 9.1.1.3). This information is also needed when the critical group is assumed to farm fresh lake sediments. In this case, the nuclide concentrations in the sediments must also be known, and are available from the surface water submodel (Section 5.3.2).

TABLE 6-6

PROCESSES AND PATHWAYS CONTRIBUTING TO SOIL CONCENTRATIONS IN THE VARIOUS FIELDS

Soil and Fields	Nuclide Transport Processes				Contributing Pathways		
	Advection with Water	Gaseous Evasion	Cropping Losses	Decay/ Ingrowth	Groundwater Contamination	Irrigation	Atmospheric Deposition
Deep Soils ($Z_s \geq 0.5$ m)							
Garden	Yes	Yes	Yes	Yes	Yes+	In 90% of runs	Yes
Forage field	Yes	Yes	Yes	Yes	Yes+	In 2% of runs	Yes
Woodlot	Yes	Yes	Yes	Yes	Yes+	No	Yes
Peat bog*	Yes	Yes	No	Yes	Yes	No	Yes
Shallow Soils ($Z_s < 0.5$ m)							
Garden	Yes**	No	No	Yes	Yes	No	Yes
Forage field	Yes**	No	No	Yes	Yes+	No	Yes
Woodlot	Yes**	No	No	Yes	Yes+	No	Yes
Peat bog*	Yes**	No	No	Yes	Yes+	No	Yes
Sediment as Soil	Advection in a saturated sediment layer	No	No	No	Groundwater contamination of sediments	No	No

* The peat bog is modelled only if the soil type is organic and the critical group burns peat for energy.
** Uniform mixing in a single layer.
\+ If area of terrestrial discharge is sufficiently large.

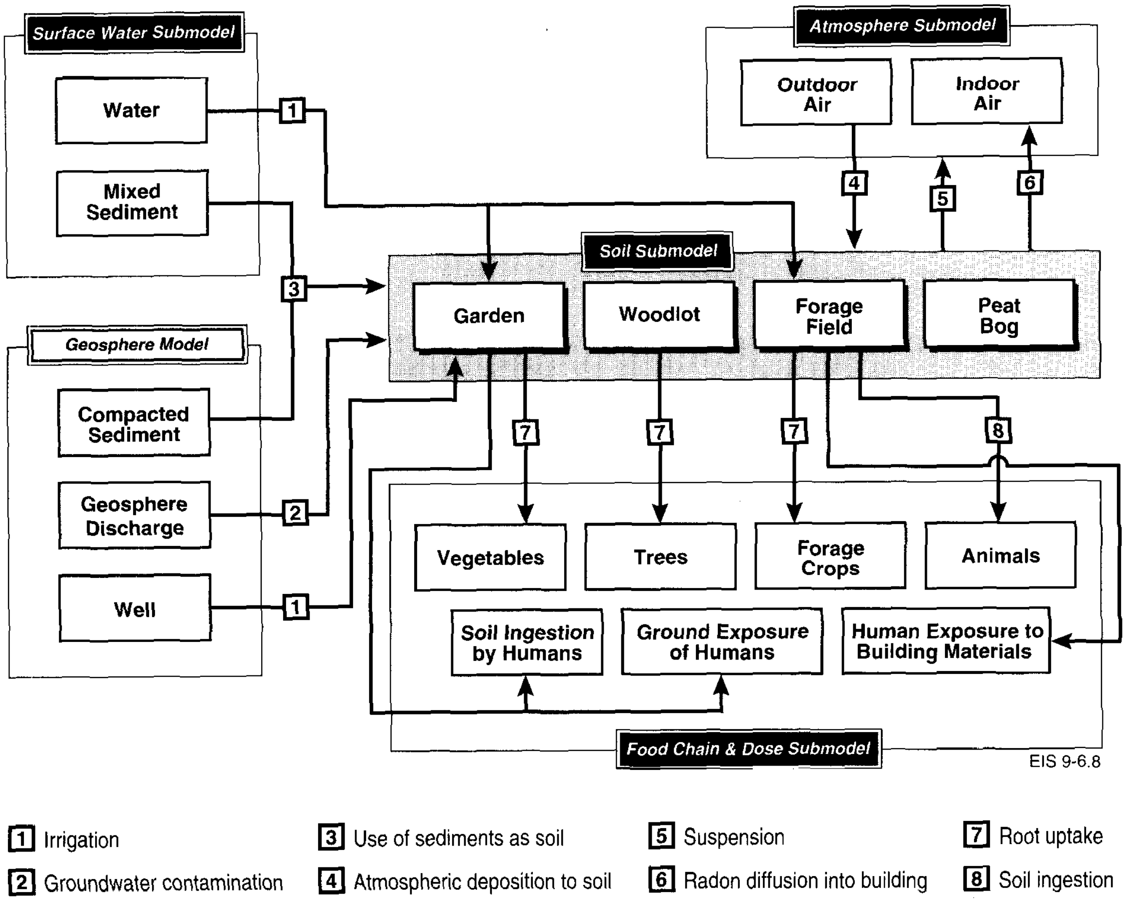


FIGURE 6-8: Interfaces (Related to Human Dose Prediction) Between the Soil Submodel (Shaded), the Geosphere Model and the Other BIOTRAC Submodels

The main output of the soil submodel is the total nuclide concentration in the soil root zone for each of the four fields used by the critical group and other biota. These concentrations are used directly in the food-chain and dose submodel to predict internal doses from soil ingestion by both animals (Section 8.3.1.6) and humans (Section 8.3.1.9), and external doses from exposure to contaminated ground (Section 8.3.2.3). The critical group may also use soil materials in building construction. In this case, we assume that they access the most contaminated soil layer of the forage field (Section 8.3.2.4). Because the soil profile becomes contaminated from either above or below, the most contaminated layer in practice is either the root zone, based on layers 1 and 2, or layer 4, the bottom layer. In the latter case, concentrations are given by Equation (6.3), with parameter values appropriate to the bottom layer. The soil concentrations are also used in the atmosphere submodel to calculate air concentrations resulting from the suspension of nuclides in particulate or gaseous forms (Section 7.3). They are used in the food-chain and dose submodel to estimate nuclide concentrations in plants and ingestion doses to man (Sections 8.3.1.1 and 8.3.1.2).

Soil concentrations are also used for evaluating potential chemical toxic effects on humans, and radiological and chemical toxic effects on the environment (Chapter 13). Soil concentrations are important in determining doses to all biota.

6.5 SOIL MODEL PARAMETERS

Most of the parameters appearing in the soil model describe physical properties of the soil itself or of the prevailing meteorological conditions. Both types of parameters are relatively easy to observe, and there exist fairly extensive, high-quality databases from which values appropriate to the Canadian Shield can be derived. The quality and quantity of the information pertaining to the soil solid/liquid partition coefficients, Kd^i, varies from nuclide to nuclide, and soil type to soil type. Experimental values for the gaseous evasion rates, η_g^i, are limited. The irrigation period, t_{irr}, is a cultural parameter and therefore subject to much uncertainty.

We have not distributed many of the parameters appearing in the soil model, since the model predictions are relatively insensitive to changes in their values. A distribution was assigned to a parameter only when variations in that parameter produced a substantial effect on the soil concentrations. We have introduced limited variability into some of the parameters by assigning different values to different soil types. All soil properties are uniform in the lowest three layers, but the litter layer has the characteristics of an organic soil for all soil types.

For each soil model parameter, we discuss how appropriate values and distributions were derived from the available data.

6.5.1 Soil Properties

6.5.1.1 Frequency of Occurrence of Soil Types

Soil type is treated as a sampled parameter in BIOTRAC. In each simulation, the chosen soil type is assumed to underlie all the fields used by humans and other biota (Section 1.5.4). Because many soil properties are a function of soil type, the predicted root-zone concentrations depend strongly on soil type. In order to obtain realistic concentration and dose estimates, the soil type must be sampled from a distribution that reflects the probabilities that the critical group will farm each of the four major soil types (sand, loam, clay and organic) found on the Canadian Shield. The probability will depend on the availability of a soil type at the discharge zone, the changes in availability that may occur over time, and the likelihood that the critical group would choose to farm a particular soil type. Because the critical group is always assumed to practise farming, the sum of the probabilities must equal unity.

Beals (1985b) used several different data sources to infer the primary soil type in each 4-km^2 grid area on the Ontario portion of the Canadian Shield.

He used this information to deduce the fraction of the total Shield area (excluding areas of water and bare rock) covered by each of the four soil types. The results are shown in Table 6-7. The value of 0.09 may underestimate the true frequency of occurrence of organic soils at a discharge zone. Soils high in organic content are often found on thin strips adjacent to lakes, river and wetlands, which are the likely areas for groundwater discharge. The frequency of occurrence of organic soils in the vicinity of discharge zones may therefore be higher than the frequency averaged across the Shield as a whole. Moreover, because of their strip-like geometry, the areas are easily missed when digitizing soil maps (Beals 1985b).

Tarnocai (1984) estimates that 25% of Ontario is covered by organic soil, although the percentage may be different for the Canadian Shield by itself. Finally, the new soils formed as lakes gradually fill in will be organic, so that the frequency of occurrence of organic soils will likely increase over time.

Some nuclides tend to sorb strongly to organic material, and so their concentrations may be higher in organic soils than in other soil types for a given source strength. It is therefore important not to underestimate the frequency of occurrence of organic soils. To be conservative, we assume that organic soils cover 25% of the area in the vicinity of a discharge zone, and rescale the values for the other soil types proportionately. The results are shown in Table 6-7.

TABLE 6-7

FRACTION OF THE ONTARIO SHIELD (excluding lakes and areas of bare rock) COVERED BY EACH OF THE FOUR SOIL TYPES

Soil Type	Estimates by Beals (1985a)	Rescaled Assuming Fraction of Organic Soils is 0.25	Rescaled Taking Suitability for Farming into Account
Sand	0.61	0.50	0.57
Clay	0.25	0.21	0.24
Organic	0.09	0.25	0.14
Loam	0.05	0.04	0.05
Total	1.0	1.0	1.0

Some soil types are less suitable for farming than others. In particular, organic soils are characterized by low temperatures in spring because of their high heat capacity and low thermal conductivity, and they are also nutrient-deficient, which makes them unsuitable for many crops (Ewanek and Toews 1988). Accordingly, farmers will often avoid organic soils if other soil types are available nearby. We assume that, given equal access to all soil types, the critical group chooses to farm organic soils only half as often as any of the mineral soils. Weighting the rescaled values in this way produces the suitability values in Table 6-7. These are the values used in BIOTRAC to represent the probabilities that the critical group will farm each of the four soil types.

6.5.1.2 Soil Depth, Z_s
(m)

The soil depth defines the thickness of the unsaturated overburden lying above the water table. The soil depth is required in SCEMR1 to set the gravitational potential of soil water, ψ_g, for the soil layers (Equation (6.7)); it is required in BIOTRAC to evaluate the regression equations for $(Css)^i_G$, $(tss)^i_G$, $(Css)^i_{ID}$ and $(tss)^i_{ID}$ (Equations (6.24) to (6.27)), and to calculate soil concentrations in shallow profiles (Equations (6.45) to (6.47)). The soil depth is also used in Equation (6.1) to calculate the total volume of irrigation water applied to the soil surface.

The predicted soil concentrations are quite sensitive to variations in soil depth (Sheppard M.I. and Bera 1984); the soil depth is therefore treated as a distributed parameter in BIOTRAC. The depths of Canadian Shield soils vary considerably, from very shallow to 5 m or more. However, most soils cannot exert sufficient matric potential to pull groundwater up through the profile to the root zone if the water table lies more than 2.5 m below the surface. To ensure that the critical group and other biota are always exposed to groundwater contamination, we conservatively adopted a maximum soil depth of 2.5 m. We treat this value as an upper truncation limit to a normal distribution with a mean of 1.5 m and SD of 0.5 m.

For soil depths greater than 0.5 m, the soil profile is broken down into four layers, and concentrations are predicted using the regression equations based on SCEMR1 output. If the depth is less than 0.5 m, fluctuations in the water table would likely cause the entire profile to be flooded annually. In this case, we treat the soil compartment as a single layer, and predict concentrations using the methods described in Section 6.3.5. The soil depth distribution is truncated at its lower end at 0.1 m since shallower soils cannot support natural or cultivated crops.

The database of SCEMR1 results used to develop the regression equations was generated using soil depths of 1.0 m, 1.5 m and 2.0 m for each soil type and contamination pathway.

6.5.1.3 Soil Bulk Density, ρ_s
(kg dry soil·m^{-3} soil)

The soil bulk density is the mass of a unit volume of dry soil material that includes pore spaces. It is used in SCEMR1 in a variety of places and in BIOTRAC to calculate total nuclide concentrations or masses in a given soil layer (e.g., Equations (6.13), (6.22) and (6.47)). It is also used to calculate the rate constant for cropping losses (Equation 6.44).

Bulk density shows substantial variability among soil types, but varies little within a given type. We therefore adopt single, generic values of ρ_s = 1500, 1300, 1400 and 150 kg dry soil·m^{-3} soil for sand, loam, clay and organic soils respectively (Sheppard M.I. 1992). The surface litter layer, although organic, has a lower density than organic soils in the lower horizons. We set ρ_s = 80 kg dry soil·m^{-3} soil for the litter layer.

6.5.1.4 Soil Moisture Characteristic Curves

The soil moisture characteristic curve defines the relationship between the volumetric water content of the soil, θ, and its matric water potential, ψ_m. This relationship is used in the water flow subroutines of SCEMR1 to calculate values of θ or ψ_m for given values of the other parameter (Section 6.3.1.1). The shape of the curve depends primarily on the alignment and size distribution of pores, which vary considerably from soil to soil. However, for practical purposes, a single representative curve can be used to describe the main features of the relationship for a given soil type. The characteristic curves used in SCEMR1 for sand, loam, clay and organic soils are shown in Figure 6-9 (Hanks 1965, Beals 1985b, Sheppard M.I. et al. 1987). These curves were established for use in SCEMR1; values of θ (or ψ_m) at arbitrary points on the curves were found by interpolating between the data points.

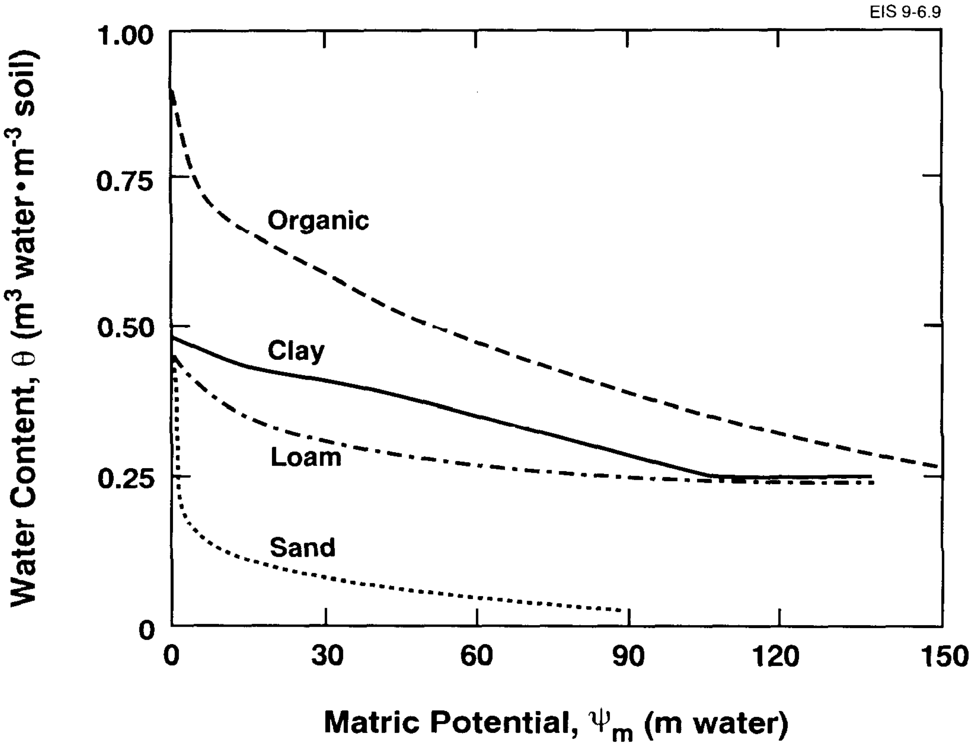


FIGURE 6-9: Soil Moisture Characteristic Curves for the Four Soil Types

6.5.1.5 Hydraulic Conductivity, K (m water·s^{-1})

In SCEMR1, the hydraulic conductivity relates the flow velocity through the soil profile to the water potential driving the flow (Equation (6.6)). The hydraulic conductivity is a complex function of the soil water content and a number of other soil properties, but it can be adequately represented by a fixed functional form for a given soil type. We generate values of K using Marshall's (1958) pore-interaction model and a solution technique developed by Green and Corey (1971). Figure 6-10 shows the hydraulic-conductivity/water-content relationships used for the postclosure assessment for each of the four soil types.

6.5.1.6 Field Moisture Capacity, θ_{FC} (m^3 water·m^{-3} soil)

The field moisture capacity is the largest water content that a soil can hold without drainage occurring. Water contents are initialized at θ_{FC} at the start of each SCEMR1 simulation (Equation (6.8)). Values of θ_{FC} are also used to calculate the amount of irrigation water added daily to the soil (Section 6.2.2). We assume that field moisture capacity for a given soil type can be adequately represented by a single value, and set θ_{FC} equal to 0.12, 0.32, 0.40 and 0.68 m^3 water·m^{-3} soil for sand, loam, clay

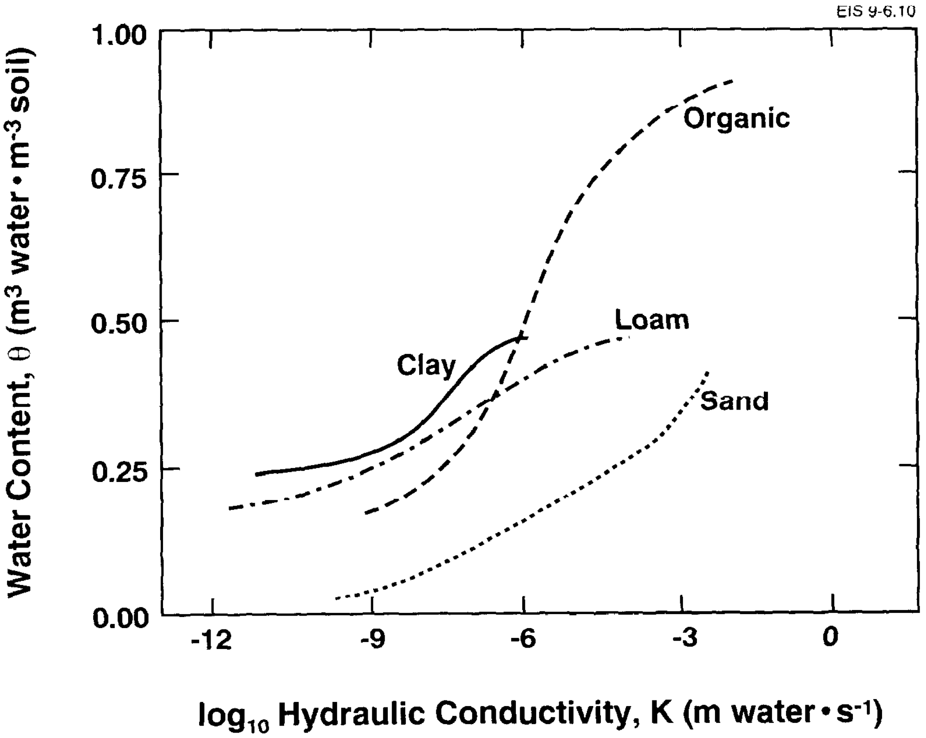


FIGURE 6-10: Relationship Between the Volumetric Soil Water Content, θ, and the Hydraulic Conductivity, K, for the Four Soil Types

and organic soils respectively (Sheppard M.I. 1992). This is the water content in each of the four soil types held at 1 m of water pressure.

6.5.1.7 Steady-State Water Content of Layer 4, θ (m^3 water$\cdot m^{-3}$ soil)

The steady-state water content of the soil refers to the content achieved when the daily patterns of water flows and contents predicted by SCEMR1 remain unchanged from year to year. Values of θ are used in Equation (4.11) to help calculate the pore-water concentration, C^i_{pw}, in layer 4 (Equation (4.14)), and in Equations (6.3) and (6.47) to calculate the total soil concentration in layer 4 for deep and shallow soil profiles respectively.

The steady-state contents are calculated by SCEMR1; they depend on soil type, but are essentially independent of the other soil parameters. Fixed values can therefore be assigned to each soil type, and we set θ equal to 0.12, 0.30, 0.32 and 0.40 m^3 water$\cdot m^{-3}$ soil for sand, loam, clay and organic soils respectively, consistent with SCEMR1 output.

6.5.1.8 Cation Exchange Capacity, CEC (mol$\cdot kg^{-1}$ dry soil)

The cation exchange capacity of a soil is the maximum concentration that can be achieved on the soil solids for nuclides in cation form. If SCEMR1 predictions of the soil solid concentration exceed CEC, the concentration is reset to CEC and the excess amount of nuclide is returned to solution (Section 6.3.1.2). Although some nuclides may reach the soil as anions, the CEC is always larger than the anion capacity, and is used conservatively. As with most other soil properties, we adopt fixed values of CEC for each soil type. We used values of 0.55, 0.60, 0.60 and 0.95 mol$\cdot kg^{-1}$ dry soil for sand, loam, clay and organic soils respectively (Buckman and Brady 1969). The CEC limits are applied to the root-zone concentrations so that these values reflect an effective exchange capacity for a 0.2-m-deep mineral or organic layer overlain by a 0.1-m-deep litter layer.

6.5.2 Meteorological Parameters

6.5.2.1 Daily Meteorological Inputs

SCEMR1 is driven by an annual cycle of daily average values for effective precipitation (precipitation minus runoff), air temperature, vapour pressure, wind speed and solar radiation. This information is used to predict daily evapotranspiration from the root zone and, subsequently, moisture conditions throughout the soil profile (Section 6.3.1.1). High-quality climatic data are available for many Canadian Shield sites through the Atmospheric Environment Service (AES) (Environment Canada 1979). Since we assume that the same sequence of daily meteorological values is repeated from year to year, the input data should reflect long-term average conditions on the Canadian Shield. We identified Geraldton, Ontario, as having a climate representative of Shield conditions. Daily precipitation and temperature data for Geraldton for individual years were obtained from AES archives, and compared with 30-a normals for the site (Environment Canada 1982b). The data for 1974 May to 1975 April most closely matched the long-term precipitation mean, and precipitation and temperature values for this period were used to drive SCEMR1 for our assessment simulations. The daily values for each of the five climate parameters (with total precipitation replacing effective precipitation) are shown in Figure 6-11. The mean annual temperature was 5°C.

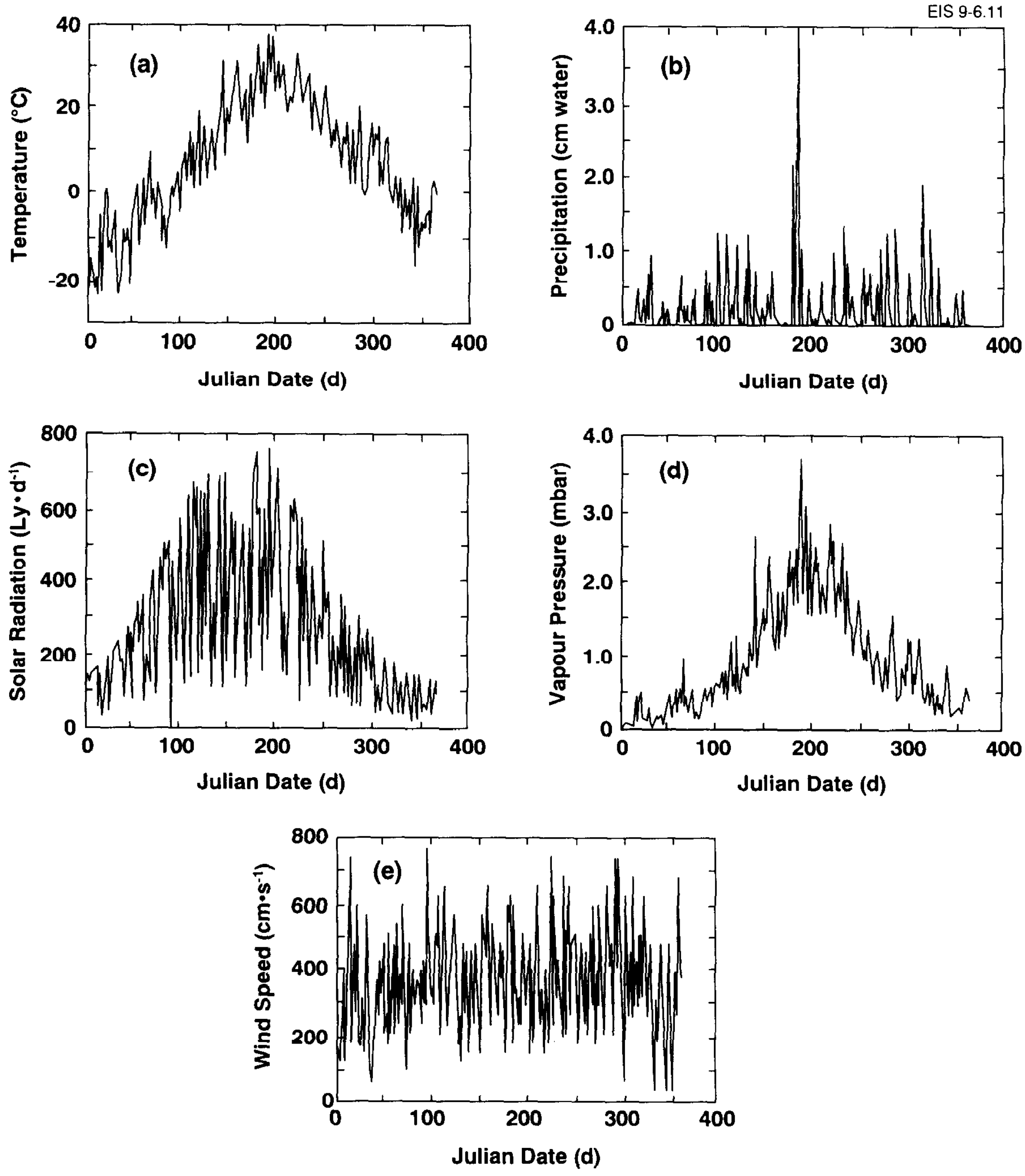


FIGURE 6-11: Annual Distribution of the Climatic Inputs Used to Drive SCEMR1. (a) Temperature, (b) precipitation, (c) solar radiation, (d) vapour pressure and (e) wind speed.

6.5.2.2 Effective Precipitation, Pe ($\text{m water}\cdot a^{-1}$)

Effective precipitation, which is used in SCEMR1 and elsewhere in BIOTRAC, is the only meteorological parameter treated probabilistically in BIOTRAC. We assume that the important aspects of its variability can be expressed through its total annual value (Section 6.3.2). In this capacity, it is used in regression Equation (6.1) to predict I_w, and in Equations (6.24) to (6.27) to predict the steady-state soil concentrations and the times at which they occur.

Effective precipitation is one component of the water balance, which also involves total precipitation, P ($\text{m water}\cdot a^{-1}$), and runoff, R ($\text{m water}\cdot a^{-1}$). Both P and R are used elsewhere in BIOTRAC, and they must be sampled in a consistent manner because all three parameters are closely linked. The sampling scheme and the PDFs adopted for P, R and Pe are discussed in detail in Section 9.1.3. Briefly, Pe in a given simulation is calculated from the water balance equation

$$Pe = P - R, \tag{6.54}$$

where values of P and R are drawn from their respective distributions (Section 6.5.2.1). Values of Pe calculated using Equation (6.54) are subject to the restriction $Pe \geq 0.2$ $\text{m water}\cdot a^{-1}$ because values smaller than this leave too little water available to support agriculture. Pe values generated in this way are normally distributed, with a mean of 0.47 $\text{m water}\cdot a^{-1}$ and SD of 0.064 $\text{m water}\cdot a^{-1}$. This distribution is very similar to the observed distribution of evapotranspiration (ET) values. This is to be expected since $Pe \sim ET$.

The Pe distribution describes the variation in annual effective precipitation across the entire Canadian Shield in Ontario, and is not specific to Geraldton. However, we assume that the Pe value sampled in each simulation is made up of daily values that show the same pattern as daily precipitation at Geraldton. As noted in Section 6.3.2, the daily Pe values, $(Pe)_d$, used in SCEMR1 simulations to generate the regression database were calculated from the annual value, $(Pe)_a$, using

$$(Pe)_d = (P)_{ds} \cdot (Pe)_a / (P)_{as} \tag{6.55}$$

where $(P)_{ds}$ and $(P)_{as}$ are the daily and annual precipitation respectively at Geraldton, for which $(P)_{as} = 0.78$ $\text{m water}\cdot a^{-1}$. The five annual Pe values used to generate the regression database were $(Pe)_a = 0.3$, 0.4, 0.6, 0.78 and 1.0 $\text{m water}\cdot a^{-1}$. The corresponding daily Pe values input to SCEMR1 were found from Equation (6.55) by multiplying the $(P)_{ds}$ values for Geraldton by factors $(Pe)_a/(P)_{as} = 0.38$, 0.51, 0.77, 1.0 and 1.28 respectively. Therefore, although the sampled Pe values for BIOTRAC simulations represent annual conditions anywhere on the Canadian Shield in Ontario, the distribution of daily values corresponds to the precipitation pattern at Geraldton.

6.5.3 Soil Partition Coefficient, Kd^i (m^3 water·kg^{-1} dry soil)

The soil solid/liquid partition coefficient is the ratio of the nuclide concentration on soil solids to the concentration in the pore water. It is used in SCEMR1 in equations such as (6.13) and (6.21) to describe sorption, the process that retards nuclide movement through the soil profile and allows soil concentrations to build up over time. Kd^i values are also used in regression Equations (6.24) to (6.27) to calculate the steady-state soil concentrations and the times to steady state in BIOTRAC.

Sorption is a complex phenomenon. It involves processes such as chemical precipitation and complexation, depends on variables such as soil pH and redox conditions, and is affected by the presence of microbes and colloids. The partition coefficient is a bulk parameter that describes the net effect of these processes and variables without modelling them in detail. It is an empirical parameter that is element- and soil-type-dependent. Partition coefficients implicitly assume that sorption is a reversible process and that the equilibrium between solid and liquid phases is reached instantaneously.

The information on Kd^i values is variable. A large amount of high-quality data is available for some nuclides and some soil types; for others, data are less complete. Sheppard M.I. et al. (1984a), Sheppard M.I. and Thibault (1990) and Thibault et al. (1990) reviewed and synthesized the published data for nuclides of importance in nuclear fuel waste management (Section 1.2.1). On the basis of the available information, they were able to define distributions for only about one third of the nuclides of interest. In these cases, the Kd^i values were lognormally distributed, and showed considerable variability as a result of the many processes and variables they included. The GM values for these nuclides as a function of soil type are listed in Table 6-8. Note that all or most of the values for carbon, iodine and technetium are based on literature data. Sheppard M.I. (1992) recommends a GSD of 10 for all nuclides and soil types to cover the range of reported values.

Kd^i values for the missing nuclides were determined by taking advantage of the correlation between Kd^i and the plant/soil concentration ratio, Bv^i. Bv^i is the ratio of nuclide concentration in plant material to that in soil, and quantifies the transfer of nuclides from soil to plants (Section 8.5.1.1). For a given nuclide, Kd^i is negatively correlated with Bv^i (Sheppard S.C. and Sheppard M.I. 1989). A nuclide with a high Kd^i value will be tightly bound to soil solids; little will be available in the aqueous phase for root uptake, and its Bv^i value will be low. Regression equations based on the nuclides for which good data were available were developed to describe the relationship between Kd^i and Bv^i for each soil type (Sheppard M.I. and Thibault 1990, Thibault et al. 1990). These regression equations were then used to predict GM values of Kd^i for the missing nuclides (Table 6-8). The GSD for these nuclides was set to 10.

Regression Equations (6.24) to (6.27) were derived assuming that Kd^i was expressed in units of L water·kg^{-1} dry soil. Values of Kd^i in these units can be obtained by multiplying the values listed in Table 6-8 by a factor of 1000.

TABLE 6-8

GEOMETRIC MEAN VALUES (m^3 water·kg^{-1} dry soil) OF THE SOIL SOLID/LIQUID PARTITION COEFFICIENT, Kd^i, DISTRIBUTIONS FOR EACH ELEMENT BY SOIL TYPE

Element	Sand	Loam	Clay	Organic
Ac	0.45	1.5	2.4	5.4
Am	1.9*	9.6*	8.4*	112.0*
Be	0.25	0.80	1.3	3.0
Bi	0.1	0.45	0.60	1.5
Br	0.015	0.050	0.075	0.18
C	0.005*	0.020	0.001	0.070
Ca	0.005	0.030	0.050	0.090
Cd	0.080*	0.040*	0.56*	0.80*
Cr	0.070*	0.030*	1.5	0.27*
Cs	0.28*	4.6*	1.9*	0.27*
Hf	0.45	1.5	2.4	5.4
I	0.001*	0.005*	0.001*	0.025*
K	0.015	0.055	0.075	0.20
Mo	0.010*	0.125	0.090*	0.025*
Nb	0.16	0.55	0.90	2.0
Ni	0.40*	0.30	0.65*	1.1*
Np	0.005*	0.025*	0.055*	1.2*
P	0.005	0.025	0.035	0.090
Pa	0.55	1.8	2.7	6.6
Pb	0.27*	16.0*	0.55	22.0*
Pd	0.055	0.18	0.27	0.67
Po	0.15*	0.40*	3.0	7.3
Pu	0.55*	1.2*	5.1*	1.9*
Ra	0.50*	36.0*	9.1*	2.4
Rb	0.055	0.18	0.27	0.67
Re	0.010	0.040	0.060	0.15
Sb	0.045*	0.15	0.25	0.55
Se	0.15	0.50	0.74	1.8
Si	0.035	0.11	0.18	0.40
Sm	0.245	0.80	1.3	3.0
Sn	0.130	0.45	0.67	1.6
Sr	0.015*	0.02*	0.11*	0.15*
Ta	0.220	0.90	1.2	3.3
Tc	0.0001*	0.0001*	0.001*	0.001*
Te	0.125	0.50	0.72	1.9
Th	3.2*	3.3	5.8*	89.0*
U	0.035*	0.015*	1.6*	0.41*
Y	0.17	0.72	1.0	2.6
Zr	0.60	2.2	3.3	7.3

* Values with an asterisk come from the literature; values without an asterisk are default values predicted using the plant/soil concentration ratio.

Because of its high organic content and greater microbial activity, the surface litter layer of the soil has a higher sorptive capacity than the underlying layers. In each SCEMR1 simulation, the Kd^i value of the litter layer was set ten times higher than the value chosen for the bottom three layers. The Kd^i value input to the regression model represents the sorptive properties of the lower layers. However, the results will reflect the Kd^i profile in the soil because the regression equations are simply a parameterization of SCEMR1 predictions.

The database of SCEMR1 results used to develop the regression equations was generated using Kd values of 10^{-5}, 10^{-4}, 10^{-3}, 10^{-2} and 10^{-1} m^3 water$\cdot$kg^{-1} dry soil for each soil type and contamination case. Above 0.1 m^3 water$\cdot$kg^{-1} dry soil, very little additional sorption occurs, and the regressions give good results for nuclides with higher Kd values.

6.5.4 Gaseous Evasion Rate from Soil, η_s^i (a^{-1})

In BIOTRAC, gaseous evasion rates describe the fraction of the soil inventory of gaseous nuclides that is lost to the atmosphere per unit time. Values are required in Equations (6.36), (6.38) and (6.39) to calculate the root-zone soil concentrations. Values are needed only for ^{14}C, ^{129}I and ^{79}Se. Of the remaining gaseous nuclides, ^{39}Ar, ^{3}H, ^{81}Kr and ^{85}Kr are not treated in the soil model. Radon concentrations are not reduced as a result of outgassing because soil concentrations for this nuclide are governed by its short half-life.

Evasion rates for iodine have been determined in a number of field experiments under a variety of conditions (Prister et al. 1977, Garland et al. 1987, Sheppard M.I. et al. 1987). The most relevant data for our purposes were obtained by Sheppard M.I. and Thibault (1991) in a long-term experiment carried out under Canadian Shield conditions. The experiment involved soils contaminated at the water table. These data suggest an evasion rate of 3.2×10^{-2} a^{-1}, which lies within the range of values reported in other studies. We assume η_s^I is lognormally distributed, and use a GSD that covers the full range of the published values (Sheppard M.I. 1992). Accordingly, the iodine evasion rate used for postclosure assessment is lognormally distributed with GM = 3.2×10^{-2} a^{-1} and GSD = 10.

Information on evasion rates for carbon that is not organically derived is very limited. Sheppard M.I. et al. (1991) measured the rate of ^{14}C loss from contaminated soils in outdoor lysimeter experiments. They found that η_s^C was not a strong function of soil properties, with a mean value of 12.0 a^{-1} for clay and loam, and 21.2 a^{-1} for sand and organic soil. Too few results were available to define a distribution. We assume that η_s^C values are distributed in the same way as η_s^I, and adopt a lognormal PDF with GSD = 10 for all soil types. We chose a GM value of 8.8 a^{-1}, which is slightly conservative with respect to the observed data.

Experimental measurements of the selenium evasion rate from bare soils indicate that $\eta_s^{S\bullet} \simeq 3.2 \times 10^{-2}$ a^{-1} (Zieve and Peterson 1981, 1984). Higher values are observed when the soil is vegetated (Lewis 1976). We assume that $\eta_s^{S\bullet}$ is lognormally distributed, and set its GSD equal to 10 to include the high loss rates associated with vegetation. The GM is 3.2 x 10^{-2} a^{-1}. The PDF for $\eta_s^{S\bullet}$ is therefore identical to that for η_s^I.

Evasion rates are also used to calculate air concentrations in the atmosphere model (Section 7.3.4) where they appear in units of s^{-1} (Amiro 1992b). In these units, η_s^i values for iodine and selenium are distributed lognormally with GM = 10^{-9} s^{-1} and GSD = 10. For carbon, η_s^c is distributed lognormally with GM = 2.8 x 10^{-7} s^{-1} and GSD = 10.

6.5.5 Cultural Parameters

Four of the parameters appearing in the soil model are determined to some extent by human cultural practices: the fraction of nuclides taken up by plants that is permanently lost from the soil, the probability of irrigating a garden or forage field, the time interval over which aerial irrigation is practised, and the probability that fresh lake sediment will be used for growing crops. In one sense, values for these parameters are well defined because present human behaviour related to these parameters can be studied and documented. On the other hand, the values are uncertain because they must reflect possible changes in the behaviour of the critical group over very long periods of time.

6.5.5.1 Fraction of Root Uptake Permanently Lost From Soil, ϵ (unitless)

The parameter ϵ is the fraction of nuclide mass taken up by plants through their roots that is permanently lost from the soil. It is required in Equation (6.44) in BIOTRAC to calculate the rate constant for cropping losses.

Values of ϵ can be quite variable, depending on agricultural management practices. Since low loss rates are conservative, we take our value from experience in nutrient-efficient farming in which recycling is practised to the extent possible. Most of the relevant information comes from the study of phosphorus for which cropping is the dominant loss mechanism. The permanent loss resulting from root uptake is about 5% of the crop inventory (L. King, North Carolina State University, personal communication, 1989). Other experts have only indicated that the permanent loss is less than 10% (P. Warman, Nova Scotia Agricultural College, personal communication, 1989). We have adopted a conservatively low value of 0.05 for BIOTRAC.

6.5.5.2 Probability of Irrigation, PI (unitless)

Few data are available to establish the probability that a garden on the Canadian Shield will be watered, or a forage field irrigated (Zach and Sheppard 1992). Sheppard S.C. (1985) has shown that irrigation can benefit crops grown on the Canadian Shield, and irrigation can become essential during hot, dry weather. Although irrigation in a formal sense is rare, gardens are commonly watered because people like to do so regardless of need or economics. Because irrigation can increase soil nuclide concentrations and the deposition rate to leaves, we set the probability of watering a garden in BIOTRAC at the conservatively high value of 0.9. On the other hand, forage fields are usually not irrigated because the potential benefits do not warrant the costs involved. We have adopted the small but finite value of 0.02 for the probability of irrigating a forage field. In BIOTRAC, the probability of aerial irrigation is expressed through a switch, PI, that determines whether or not irrigation will occur in any particular simulation. The exact source of irrigation water is then determined by another switch (Section 9.1.2).

6.5.5.3 Irrigation Period, t_{irr}
(a)

The irrigation period in BIOTRAC defines the time interval over which the garden has been watered or the forage field irrigated. Values of t_{irr} are required in Equation (6.39) to calculate soil concentrations for an irrigation source.

It is unlikely that irrigation of a specific plot of land on the Canadian Shield would continue uninterrupted for thousands of years. Land-use patterns, including irrigation, shift in response to human needs or climatic fluctuations. Moreover, prolonged irrigation is accompanied by salt buildup, which renders the soil useless for raising crops (Reeve and Fireman 1967).

There are no data available from which t_{irr} values for the Shield can be deduced (Zach and Sheppard 1992). However, it is reasonable to assume that values of a few years or a few tens of years are quite common, and extended periods over many human generations are possible but much less probable. This suggests that t_{irr} values are distributed lognormally, and we chose a GM of 100 a and a GSD of 4 for the postclosure assessment. This distribution is truncated at its lower end at 50 a, assuming that irrigation continues for at least one human generation. An upper truncation limit of 10 000 a is also applied, corresponding to the period over which quantitative methods must be used to assess the disposal concept (AECB 1987). This distribution applies to both the garden and the forage field.

6.5.5.4 Probability of Sediment Use, PS
(unitless)

The probability of sediment use defines the likelihood that the material making up the root-zone soil layer of the fields used by humans and other biota is fresh lake sediment. The sediment may have been dredged and applied to the fields, or the critical group may have moved on to sediments recently exposed by natural or intentional drainage. This probability is not meant to include the likelihood that the critical group will farm sediments that become available through the gradual filling in of the lake. Infilling is a slow process, and the sediments would mature and become indistinguishable from organic soils in the time required to generate an area sufficient for farming. Matured sediments were accounted for in BIOTRAC by setting the frequency with which the critical group farms organic soils (Section 6.5.1.1).

To our knowledge, there are no data on the frequency with which farmers on the Canadian Shield use fresh sediment for agricultural purposes. However, we believe that this practice is not common, and arbitrarily set the probability of sediment use to the low but finite value of 0.01. In BIOTRAC, this probability is expressed through a switch, PS, that determines whether or not sediment will be used as soil throughout a simulation.

6.5.6 Parameters Documented Elsewhere

The soil submodel contains a number of parameters that appear in other parts of BIOTRAC. These parameters are listed in Table 6-9, together with the sections where they are documented. In addition, SCEMR1 contains a number of parameters pertaining to atmospheric conditions and vegetation cover that have not appeared explicitly in the discussion given here. These parameters show little variability and have been assigned fixed values representative of Canadian Shield conditions. The values and data sources are documented by Sheppard M.I. (1992).

TABLE 6-9

SOIL SUBMODEL PARAMETERS COMMON TO OTHER SUBMODELS AND THEIR SECTIONS OF DOCUMENTATION

Parameter	Equation	Section Where Documented
Dry deposition velocity, Vd ($m \cdot a^{-1}$)	6.16 and 6.17	7.5.4.1
Precipitation rate, P (m water$\cdot a^{-1}$)	6.16 and 6.17	9.1.3
Washout ratio, Wr (unitless)	6.16 and 6.17	7.5.4.2
Plant/soil concentration ratio, Bv^i (($mol \cdot kg^{-1}$ wet biomass)/($mol \cdot kg^{-1}$ dry soil))	6.42 to 6.44	8.5.1.1
Crop yield, Y_j (kg wet biomass$\cdot m^{-2} \cdot a^{-1}$)	6.42 to 6.44	8.5.8.1

6.6 MODEL VALIDATION

In the case of the soil model, validation means demonstrating that the processes responsible for nuclide transport in soils are adequately formulated and that the predicted soil concentrations are realistic.

The model containing the subroutines on which SCEMR1 is based, TEHM (Huff et al. 1977), has been experimentally validated using data from contaminant transfer in a watershed (Begovich and Jackson 1975, Munro et al. 1976). SCEMR1 predictions have been continuously verified against those of TEHM using a test data set supplied by Oak Ridge National Laboratory. This has ensured that the basic solution techniques of the original code were preserved during the changes made in developing SCEMR1.

The SCEMR1 model itself has been validated experimentally in two series of tests involving the migration of several nuclides in different soil types. All the tests were conducted in outdoor lysimeters 0.13 m in diameter and 0.40 or 0.64 m deep. The bottoms of the lysimeters were sealed and the water tables were maintained artificially. Plants were grown in the lysimeters in some tests but not in others, but in all cases the lysimeters were exposed to natural atmospheric conditions. Nuclides were injected into the soil profile in discrete layers at different depths, and were left undisturbed over the growing season. The cores were then sectioned and the total nuclide concentrations were analyzed in 0.02-m-thick sections. These observations were compared with the predictions of SCEMR1, which was run in a five-layer version, with inputs derived from the experimental data whenever possible. In addition, observed evapotranspiration amounts were compared with the SCEMR1 predictions to validate the water flow subroutines in the model.

In the first series of experiments, uranium and chromium were placed in the unsaturated zone 0.15 and 0.3 m below the surface of a coarse sand and an organic clay loam (Sheppard M.I. et al. 1984b). In the second series, technetium, uranium and iron were injected above, at, and below the water table in an organic clay loam (Sheppard S.C. and Evenden 1985). In this case, the nuclides experienced a redox gradient in migrating across the water table. This was simulated in SCEMR1 using a layered-partition-coefficient profile.

In general, SCEMR1 performed well in predicting the redistribution of nuclides throughout the soil profile. In most cases, the model predictions for both evapotranspiration and nuclide concentration as a function of depth agreed with observations within experimental error. Furthermore, acceptable results were obtained using a constant Kd profile, even in the second series of tests where the aeration conditions varied strongly with depth. This supports our use of a constant Kd value in the lower three layers of SCEMR1 when the model is used for the postclosure assessment.

These and further validation studies (Sheppard M.I. and Hawkins 1991a) have tested the soil model under a wide range of conditions. The comparisons involved soils with very different chemistries, hydraulic properties and aeration conditions. They included eight nuclides with very different sorptive properties. The model was successful in simulating both upward and downward migration from a variety of injection points. The observational data were collected under conditions that are typical of the Canadian Shield during the growing season and throughout several complete annual cycles of freezing and thawing. We conclude that SCEMR1 adequately simulates nuclide behaviour in soils, and that it produces realistic soil concentrations on annual time scales.

The final step in the validation of our soil model is to show that the regression model used in BIOTRAC is able to duplicate SCEMR1 predictions. To do this, we compared the normalized steady-state concentrations, $(Css)^i$, and times to steady state, $(tss)^i$, predicted by the two approaches. For the groundwater case, $(Css)^i_G$ values calculated from the regression model were always within a factor of six of the values predicted by SCEMR1; the $(tss)^i_G$ values were always within a factor of five. These factors represent extreme cases, and the agreement was usually better. For the irrigation/

deposition case, $(Css)^i_{ID}$ and $(tss)^i_{ID}$ values calculated from the regression equations were always within factors of two and three respectively of the SCEMR1 values. Again, the agreement in general was usually better. Similar agreement was obtained between the SCEMR1 predictions of the root-zone soil concentration at any time, and the corresponding predictions of the time-dependent form of the regression equations (Equations (6.28), (6.29) and (6.30)). In all cases, the predictions of the regression equations were scattered randomly about the SCEMR1 results, reflecting the statistical nature of the regression fit. A comparison of time-dependent soil concentrations as predicted by the two methods for two specific sets of parameter values is shown in Figure 6-5. Given the many other sources of uncertainty in the soil model and in BIOTRAC as a whole, we consider this level of agreement between the regression model and SCEMR1 to be satisfactory.

6.7 MODEL DISCUSSION

6.7.1 Assumptions

A number of assumptions were made in deriving the soil model. In this section, we restate, evaluate and review the main assumptions and discuss their effects on the model predictions.

1. *The soil profile is described by a surface litter layer with the properties of an organic soil, underlain by three layers of mineral or organic soil with uniform properties.* Canadian Shield soils commonly show an organic-enriched surface layer. The high Kd^i values assigned to this layer allow nuclide concentrations to build up to conservatively high values. In a real system, soil properties vary somewhat with depth; however, our validation studies have shown that nuclide concentrations in a layered soil profile can be accurately predicted when soil properties are held constant with depth (Section 6.6). By breaking the soil down into layers, we can model the slow migration of nuclides through the profile and predict a depth-dependent concentration. On the other hand, our assumption of uniform and instantaneous mixing within each layer is conservative since it allows for the fastest possible rate of migration through the soil. A well-mixed root zone 0.3 m deep is consistent with cultivation of a garden or agricultural field, and accounts for bioturbation caused by burrowing animals and growing roots.

2. *Nuclide transport through the soil profile occurs by advection only.* This is a reasonable assumption because according to our estimates all the other transfer mechanisms together, including vapour diffusion of volatile nuclides, account for less than 5% of the mass transport.

3. *The model is one-dimensional.* The model considers water and nuclide flows in the vertical direction only, the primary direction of flow in unsaturated soils. Lateral flow will occur at the surface in the form of runoff following precipitation events. We account for this flow in the water balance by driving the model with an effective precipitation equal to precipitation minus

runoff. The saturated region below the water table may also experience lateral flow, which carries away any nuclides draining from the profile.

4. *Soil concentrations are not depleted when nuclides are lost through surface runoff, wind erosion or the suspension of particulate matter.* These processes are difficult to model without a specific site, and omitting them results in an overestimate of soil nuclide concentrations. Similarly, only a small fraction of the nuclide mass taken up by plant roots is permanently removed from the soil. By largely ignoring these depletions, we implicitly model the recycling of the nuclides that would occur when decayed plant material and animal and human wastes are returned to the soil.

5. *Nuclide sorption can be modelled using the partition coefficient, Kd.* Because nuclide releases from the geosphere vary slowly with time, the assumptions concerning reversibility and instantaneous equilibrium implicit in the Kd are reasonable. Furthermore, the Kd is an empirical parameter that accounts for the effects of many chemical and biochemical processes that individually are not well understood. The Kd is invariably used in assessment models because of the lack of a good practical alternative.

6. *The deposition of airborne nuclides to the soil is modelled analogously to irrigation.* We assume that the annual depositional flux to the surface is contained in a volume of water equal to the annual irrigation volume, and is applied in the same daily pattern as irrigation from May to September only. Although this appears to be artificial, it is actually a reasonably good representation of how deposition occurs. In nature, a considerable fraction of the total annual deposition reaches the soil with water during precipitation events, which are episodic and of short duration. Rates are much lower in the intervening dry periods. Deposition is effectively confined to the summer since many of the nuclides deposited in winter to the snowpack would run off with the spring melt. It is therefore reasonable to model deposition using the irrigation formulation. It is also conservative since the entire annual flux is deposited in the growing season from May to September.

7. *The pattern of daily precipitation is assumed to be the same in every simulation, regardless of the sampled value of the total effective precipitation.* Nuclide migration through the soil profile depends on daily water flows, but test simulations using SCEMR1 have shown that the long-term concentrations are essentially independent of the exact daily precipitation pattern as long as the pattern is typical of Canadian Shield conditions. The model does not cover patterns characterized by droughts or heavy rains extending over periods of months or more.

8. *The response function form of the soil model, which is based on the regression equations, is an adequate representation of SCEMR1 results.* This assumes that the variability in soil nuclide concentrations is controlled by variations in four key parameters: soil type, soil depth,

annual effective precipitation and nuclide partition coefficient. This has been verified through a sensitivity analysis of SCEMR1 (Sheppard M.I. and Bera 1984). It also assumes that the regression equations provide a satisfactory statistical fit to SCEMR1 results. This was established through a thorough comparison of the predictions of the two models (Section 6.6). Finally, it assumes that the root-zone soil concentrations increase with time in the exponential fashion described by Equations (6.28) to (6.30). SCEMR1 results show this to be the case for irrigation and deposition sources (Figure 6-4). For groundwater sources, the time-dependent concentration curve is sigmoidal, and the exponential form overestimates concentrations at short times.

9. *We have made many assumptions in defining the modes through which the soil becomes contaminated; most of these assumptions lead to an overestimate of soil concentration.* We assume that terrestrial discharge always occurs, and we generally place the most highly contaminated discharge zones beneath the fields that lead to the highest consequences. Sediments are distributed among fields in the same way when sediments are used as soil, a practice that is assigned a conservatively high probability of occurrence. Furthermore, all the soil types have a leaf litter layer that tends to enhance nuclide concentrations in the root zone. The assumption that irrigation maintains the soil water content at field capacity results in a conservatively large amount of water (and therefore of nuclides) being applied to the soil. We also assume that irrigation is practised continuously over long periods of time, which results in an overestimate of soil concentrations. Organic soils, which are highly sorbing and accumulate nuclides readily, are assigned a relatively high frequency of occurrence. Finally, our distribution of soil depths tends to overestimate the frequency of occurrence of shallow soils, and concentrations from groundwater discharge into such soils are calculated conservatively (Section 6.3.5).

10. *The soil concentrations predicted by SCEMR1 reflect seasonal trends in several ways.* Meteorological conditions are specified on a daily basis, irrigation occurs during the growing season only, and the seasonal cycle of leaves appearing and senescence is modelled. On the other hand, SCEMR1 does not recognize the winter season. Precipitation is assumed to fall as rain throughout the year, and the ground is assumed not to freeze. Snow is not allowed to accumulate, and spring melt and runoff are not modelled. Under these circumstances, water flow and nuclide transfer remain active in the soil profile throughout the year. However, SCEMR1 simulations in which these processes were suspended during the winter, and a high precipitation rate imposed in the spring to simulate runoff, produced long-term predictions similar to those obtained in the absence of a winter season. This was also confirmed by simulations in which SCEMR1 was started up at different months of the year (Sheppard M.I. et al. 1985). It appears that the low rates of precipitation and evapotranspiration that occur in winter are unable to drive any significant redistribution of

the nuclides. Disregarding snowmelt, with its attendant high leaching rates, is likely conservative.

11. *The model includes temporary, or seasonal, wetlands implicitly.* Although the model does not include such wetlands explicitly, they are included as part of the treatment of shallow soils, which may be subject to seasonal flooding with groundwater due to fluctuation in the water table. Permanent wetlands are considered part of the surface water model (Chapter 5).

12. *The model does not allow for the long-term evolution of the soil profile, or for major changes in the meteorological conditions driving the system.* Although pedogenesis is not modelled, the soil parameters are based on data from geologically young to geologically old soils on the Canadian Shield, and so reflect soils in various stages of development. In the absence of human interference, the climate is not expected to show major departures from current conditions until the next glaciation occurs. Small fluctuations are accounted for through our distribution for effective precipitation (Section 6.5.2.2). The model appears to be relatively insensitive to minor changes in weather patterns throughout the year. Water table fluctuations are accounted for in calculating nuclide concentrations in shallow soils (Section 6.3.5); they are not likely to influence the root-zone concentrations in soils more than 0.5 m deep. Isolated events such as flooding, which are not modelled explicitly, are effectively included in the irrigation case. Soil concentrations induced by flooding would likely be lower than those caused by irrigation. Flooding is an episodic event that would probably involve surface water, whereas irrigation is assumed to occur frequently over long periods of time, and would often involve well water. We conclude that, apart from glaciation, the model accounts for the effects of environmental change on the predicted soil concentrations.

6.7.2 Evaluation

Contaminant transport in soils is a well-studied discipline. The underlying concepts are reasonably well understood and have been translated into a variety of predictive models. We have based our model for the postclosure assessment on a detailed mechanistic code, SCEMR1, which simulates the movement of water and nuclides on a daily basis. This is a more complex approach than is commonly used in assessment models, but is more realistic, and produces results that compare favourably with those of the simpler models (Section 11.5). The predictions of the model have been successfully validated against observations over a wide range of conditions. This suggests that the model contains all of the relevant processes, that the processes are adequately simulated, and that the model predictions are realistic.

The model was specifically designed for the postclosure assessment. In particular, the simulation of daily water and nuclide flows allows the model to predict root-zone soil concentrations arising from the contamination of the profile at the water table by an underground source. The model accounts for all contamination modes, including terrestrial discharge, aerial irrigation and atmospheric deposition, and formulates each pathway in a conservative way. SCEMR1 predictions have been cast into a simple regression model suitable for a probabilistic assessment. This model interfaces smoothly with the geosphere model and the other three submodels of BIOTRAC.

The parameter values and distributions required by the soil model were derived from the best available observational data from the Canadian Shield (Section 6.5). Most of the information was extracted from the literature, although some was supplied by our own studies (e.g., Sheppard M.I. et al. 1987, Sheppard M.I. and Thibault 1991). Relevant, high-quality data were available for most parameters, with the exception of gaseous evasion rates, partition coefficients for some nuclides, and several cultural parameters. The missing partition coefficient values were generated from the known relationship between sorption and root uptake. Values for the gaseous evasion rates and cultural parameters were determined from literature data and through expert opinion, with due regard for conservatism when information was scarce.

We conclude that the soil submodel and its associated database provide a reasonable description of nuclide behaviour in soils of the Canadian Shield, and that it will not underestimate consequences for humans and other biota when used for the postclosure assessment of the concept for disposal of Canada's nuclear fuel waste.

7. ATMOSPHERE SUBMODEL

7.1 THE ATMOSPHERE COMPARTMENT

The atmosphere is a potential recipient of nuclides that escaped from the vault. It receives its nuclide load via suspension from contaminated water bodies, soils and vegetation. The atmosphere dilutes nuclides reaching it, but is also very effective at redistributing them. Regions away from the immediate vicinity of the discharge zone can become contaminated very quickly through the action of atmospheric transport and deposition. Atmospheric processes must therefore be taken into account in assessing the consequences of a geological disposal facility. This chapter summarizes the model developed to treat the atmospheric pathways for the postclosure assessment. The information was extracted from the atmosphere submodel report (Amiro 1992b), which contains further details. The model has also been published in the open literature by Amiro and Davis (1991).

Because we focus on dose prediction for the critical group (AECB 1987), we do not consider the long-range transport of nuclides by the atmosphere. Atmospheric dispersion will cause airborne nuclide concentrations to decrease with increasing distance from the discharge zone. Humans located away from the source will be at lower risk than the critical group. This is also true for non-human biota (Section 1.5.4). Accordingly, we consider only local atmospheric transport within the air compartment that overlies the discharge lake and the various fields used by the critical group.

Furthermore, we focus on the layer of air fairly close to the ground, where the critical group and other biota are located. We calculate concentrations at a height of 1.5 m above the ground, the height at which human inhalation occurs. We assume that concentrations at this height are also representative of concentrations experienced by plants and animals at large. A single concentration is therefore used as input to the food-chain and dose submodel to calculate human doses due to inhalation and immersion, and deposition to vegetation.

As noted in Section 2.1, the Canadian Shield in Ontario experiences a humid continental climate, with extremes of temperature and sufficient precipitation to support agriculture. Temperature, precipitation and evapotranspiration all show a general increase from northwest to southeast across the region. The air is fairly clean, particularly away from urban areas, with annual average dust loads of about 20 $\mu g \cdot m^{-3}$ air. Despite these general patterns, many sites show local anomalies that depend on their topography and their location with respect to the Great Lakes. At a given location, the state of the atmosphere varies strongly from day to day and from season to season, but exhibits relatively little variability from year to year. Extreme climatic conditions are similar across the region.

The atmosphere model is essentially generic. It simulates processes characteristic of the atmosphere above the Shield in Ontario, and its parameter values reflect present meteorological conditions. In some cases where parameters show little spatial variability, fixed values have been adopted in place of distributions. For convenience, and to be consistent with the site-specific approach taken in the geosphere model, some values were set to be representative of the WRA. We have also made some assumptions concerning the geometry and physical locations of the discharge lake and the various fields used by the critical group and other biota (Sections 7.3.1.1 and 7.3.1.4). Such information is required to quantify the effects of atmospheric dispersion where one field acts as a source of contamination and a second as a receptor.

The atmosphere model is driven by the nuclide concentrations in soils (or sediments), surface waters and vegetation (Figure 7-1). These concentrations are used to calculate the rate at which nuclides are suspended into the atmosphere by a variety of mechanisms. This information is combined with a dispersion factor to estimate concentrations in air. These concentrations are used directly in the food-chain and dose submodel to calculate internal doses to humans resulting from inhalation, and external doses from air immersion. They are also used to calculate similar doses for non-human biota (Chapter 13). Airborne nuclide concentrations are used to calculate the rate at which nuclides are transferred from the atmosphere to underlying surfaces. This information is employed in estimating concentrations in the soil and vegetation following atmospheric deposition.

7.2 QUALITATIVE DESCRIPTION OF THE ATMOSPHERE MODEL

7.2.1 Atmospheric Processes

The atmosphere model considers three main processes: suspension of nuclides into the atmosphere, dispersion of the nuclides by atmospheric turbulence,

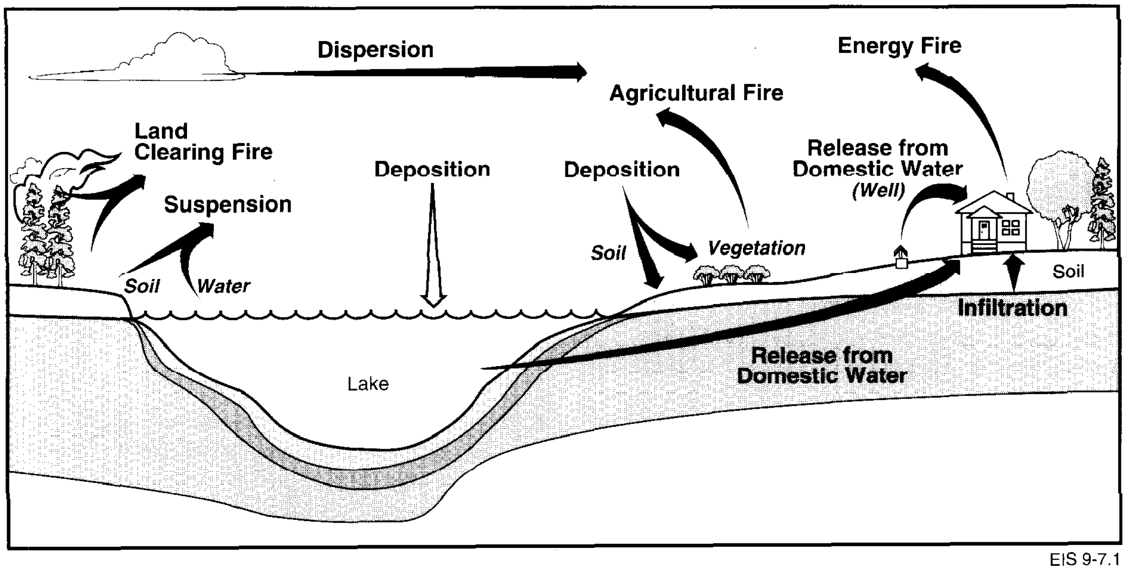


FIGURE 7-1: Transport Processes in the Atmosphere Considered in BIOTRAC. Closed arrows indicate explicit consideration in BIOTRAC, and open arrows indicate implicit consideration.

and deposition back to the underlying surface. The need to model suspension as a primary source of contamination is unique to assessments that deal with ground sources. In most cases of atmospheric pollution, contaminants are released directly to the atmosphere from industrial stacks, for which the source term can be readily estimated.

Suspension, dispersion and deposition are all complex processes, but suspension is distinguished by the very large number of mechanisms through which it operates. Many of these are natural mechanisms: soil erosion by wind action, forest fires, gaseous emissions from soil and water bodies, and so on. Human activities can be equally important: wood burning for energy, soil cultivation and agricultural fires may make a significant contribution to the total nuclide load in the atmosphere. The model must therefore make some assumptions about human cultural behaviour in order to arrive at reasonable and conservative air concentrations. In particular, some activities carried on inside houses can raise indoor air concentrations above outdoor levels. For instance, the use of contaminated water for showering or in humidifiers could lead to the release of volatile nuclides to the indoor air. Accordingly, we calculate both indoor and outdoor nuclide concentrations in air, taking account of the appropriate processes in each case.

Many nuclides that reach the atmosphere are attached to particulate matter. These are treated generically as particles, using parameter values that are not element-specific. Nuclides that exist in gaseous form are generally treated individually, taking account of element-specific behaviour. Apart from this breakdown into particulate and gaseous forms, the chemical species of a contaminant is not modelled in most cases. For some elements there are only a few possible forms, whereas for others many species may exist. Wherever possible, we model the most mobile form or combination of forms to ensure that the atmospheric nuclide concentrations are not underestimated.

Amiro (1985) identified and evaluated mechanisms by which nuclides could be suspended into the atmosphere. The mechanisms that could conceivably contribute significantly to the air concentration have been included in the atmosphere model. Where it is practical, and where the theoretical framework and observational data exist, mechanisms have been modelled explicitly; otherwise, they have been included implicitly in a bulk formulation that describes several mechanisms. Several suspension processes are included in the atmosphere submodel.

1. Mechanisms that suspend particulate material from terrestrial sources. These include natural phenomena such as wind erosion of soil and pollen release from vegetation, and human activities such as soil cultivation, driving on dusty roads and construction. These processes are modelled collectively using a mass loading approach.

2. Mechanisms that suspend particulate material from aquatic sources. These include processes such as wave breaking and bubble bursting, which are also modelled collectively.

3. Evasion of gases from terrestrial sources. Gaseous evasion from soil and vegetation, caused by inorganic and microbial processes, operates primarily outdoors, but radon is allowed to diffuse into buildings and contribute to the indoor concentration.

4. Evasion of gases from aquatic sources. As above, this mechanism operates largely outdoors, but ^{14}C, ^{129}I and ^{222}Rn are also assumed to be released as gases from water brought into the house for domestic use.

5. Nuclides may be released to the atmosphere when wood or peat is burned in stoves for space heating, when stubble is burned on agricultural fields, and when forests or peat are burned in forest fires or to clear lands.

Most suspension mechanisms are not active continuously, but occur over short periods of time when certain conditions are met. For example, significant soil erosion occurs only when wind speeds are high. In general, we obtain annual average air concentrations by averaging the source terms from each episode over a one-year period. In doing this, we assume that a high source term acting over a short period of time (when the physical system is in some particular state) has the same effect as a low source term applied to a system characterized by its annual average properties. The effects of this and other assumptions on the model output are discussed in Section 7.7.1.

Once in the air, nuclides are dispersed according to the prevailing conditions of atmospheric turbulence, which depend on local values of wind speed and atmospheric stability. Airborne nuclides are also subject to deposition back to soils or crops, and may eventually reach humans through the food chain. Deposition to water bodies is not considered in the model. As noted in Section 5.2, the discharge lake receives the entire nuclide flow out of the geosphere, and surface water concentrations are usually not reduced when nuclides are transferred to the atmosphere. Nuclides input to the water column through deposition would duplicate mass already in the system (Section 9.3).

The deposition process and the rate at which it occurs depend partly on whether precipitation is falling. We have separate models of dry and wet deposition covering both particulate and gaseous nuclides. We assume conservatively that air concentrations are not reduced when nuclides are lost to the underlying surface.

7.2.2 Modelling Approaches

Our understanding of the various suspension mechanisms is uneven. Some processes have been studied extensively and are reasonably well understood. For these, it is possible to formulate models to predict the nuclide flux to the atmosphere, which can be coupled to a dispersion model to calculate air concentrations. Occasionally, sufficient information exists to allow the flux models to provide a detailed, mechanistic representation of the suspension process, e.g., gaseous evasion of radon from soils (Rogers et al. 1980). More often, the fluxes are expressed using simple bulk parameters that empirically account for the effects of several processes. We have taken this approach in modelling the fire pathways, gaseous evasion from soils and water (with the exception of ^{129}I evasion from water), and the release of gaseous nuclides from domestic water to indoor air. For still other suspension mechanisms, too little information is available to allow fluxes to be estimated. In these cases, we have assumed that airborne nuclide concentrations are simply proportional to the concentration in the source compartment. The proportionality constants, which can be interpreted as mass loading parameters, are empirically derived and implicitly account for dispersion and suspension. We have used this approach to model the suspension of particles from terrestrial and aquatic sources, the gaseous evasion of ^{129}I from water, and the diffusion of radon from soils into the indoor air compartment.

Atmospheric dispersion has a long history of study and is reasonably well understood. A variety of validated dispersion models is available to treat a given source configuration. Most of the suspension pathways that we consider involve a ground-level area source: a contaminated field or lake. For these cases, we have based our dispersion model on the trajectory simulation approach (Wilson 1982a), which was recommended by Culkowski (1984) for surface emissions. Special dispersion relations were used for the wood-burning-for-energy pathway, which involves an elevated point source influenced by building-induced turbulence, and for gaseous releases from domestic water, which involve dispersion inside a building.

Atmospheric deposition has been studied extensively, and mechanistic models are available to treat parts of the process under special circumstances (Sehmel 1980). However, simpler models are invariably used for assessment purposes. We have used deposition velocities to model the dry deposition process (Sehmel 1980) and the washout ratio to treat wet deposition (Slinn 1978).

The models that we have employed to address the various atmospheric pathways vary considerably in complexity. Each was selected only after a careful evaluation of the models available in the literature. Each reflects our current level of understanding of the process and the amount and quality of the data available to model it. Furthermore, each was formulated taking into account the need for efficient, generic assessment models able to predict annual average airborne nuclide concentrations.

Transport processes in the atmosphere occur very rapidly, with time scales on the order of minutes to hours. Nuclide concentrations in air therefore adjust very quickly to changes in the concentration of the source compartments. The transient aspects of atmospheric processes are not important when calculating annual average concentrations. Accordingly, the atmosphere model is expressed as an equilibrium steady-state model. The rapid time scales also mean that radioactive decay and ingrowth need not be considered in the model.

7.3 MATHEMATICAL FORMULATION OF THE ATMOSPHERE MODEL

In the equations of the atmosphere submodel, soil concentrations of nuclides are designated by C^i_s, which relates to the soil submodel (Table 6-6). However, soil concentrations may also be based on sediment concentrations, C^i_{se}, as calculated in Equation (5.15). The probability of sediment use as soil, PS, is discussed in Section 6.5.5.4.

7.3.1 Dispersion Factors

Airborne nuclide concentrations, $(C^i_a)_k$ ($\mathrm{mol \cdot m^{-3}}$ air), for a number of pathways, k, are calculated using an expression of the form

$$(C^i_a)_k = (Q^i)_k \cdot (DISP)_k \quad . \tag{7.1}$$

Here, $(Q^i)_k$ is the flux of nuclide i from the source compartment to the atmosphere via pathway k, and is the quantitative expression of the suspension process. $(DISP)_k$ is the atmospheric dispersion factor corresponding to pathway k. If the source is distributed over an area, $(Q^i)_k$, has units of $\mathrm{mol \cdot m^{-2} \cdot s^{-1}}$ and $(DISP)_k$ has units of $\mathrm{s \cdot m^{-1}}$. For a point source, $(Q^i)_k$ has units of $\mathrm{mol \cdot s^{-1}}$ and the units of $(DISP)_k$ become $\mathrm{s \cdot m^{-3}}$ air. Since dispersion relations are common to several pathways, we derive them here for future reference.

7.3.1.1 Ground-Level Area Sources

Several suspension pathways involve a ground-level area source such as a contaminated field or lake. Material suspended from each point of such a source results in an air concentration that depends on the source flux, the prevailing dispersion conditions and the distance from the source to the receptor. The total air concentration at a given downwind distance equals the sum of contributions from all parts of the source, and depends on the geometry of the source, the relative locations of the source and receptor, and the wind direction. These factors would not be known for a generic site. We assume that the source is circular in geometry, with the receptor located at its centre (Figure 7-2). With the further assumption that the flux is constant across the source, this implies that air concentrations

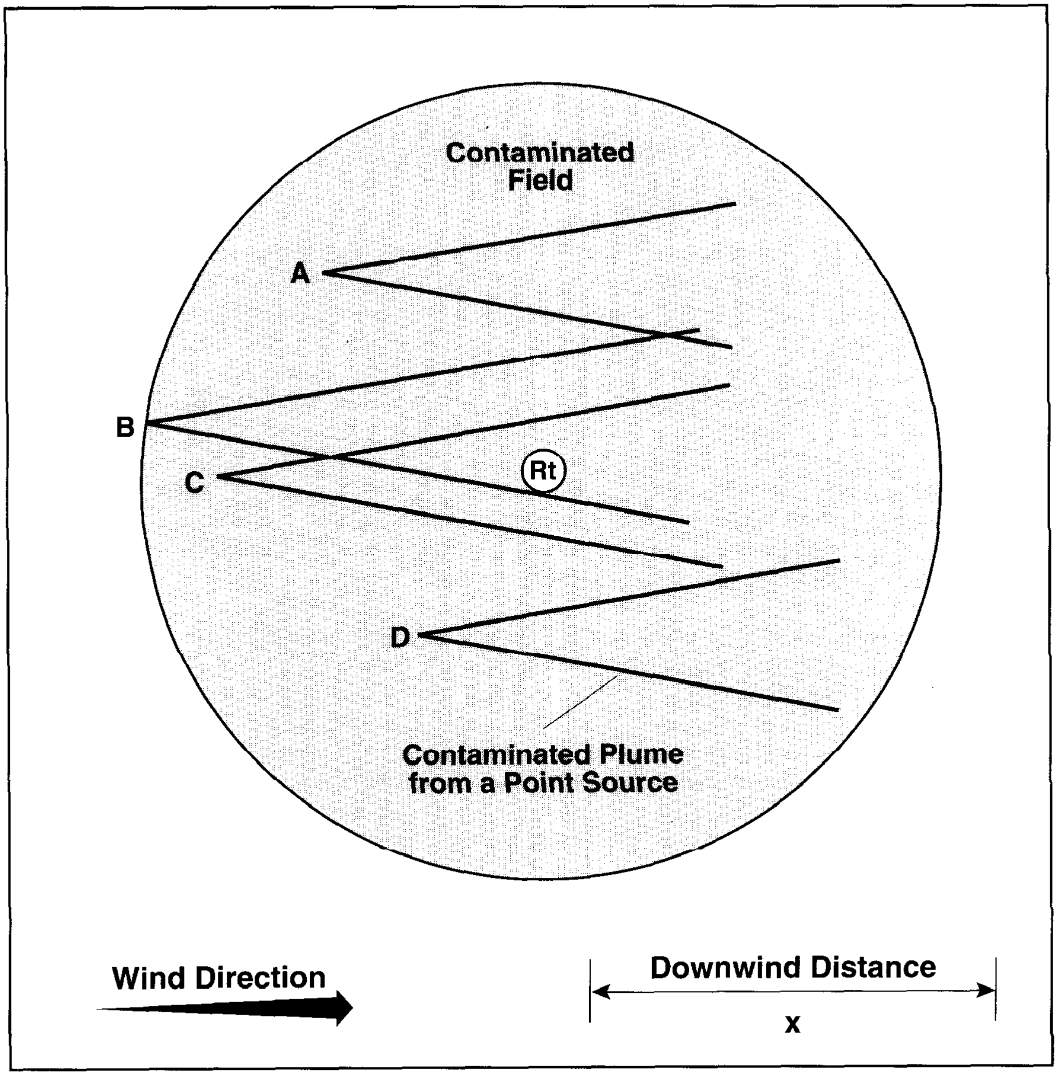


FIGURE 7-2: Air Concentration at a Receptor, Rt, at the Centre of a Circular Field. Points B and C contribute to the concentration at Rt, but points A and D do not. The downwind distance from the receptor, Rt, to the upwind leading edge of the source is designated by x.

are independent of wind direction. Moreover, a centrally located receptor is conservative. Concentrations at a fixed point on the perimeter of the source would be somewhat higher than in the centre for some wind directions, but annual average concentrations would be lower since winds typically blow from a given direction only a small proportion of the time.

Traditionally, a class of models known as Gaussian models has been used to simulate atmospheric dispersion for assessment purposes (Turner 1970).

However, the Gaussian models are applicable only when the turbulence properties of the atmosphere are homogeneous. They are therefore not suitable for a ground-level source because meteorological parameters vary strongly with height near the ground. Instead, we have used a statistical-trajectory model that was developed for a ground-level area source (Wilson 1982a, 1982b). Culkowski (1984) recommended this model for evaluating low-level waste disposal facilities.

For a surface source, Wilson's model provides normalized airborne nuclide concentrations, or dispersion factors, C_a^i/Q^i ($s\cdot m^{-1}$), as a function of wind speed, atmospheric stability, receptor location and a surface roughness parameter, z_o (m), which is a scale height related to the height of roughness elements (vegetation, buildings, etc.) at the location of interest. We used wind speed and atmospheric stability data from the WL site (Davis and Reimer 1980) to calculate an annual average value of C_a^i/Q^i for a receptor height of 1.5 m and various distances, x (m), of the receptor from the upwind leading edge of the source (Figure 7-2). Since our receptor is assumed to be at the centre of a circular source, the concentrations calculated for a particular value of x are the concentrations that would arise from a source with area, A (m^2), equal to πx^2.

We derived values of C_a^i/Q^i for a number of discrete values of A for both terrestrial and aquatic surfaces. A separate treatment for the two types of sources is necessary because z_o for a lake is quite different from z_o for a vegetated area. For the terrestrial calculation, we used $z_o = 0.15$ m, a value that lies between the values for farmland and forest (Monteith 1973). For the aquatic calculation, we set $z_o = 1.3 \times 10^{-4}$ m, a value typical of lakes (Brtko and Kabel 1978). Analytical curves were then fitted to the values (Figure 7-3). For the terrestrial source, the data can be represented by

$$(C_a^i/Q^i)_T = 4.87\cdot A_T{}^{1/8} - 3.56 \qquad (7.2)$$

where A_T is the terrestrial source area (m^2), which may be represented by various fields (Section 9.1.1). Equation (7.2) provides a good fit to the data over the range of field areas that could be expected to become contaminated in the vicinity of a discharge zone (Section 9.1.1.3). For an aquatic source, the data suggest an expression of the form

$$(C_a^i/Q^i)_A = \exp\left[5\cdot\ln(\ln A_l) - 9\right] \qquad (7.3)$$

where A_l is the lake area, which ranges between 10^4 and 10^6 m^2 (Section 5.5.3). Equation (7.3) provides a good fit for lake areas near 10^5 m^2 (Figure 7-3), and overestimates Wilson's predictions at smaller and larger values. Wilson presents his results in both tabular and analytical forms. The tabular results are an exact numerical solution of the governing equations, and were used to generate most of the theoretical values plotted in Figure 7-3. The exceptions are the points beyond $A_l = 10^5$ m^2 on the aquatic dispersion curve; Wilson's table does not extend to these values for small values of z_o. These values were calculated from the analytical solution, which is an approximate solution to the governing equations (Wilson 1982b).

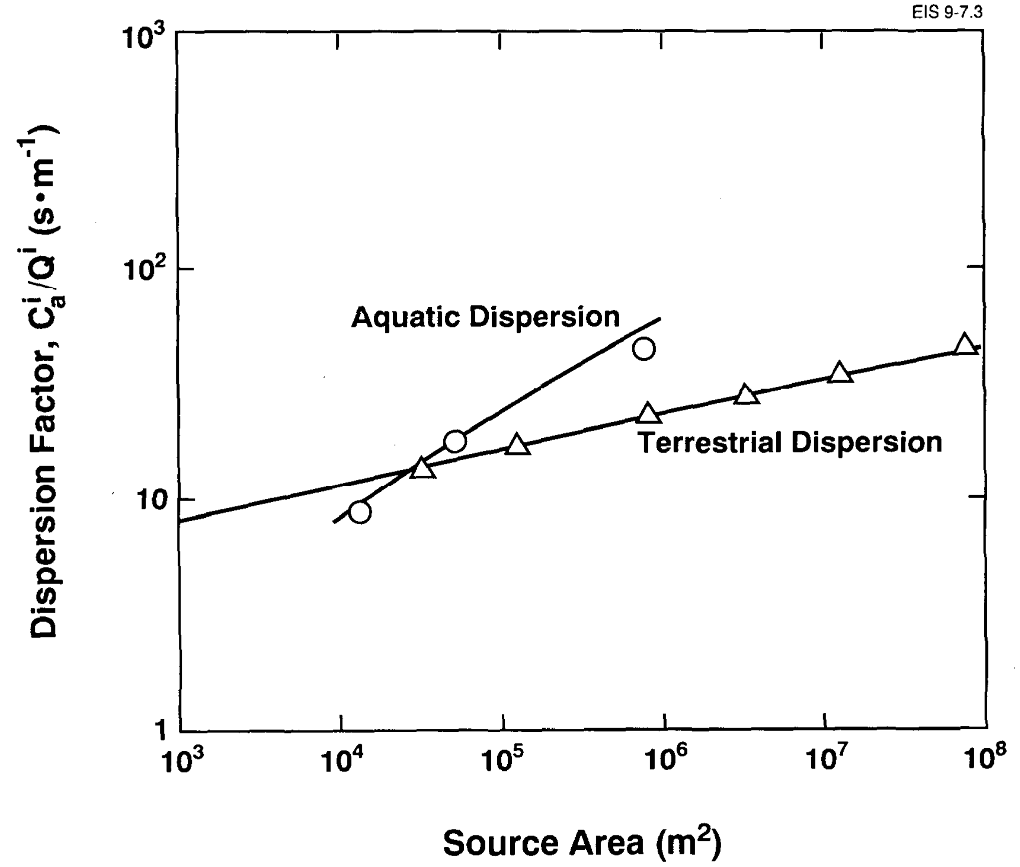


FIGURE 7-3: Dispersion Factors for Terrestrial and Aquatic Sources. The points are calculated from the model of Wilson (1982b). The lines correspond to Equation (7.2) for terrestrial and to Equation (7.3) for aquatic sources.

Equations (7.2) and (7.3) provide dispersion factors for wind speed and atmospheric stability conditions representative of WL. Atmospheric stability does not vary substantially among locations on the Ontario portion of the Shield. Also, because of its long cold winters, WL has a relatively high proportion of stable atmospheric conditions, which lead to relatively large C_a^i/Q^i values. Accordingly, from the point of view of atmospheric stability, the application of Equations (7.2) and (7.3) to a generic location will lead to representative or slightly conservative results. On the other hand, wind speed varies substantially from site to site across the Canadian Shield, with WL values being close to average. We introduce spatial variability of wind speed into Equations (7.2) and (7.3) through a weighting factor, UWGHT (unitless), which is the ratio of wind speed at a generic site to the average wind speed at a number of sites across the Shield. Then the final form of the annual average atmospheric dispersion factor for a ground-level terrestrial area source, $(DISP)_T$ (s·m^{-1}), becomes

$$(DISP)_T = \left[4.87 \cdot A_T^{1/8} - 3.56\right]/UWGHT \qquad (7.4)$$

and for an aquatic source, $(DISP)_A$ ($s \cdot m^{-1}$),

$$(DISP)_A = \exp\left[5 \ln(\ln A_1) - 9\right]/UWGHT \qquad (7.5)$$

In these relations, UWGHT appears in the denominator because higher wind speeds lead to greater dilution and lower nuclide concentrations in air.

7.3.1.2 Elevated Point Sources

In the biomass-burning-for-energy pathway, emissions to the atmosphere occur through a chimney. Since most chimneys of houses are relatively short, we assume conservatively that the emissions become entrained in the cavity that forms in the lee of the building as a result of the disruption of the air flow by the building. The normalized air concentration in the cavity is usually calculated from the empirical expression (Hanna et al. 1982)

$$C_a^i/Q^i = KK/(BW \cdot BH \cdot UCAV) \qquad (7.6)$$

where C_a^i/Q^i ($s \cdot m^{-3}$) is the air concentration normalized by the source strength, or dispersion factor. In Equation (7.6),

KK is a building wake entrainment parameter (unitless),

BW is the building width (m),

BH is the building height (m), and

UCAV is the ambient wind speed ($m \cdot s^{-1}$).

The values assigned to the parameters in Equation (7.6) and in subsequent equations for the atmosphere model are discussed in Section 7.5.

Releases from a chimney can contaminate the air on only one side of the building at a time. If we assume that the wind blows with equal probability from all directions, the likelihood is one in four that the plume will be present behind one particular side, given a square or rectangular building. An annual average atmospheric dispersion factor for chimney releases, $(DISP)_B$ ($s \cdot m^{-3}$), is therefore obtained by dividing Equation (7.6) by four:

$$(DISP)_B = 0.25 \cdot KK/(BW \cdot BH \cdot UCAV) \quad . \qquad (7.7)$$

7.3.1.3 Indoor Releases

Nuclides may be released inside houses from contaminated domestic water. We assume that the flux to the building interior is dispersed uniformly throughout the building so that concentrations are equal everywhere. In this case, the atmospheric dispersion factor for indoor releases, $(DISP)_I$ ($s \cdot m^{-3}$), is

$$(DISP)_I = 1/(BVOL \cdot INFILT) \qquad (7.8)$$

where BVOL is the building volume (m^3) and INFILT is the infiltration rate (s^{-1}). We assume that the building is square, so that its volume is given by

$$BVOL = BW^2 \cdot BH \tag{7.9}$$

where BW and BH are the building width and height defined in Section 7.3.1.2.

7.3.1.4 Lateral Transfers Between Fields

In general, each of the four fields used by the critical group and other biota has a different soil concentration. The nuclide flux to the atmosphere from each field will therefore be different, and air concentrations will vary from field to field. In the real atmosphere, the concentration at a given location will largely reflect the concentration of the underlying field, but it will also be influenced by upwind sources. The effects of lateral transport between fields is difficult to predict for a generic site, such as ours, where the locations of the various fields with respect to each other and to the wind direction are unknown. We have accounted for these effects implicitly by calculating air concentrations in a conservative way. Where a suspension mechanism is active over all fields (e.g., particle suspension), we calculate air concentrations over the most highly contaminated field in the usual way, and assume that the air over all the fields is contaminated to this extent. Where a suspension mechanism can occur over only one field (e.g., agricultural fires), the air concentration calculated for that field is assumed to apply over all the fields. The total air concentration, which is the sum of contributions from all suspension mechanisms, is therefore the same at any point over any of the fields and exceeds the concentration that could actually be achieved anywhere. This approach ensures that the air concentration is not underestimated.

7.3.2 Air Concentrations for Special Radionuclides

Air concentrations for the vast majority of nuclides are determined by applying the concepts discussed above. However, tritium and the noble gases argon and krypton are handled in different ways. The treatment of these radionuclides is discussed in this section.

7.3.2.1 Tritium

As noted in Section 2.5.1, internal tritium doses are calculated using a specific-activity model based on tritium concentrations in lake or well water. External tritium doses are very small and can be ignored (Zach and Sheppard 1992). Tritium concentrations in air are therefore not required and are not calculated.

7.3.2.2 Argon and Krypton

The noble gas nuclides ^{39}Ar, ^{81}Kr and ^{85}Kr are inert and highly mobile in the biosphere. We assume that the soil and lake do not form a barrier to their movement so that transfer to the atmosphere can be quantified by the flux out of the geosphere, χ^i ($mol \cdot a^{-1}$). We further assume that this flux enters the lake, and derive another flux term, $(Q^i)_{AG}$ (mol·m^{-2} water·a^{-1}), by distributing χ^i evenly over the lake area, A_l (m^2):

$$(Q^i)_{AG} = \chi^i / A_l \quad . \tag{7.10}$$

Air concentrations, $(C_a^i)_{AG}$ (mol·m^{-3} air), are found by combining Equation (7.10) with the dispersion relation for aquatic areas, Equation (7.5),

$$(C_a^i)_{AG} = \frac{\chi^i}{A_l} \cdot (DISP)_A / (3.16 \times 10^7) \tag{7.11}$$

where the factor 3.16 x 10^7 s·a^{-1} converts the flux from mol·a^{-1} to mol·s^{-1}.

7.3.3 Air Concentrations from Particulate Suspension

All the nuclides, with the exception of tritium, argon and krypton, are assumed to be suspended as particles of terrestrial and aqueous matter that become entrained in the atmosphere.

7.3.3.1 Terrestrial Sources

The main mechanisms of particulate suspension from terrestrial environments are wind erosion of soil, vehicle traffic on dusty roads, and industrial and agricultural activities (Amiro 1985). We believe these processes are not well enough understood to model each of them individually. Even the science of wind erosion, which has been studied extensively (Male 1985), is unable to predict adequately the vertical flux of soil particles suspended by wind action. Moreover, numerous other suspension mechanisms, which individually contribute little to the nuclide load in the atmosphere, could in total have a significant effect.

Because particulate fluxes cannot be estimated reliably, we cannot use the flux/dispersion methodology presented in Section 7.3.1.1 to calculate air concentrations. Instead, we have used a mass loading approach that treats all the particulate suspension processes simultaneously. We calculate air concentrations from terrestrial particulate suspension, $(C_a^i)_{TP}$ (mol·m^{-3}), from

$$(C_a^i)_{TP} = ADL \cdot C_s^i \tag{7.12}$$

where ADL is the atmospheric dust load (kg dry soil·m^{-3} air) and C_s^i is the concentration of nuclide i in the surface soil layer (mol·kg^{-1} dry soil). Equation (7.12) states that all the suspended particulate material in the atmosphere is contaminated to the same extent as the soil, irrespective of the size of the suspended particles. This approach is conservative because much of the atmospheric dust load originates from distant sources and would be uncontaminated. The dust loading approach accounts for all possible terrestrial particulate suspension mechanisms and implicitly incorporates the effects of dispersion. In applying Equation (7.12), C_s^i was set equal to the greatest nuclide concentration among the garden, forage field, woodlot and peat bog soils (Section 7.3.1.4).

The ambient dust load includes suspended organic matter that reaches the atmosphere from vegetation as a result of fires or pollen release. Equation (7.12) is not strictly applicable to organic material because it is expressed in terms of a soil concentration, and so assumes a soil source. Suspension mechanisms involving biomass sources are treated explicitly as discussed in Section 7.3.5. Air concentrations will therefore be overestimated to the extent that ADL includes organic material.

7.3.3.2 Aquatic Sources

Water is suspended into the atmosphere through wind and bubble-bursting action at the lake/atmosphere interface (Junge 1963, Blanchard 1983). As the water droplets evaporate, dry aerosols are left suspended. We assume, conservatively, that all of the particles suspended initially are small enough to remain suspended in the air.

For aquatic particulate suspension we adopt a mass-loading concept similar to that used for terrestrial particulate suspension (Section 7.3.3.1), so that

$$(C_a^i)_{AP} = AADL \cdot C_l^i \tag{7.13}$$

where $(C_a^i)_{AP}$ is the concentration of nuclide i in the atmosphere (mol·m^{-3} air) caused by aquatic particulate suspension,

C_l^i is the concentration of radionuclide i in the lake (mol·m^{-3} water), and

AADL is the aquatic atmospheric dust load (m^3 water·m^{-3} air).

This mass-loading approach accounts for all the possible aquatic particulate suspension mechanisms, as well as for the effects of dispersion.

7.3.4 Air Concentrations from Gaseous Suspension

Of the nuclides in the vault, ^{39}Ar, ^{81}Kr, ^{85}Kr and ^{222}Rn are gases, and ^{14}C, ^{3}H, ^{129}I and ^{79}Se can form gaseous species. We model these nuclides as gases and assume that gaseous transport is not a major process for the other nuclides in the vault inventory (Table 1-1). Tritium, argon and krypton were discussed in Section 7.3.2; radon, iodine, carbon and selenium are considered here. Gaseous emissions include contributions from processes such as inorganic and organic chemical reactions, and microbial and other biotic activities. Specific examples are methylation of carbon, iodine and selenium, and respiration of carbon.

7.3.4.1 Radon

Terrestrial Sources

Radon behaviour in the biosphere has been studied extensively in the context of uranium mine tailings (Rogers et al. 1980) and diffusion into buildings (Jonassen and McLaughlin 1980). As a result, fairly sophisticated models of radon transport are available. Since radon is in secular equilibrium with its precursor, ^{226}Ra, radon fluxes to the atmosphere from terrestrial sources have traditionally been expressed in terms of ^{226}Ra concentrations in soil. We calculate the air concentration of radon from terrestrial sources, $(C_a^{Rn})_{TG}$ (mol·m^{-3} air), using

$$(C_a^{Rn})_{TG} = (Q^{Rn})_{TG} \cdot (DISP)_T \qquad (7.14)$$

where $(Q^{Rn})_{TG}$ is the flux of radon from terrestrial gaseous emissions (mol·m^{-2} soil·s^{-1}) and $(DISP)_T$ is defined in Equation (7.4).

The radon flux can be expressed as

$$(Q^{Rn})_{TG} = q^{Rn} \cdot C_s^{Ra} \qquad (7.15)$$

where q^{Rn} is the radon emission rate ((mol ^{222}Rn·m^{-2} soil·s^{-1})/(mol ^{226}Ra·kg^{-1} dry soil)), and C_s^{Ra} is the ^{226}Ra concentration in the soil (mol·kg^{-1} dry soil). The emission rate q^{Rn} can itself be expressed in terms of the physical properties of the soil and the radiological properties of radon and ^{226}Ra (Rogers et al. 1980, UNSCEAR 1982). In practice, we treated q^{Rn} as a sampled parameter, with a PDF constructed from 1000 values calculated from its defining equation. Values of the controlling parameters in this equation were drawn randomly from their distributions (Amiro 1992b).

In applying Equations (7.14) and (7.15), C_s^{Ra} is set equal to the largest soil concentration predicted among the garden, forage field, woodlot and peat bog. The area, A_T (m^2), used to calculate $(DISP)_T$ is set equal to the sum of the areas of these fields. This is more conservative than using the area of a single field only.

Aquatic Sources

The air concentration of radon from aquatic sources, $(C_a^{Rn})_{AG}$ (mol·m^{-3} air), is also calculated using a flux/dispersion relationship

$$(C_a^{Rn})_{AG} = (Q^{Rn})_{AG} \cdot (DISP)_A \qquad (7.16)$$

where $(Q^{Rn})_{AG}$ is the radon flux from the water to the atmosphere (mol·m^{-2} water·s^{-1}), and $(DISP)_A$ is defined in Equation (7.5). The amount of information on radon fluxes from lakes is much less than from soils. Accordingly, $(Q^{Rn})_{AG}$ is modelled using a simple aquatic transfer coefficient, ATC^{Rn} (m·s^{-1}),

$$(Q^{Rn})_{AG} = ATC^{Rn} \cdot C_l^{Rn} \qquad (7.17)$$

where C_l^{Rn} is the concentration of radon in the lake (mol·m^{-3} water).

Values of $(Q^{Rn})_{AG}$ are quite conservative over an entire year because in winter frozen lakes do not release appreciable amounts of radon.

7.3.4.2 Iodine-129

Terrestrial Sources

The experimental data on gaseous iodine emissions from soil are fairly limited, and are generally expressed as the fraction of the soil inventory lost per unit time. This suggests a model of the iodine flux, $(Q^I)_{TG}$ (mol·m^{-2} soil·s^{-1}), in terms of an evasion rate constant, η_s^I (s^{-1}),

$$(Q^I)_{TG} = Z_s \cdot \rho_s \cdot \eta_s^I \cdot C_s^I \quad . \tag{7.18}$$

Here, Z_s is the soil depth (m) from which the ^{129}I is released; we set $Z_s = 0.3$ m, the depth of the root zone, which is assumed to be well mixed (Section 6.5.1.2). ρ_s is the soil bulk density (kg dry soil·m^{-3} soil) (Section 6.5.1.3) and C_s^I is the ^{129}I concentration in soil (mol·kg^{-1} dry soil). In practice, C_s^I was set equal to the largest predicted ^{129}I concentration among the garden, forage field, woodlot and peat bog soils. The concentration of ^{129}I in air, $(C_a^I)_{TG}$ (mol·m^{-3} air), is given by

$$(C_a^I)_{TG} = (Q^I)_{TG} \cdot (DISP)_T \quad . \tag{7.19}$$

The area, A_T, used to calculate $(DISP)_T$ (Equation 7.4) was set equal to the sum of the areas of the garden, forage field, woodlot and peat bog.

Aquatic Sources

All of the studies of gaseous iodine emissions from water have been made over oceans rather than over lakes. The available data are best expressed in terms of a mass loading parameter, AIML (m^3 water·m^{-3} air), so that the ^{129}I air concentration, $(C_a^I)_{AG}$ (mol·m^{-3} air), is given by

$$(C_a^I)_{AG} = AIML \cdot C_l^I \tag{7.20}$$

where C_l^I is the ^{129}I concentration in lake water (mol·m^{-3} water). This approach accounts for all possible aquatic gaseous release mechanisms and for the effects of dispersion.

7.3.4.3 Carbon-14

Terrestrial Sources

We use the same model for the terrestrial release of carbon as we do for terrestrial emissions of iodine. Therefore, the air concentration of gaseous forms of ^{14}C arising from terrestrial sources, $(C_a^c)_{TG}$ (mol·m^{-3} air), is given by

$$(C_a^c)_{TG} = Z_s \cdot \rho_s \cdot \eta_s^c \cdot C_s^c \cdot (DISP)_T \tag{7.21}$$

where η_s^c is the evasion rate constant of carbon from soil (s^{-1}), C_s^c is the ^{14}C concentration in soil (mol·kg^{-1} dry soil), and the other parameters are defined in Equations (7.4) and (7.18).

Aquatic Sources

As noted in Section 5.3.1, the gaseous emission of carbon to the atmosphere from lake water can be modelled using an evasion rate, η_l^c (s^{-1}). Accordingly, the air concentration of gaseous forms of ^{14}C arising from aquatic sources, $(C_a^c)_{AG}$ ($mol \cdot m^{-3}$ air), is given by

$$(C_a^c)_{AG} = Z_l \cdot \eta_l^c \cdot C_l^c \cdot (DISP)_A \tag{7.22}$$

where Z_l is the mean lake depth (m), η_l^c the evasion rate constant of carbon from surface water (s^{-1}), C_l^c the ^{14}C concentration in lake ($mol \cdot m^{-3}$ water) and $(DISP)_A$ is defined in Equation (7.5).

7.3.4.4 Selenium-79

The available data support the use of an evasion rate to model the gaseous flux of selenium from the soil to the atmosphere. Therefore, ^{79}Se air concentrations resulting from gaseous emissions from terrestrial sources, $(C_a^{se})_{TG}$ ($mol \cdot m^{-3}$ air), are given by an expression similar to Equations (7.19) and (7.21) for ^{129}I and ^{14}C:

$$(C_a^{se})_{TG} = Z_s \cdot \rho_s \cdot \eta_s^{se} \cdot C_s^{se} \cdot (DISP)_T \tag{7.23}$$

where η_s^{se} is the selenium evasion rate constant from soil (s^{-1}) and C_s^{se} the ^{79}Se concentration in soil ($mol \cdot kg^{-1}$ dry soil). The other parameters are defined in Equations (7.4) and (7.18).

The experimental evidence for selenium volatilization from water is inconclusive. We assume that aquatic sources do not contribute to the air concentration of gaseous ^{79}Se species, although we allow ^{79}Se to be suspended in aqueous particles (Section 7.3.3.2).

7.3.5 Air Concentrations from Fires

All the nuclides, with the exception of ^{39}Ar, ^{3}H, ^{81}Kr and ^{85}Kr, are assumed to be suspended when contaminated vegetation or peat is burned. The suspended nuclides may be gaseous or attached to smoke particles.

7.3.5.1 Agricultural Fires

We assume that the stubble is burned on the forage field each year, thereby releasing some of the nuclides in the crop to the atmosphere. The annual flux from agricultural fires, $(Q^i)_{AF}$ ($mol \cdot m^{-2}$ $land \cdot s^{-1}$), is calculated from the nuclide inventory in the crop

$$(Q^i)_{AF} = C_D^i \cdot ff \cdot Y_j \cdot (EMFRAC^i)_{AF} \tag{7.24}$$

where C_D^i is the concentration of nuclide i in the crop biomass ($mol \cdot kg^{-1}$ wet biomass),

ff is the fire frequency (s^{-1}),

Y_j is the biomass yield of food type j (kg wet biomass·m^{-2} soil), and

$(EMFRAC^i)_{AF}$ is the fraction of the nuclide released in fire (unitless).

For an annual fire, ff = 1 a^{-1}, or 3.17×10^{-8} s^{-1}. In Equation (7.24), BIOTRAC uses the highest biomass yield, Y_j, selected during each simulation (Section 8.5.8.1).

The calculation of C_b^i is described in detail as part of the food-chain and dose submodel in Section 8.3. In general, nuclides can reach agricultural crops in three ways:

1. They may be taken up from the soil by the plant roots.
2. They may be deposited onto plant leaves during aerial irrigation.
3. They may be deposited onto plant leaves from the atmosphere.

We do not account for the contribution from atmospheric deposition in calculating C_b^i for use in Equation (7.24). To do so would count these nuclides twice since the original air concentrations were not depleted during the deposition processes. We have not included the irrigation pathway. In the forage field, the contribution to C_b^i from the interception of irrigation water from the lake is generally much less than the contribution from root uptake. Furthermore, the forage field is very rarely irrigated. Accordingly, C_b^i may be calculated assuming that nuclides reach the plant through root uptake only. We show in Section 8.3.1.1 that plant concentration for nuclide i resulting from root uptake can be predicted from

$$C_b^i = Bv^i \cdot C_s^i \tag{7.25}$$

where Bv^i is the plant/soil concentration ratio ((mol·kg^{-1} wet biomass)/(mol·kg^{-1} dry soil)) and C_s^i the concentration in the root-zone soil (mol·kg^{-1} dry soil). Since values of C_s^i are available from the soil submodel (Chapter 6), C_b^i for use in Equation (7.24) can be calculated before the food-chain model itself is run. In applying Equation (7.25), C_s^i was set equal to the predicted concentration in the soil of the forage field.

The air concentration resulting from agricultural fires, $(C_a^i)_{AF}$ (mol·m^{-3} air), is given by

$$(C_a^i)_{AF} = (Q^i)_{AF} \cdot (DISP)_T \tag{7.26}$$

where $(Q^i)_{AF}$ is defined in Equation (7.24) and $(DISP)_T$ is defined in Equation (7.4). The area A_T that appears in $(DISP)_T$ was here set equal to the area of the forage field. Because $(DISP)_T$ was developed for a passive dispersion source, it will overestimate fire-induced air concentrations since fires enhance the dispersive power of the atmosphere.

7.3.5.2 Biomass Combustion for Energy

We assume that the home of the critical group is heated by burning contaminated wood or peat. The fuel is burned in a stove and nuclides are released to the atmosphere through a chimney. When wood is used as the fuel, the air concentration caused by home-heating fires, $(C_a^i)_{EF}$ ($mol \cdot m^{-3}$ air), is calculated from the nuclide inventory in the wood

$$(C_a^i)_{EF} = \frac{C_b^i \cdot FUELUS \cdot (EMFRAC^i)_{EF}}{EW} \cdot (DISP)_B \tag{7.27}$$

where	C_b^i	is the concentration of nuclide i in the wood fuel ($mol \cdot kg^{-1}$ wet biomass),
	FUELUS	is the amount of energy required to heat a single family home ($MJ \cdot s^{-1}$),
	$(EMFRAC^i)_{EF}$	is the fraction of nuclide i released in fire (unitless),
	EW	is the convertible energy content of wood ($MJ \cdot kg^{-1}$ wet biomass), and
	$(DISP)_B$	is the dispersion coefficient for elevated point sources ($s \cdot m^{-3}$) defined in Equation (7.7).

Trees in a woodlot are subject to the same exposure pathways as pasture crops in a forage field, but we assume that trees are never irrigated. As shown in Section 7.3.5.1, C_b^i in Equation (7.27) can be calculated using Equation (7.25), where C_s^i is set equal to the soil nuclide concentration predicted by BIOTRAC for the woodlot.

Residents of the Canadian Shield rarely, if ever, use peat as a source of home heating fuel today. But because nuclide concentrations in peat could exceed concentrations in wood, this exposure pathway cannot be ignored. For peat burning, the air concentration is given by

$$(C_a^i)_{EF} = \frac{C_s^i \cdot FUELUS \cdot (EMFRAC^i)_{EF}}{EP} \cdot (DISP)_B \tag{7.28}$$

where C_s^i is the concentration of nuclide i in the peat bog or soil ($mol \cdot kg^{-1}$ dry peat) and EP is the convertible energy content of peat ($MJ \cdot kg^{-1}$ dry peat).

7.3.5.3 Forest and Land-Clearing Fires

At a given site on the Canadian Shield, forest fires occur naturally about once per century. Forests and peat may also be deliberately burned to make land suitable for agriculture. Although such events occur infrequently, any single generation could be exposed to clearing fires. Accordingly, we assume that the trees and peat in an area equal to the area of the woodlot

(Section 9.1.1.3) burn every 50 a. The resulting nuclide flux to the atmosphere, $(Q^i)_{LF}$ (mol·m^{-2} land·s^{-1}), is estimated from the nuclide inventory in the material burned:

$$(Q^i)_{LF} = [C_b^i \cdot FY + C_s^i \cdot PY] \cdot (EMFRAC^i)_{LF} \cdot f_1 \qquad (7.29)$$

where C_b^i is the nuclide concentration of nuclide i in the trees (mol·kg^{-1} wet biomass),

FY is the tree mass consumed in the fire (kg wet biomass·m^{-2} land),

C_s^i is the nuclide concentration in peat (mol·kg^{-1} dry peat),

PY is the mass of peat burned (kg dry peat·m^{-2} land),

$(EMFRAC)^i{}_{LF}$ is the fraction of nuclide i released in fire (unitless), and

f_1 is the frequency of forest or land-clearing fires (s^{-1}).

For fires with a 50-a return period, $f_1 = 0.02$ a^{-1} or 6.34×10^{-10} s^{-1}.

Equation (7.25) defines C_b^i, where C_s^i is equal to the soil concentration of the woodlot. The term $C_s^i \cdot PY$ in Equation (7.29) is included only if the soil type is organic (Section 6.5.1.1), in which case the soil concentration is that of the woodlot here as well.

The air concentration of nuclide i from forest and land-clearing fires, $(C_a^i)_{LF}$ (mol·m^{-3} air), is found by combining the flux with the dispersion factor for a terrestrial area source

$$(C_a^i)_{LF} = (Q^i)_{LF} \cdot (DISP)_T \quad . \qquad (7.30)$$

The area A_T used to calculate $(DISP)_T$ in Equation (7.4) equals here the area of the woodlot.

7.3.6 Air Concentrations from Indoor Sources

In Sections 7.3.3 to 7.3.5, we deal with air concentrations arising from outdoor sources, which includes biomass combustion for energy. Here we consider concentrations involving two indoor sources. We discuss the diffusion of radon from soil into houses and the release of nuclides from contaminated water brought into the house for domestic use.

7.3.6.1 Radon Diffusion into Buildings

In general, most of the radon present in indoor air originates from ^{226}Ra in the soil surrounding houses (Bruno 1983). However, it has proven difficult to relate the observed soil ^{226}Ra concentrations to the measured radon concentrations in the indoor air. In areas where the soil ^{226}Ra concentrations show little spatial variability, indoor radon concentrations can range over several orders of magnitude (McGregor et al. 1980). Consequently, there is no observed correlation between soil ^{226}Ra and indoor radon concentrations (George and Breslin 1980). This situation arises because indoor radon concentrations depend on a large number of factors, including the number and size of cracks in the foundation of the house, diffusion coefficients in soil and building materials, infiltration rates and the lifestyle of the building's occupants.

Because the detailed prediction of radon concentrations in indoor air is difficult, we relate the indoor radon concentration, $(C^{Rn})_{IGS}$ (mol·m^{-3} air), to the soil ^{226}Ra concentration, C_s^{Ra} (mol·kg^{-1} dry soil), through a simple transfer coefficient

$$(C_a^{Rn})_{IGS} = INDRN \cdot C_s^{Ra} \quad . \tag{7.31}$$

INDRN is the indoor radon transfer coefficient ((mol ^{222}Rn·m^{-3} air)/(mol ^{226}Ra·kg^{-1} dry soil)). INDRN is a sampled parameter (Section 7.5.1.6) so that a given ^{226}Ra soil concentration can cause a wide range of radon air concentrations. C_s^{Ra} is based on the garden or forage field, whichever has the higher concentration in a given simulation.

In theory, other volatile nuclides could also diffuse from the soil into houses and build up to potentially high concentrations. However, this pathway is considered for radon only. Air concentrations for argon and krypton radionuclides are calculated so conservatively (Section 7.3.2.2) that any indoor buildup is likely covered. Air concentrations for tritium are not needed (Section 7.3.2.1). No data are available to quantify infiltration of ^{14}C, ^{129}I or ^{79}Se. However, if an expression like Equation (7.31) were applied to ^{14}C or ^{129}I, the resulting air concentrations would be much less than those predicted for release from domestic water (Section 7.3.6.2) for typical soil and water concentrations. So we focus on the release of these nuclides from domestic water only.

7.3.6.2 Release from Domestic Water

Nuclides can be released to indoor air from contaminated water brought into the house and used for activities such as showers, or in appliances such as dishwashers and humidifiers (Lowry et al. 1987, Giardino et al. 1988). We calculate the flux of a nuclide released to the indoor air, $(Q^i)_{IGW}$ (mol·a^{-1}), based on the nuclide inventory in the water required by the critical group for domestic purposes:

$$(Q^i)_{IGW} = C_w^i \cdot Uwc \cdot Nph \cdot RELFRAC^i \tag{7.32}$$

where C_w^i is the concentration of nuclide i in the domestic water (mol·m^{-3} water),

Uwc is annual water demand of each member of the household (m^3 water·a^{-1}·p^{-1}),

Nph is the number of people per household (p), and

RELFRAC^i is the fraction of the inventory in domestic water released to indoor air (unitless).

C_w^i can refer to the concentration in well water or lake water (Equations (4.18) and (5.6)), whichever source is used in BIOTRAC to satisfy domestic demands (Section 9.1.2). The flux is assumed to be dispersed uniformly throughout the building so that the air concentration, $(C_a^i)_{IGW}$ (mol·m^{-3} air), is given by

$$(C_a^i)_{IGW} = (Q^i)_{IGW} \cdot (DISP)_I / (3.16 \times 10^7) \qquad (7.33)$$

where $(DISP)_I$ is the indoor dispersion coefficient defined in Equation (7.8), and the factor 3.16×10^7 s·a^{-1} converts the flux from mol·a^{-1} to mol·s^{-1}.

Although all the nuclides could be released through the domestic use of water, we believe that significant air concentrations could result for gaseous species only. Since ^{39}Ar, ^{3}H, ^{81}Kr, and ^{85}Kr are treated by other methods and since the evidence for ^{79}Se volatilization from water is unclear, we apply this pathway to ^{14}C, ^{129}I and ^{222}Rn only.

7.3.7 Total Air Concentrations

In the previous sections, we calculated air concentrations arising from specific suspension mechanisms. The total air concentration for a given nuclide is the sum of these concentrations, considering only those mechanisms in which the nuclide can be involved. The total concentrations for ^{14}C, ^{129}I, ^{79}Se and ^{222}Rn include the contributions of both particulate and gaseous forms. Although this may result in double accounting, it compensates for our uncertainty regarding the chemical form of these nuclides. The total indoor and outdoor concentrations involve different contributions. All outdoor sources are assumed to contribute to the outdoor concentration; they all contribute to the indoor concentration as well, except for the biomass combustion-for-energy pathway (Section 7.3.5.2). Nuclides released from a chimney are unlikely to be transported back into the house. Indoor sources are assumed to contribute to indoor air concentrations only.

The mechanisms that contribute to the air concentration for the various nuclides are indicated in Tables 7-1 and 7-2 for outdoor and indoor air respectively. These tables can be used to establish the total concentration for each nuclide as calculated by BIOTRAC. For example, the total indoor concentration for ^{129}I is given by

$$C_a^I = (C_a^I)_{TP} + (C_a^I)_{AP} + (C_a^I)_{TG} + (C_a^I)_{AG} + (C_a^I)_{AF} + (C_a^I)_{LF} + (C_a^I)_{IGW} \quad . \qquad (7.34)$$

Argon-39, ^{81}Kr and ^{85}Kr are not listed in the tables but the total concentrations for both outdoor and indoor air are given by Equation (7.11).

7.3.8 Deposition

Apart from the noble gases, all nuclides in the atmosphere are subject to deposition to underlying surfaces. Some deposition processes such as gravitational settling, particle impaction, gaseous sorption by plants, and molecular diffusion occur whether or not precipitation is falling. These

TABLE 7-1

PATHWAYS CONTRIBUTING TO OUTDOOR AIR CONCENTRATIONS

Pathway	Nuclide				All Other Nuclides
	^{14}C	^{79}Se	^{129}I	^{222}Rn	
Terrestrial Particles (TP)	X	X	X	X	X
Aquatic Particles (AP)	X	X	X	X	X
Terrestrial Gases (TG)	X	X	X	X	
Aquatic Gases (AG)	X		X	X	
Agricultural Fires (AF)	X	X	X	X	X
Energy Fires (EF)	X	X	X	X	X
Land-Clearing Fires (LF)	X	X	X	X	X

TABLE 7-2

PATHWAYS CONTRIBUTING TO INDOOR AIR CONCENTRATIONS

Pathway	Nuclide				All Other Nuclides
	^{14}C	^{79}Se	^{129}I	^{222}Rn	
Terrestrial Particles (TP)	X	X	X	X	X
Aquatic Particles (AP)	X	X	X	X	X
Terrestrial Gases (TG)	X	X	X	X	
Aquatic Gases (AG)	X		X	X	
Agricultural Fires (AF)	X	X	X	X	X
Land-Clearing Fires (LF)	X	X	X	X	X
Diffusion into Buildings (IGS)				X	
Release from Domestic Water (IGW)	X		X	X	

processes together are responsible for what is termed dry deposition. Wet deposition occurs in the presence of precipitation, and refers to the process of washout in which precipitation falling through the contaminated air scavenges nuclides and carries them to the surface. Wet and dry deposition are about equally effective over the long term for North American climates in removing contaminants from the atmosphere (Slinn 1977). In our atmosphere model, we treat wet and dry deposition as separate processes that are summed to give the total rate of deposition of nuclide i, D^i ($mol \cdot m^{-2} \cdot d^{-1}$), to the underlying surfaces.

Deposition occurs to all types of surfaces, but in our model we allow deposition to the soil and vegetation only and not to the lake. As noted in Section 5.2, the lake receives the entire nuclide flow out of the geosphere. The nuclide load in the lake therefore reflects inputs from all sources, and there is no need to consider deposition separately. To do so would be to count this contribution twice (Section 9.3).

We follow the approach traditionally taken in assessment models (CSA 1987) and simulate dry deposition, DD^i ($mol \cdot m^{-2} \cdot d^{-1}$) using a deposition velocity, Vd ($m \cdot d^{-1}$) (Sehmel 1980),

$$DD^i = Vd \cdot C_a^i \quad . \tag{7.35}$$

The deposition velocity is an empirical parameter that depends on many physical and chemical properties of the atmosphere, the contaminant and the surface (Section 7.5.4.1). Wet deposition, DW^i ($mol \cdot m^{-2} \cdot d^{-1}$), is modelled in an equally simple way using a washout ratio, Wr (unitless) (Slinn 1978),

$$DW^i = P \cdot Wr \cdot C_a^i \quad . \tag{7.36}$$

Here, P is the daily average precipitation rate (m water$\cdot d^{-1}$), which is discussed further in Section 9.1.3. The washout ratio is the ratio of the nuclide concentration in precipitation reaching the ground (kg nuclide$\cdot m^{-3}$ water) to the air concentration of the nuclide (kg nuclide$\cdot m^{-3}$ air). The product $P \cdot Wr$ is effectively a wet deposition velocity. The same values of Vd and Wr are used for deposition to both soil and vegetation. We assume conservatively that deposition does not deplete the nuclide concentration in air.

Not all the nuclides are subject to deposition. The argon, krypton and radon noble gas radionuclides, being inert, do not accumulate on surfaces, and are not allowed to deposit in our model. There is no need to deposit tritium, which is handled through a specific-activity model (Section 2.5.1). We deposit ^{14}C to soil, but not to vegetation. The uptake of ^{14}C through plant leaves is accounted for implicitly in the parameter describing the uptake of this nuclide from soil (Section 8.5.1.1).

Similarly, not all the suspension mechanisms contribute to the air concentration C_a^i used in Equations (7.35) and (7.36). As explained in Section 6.2.2, only nuclides originating from lake water are allowed to deposit to the soil. Apart from ^{129}I and the nuclides discussed in the previous paragraph, all the airborne nuclides arising from outdoor sources are allowed to deposit to vegetation. However, the parameter describing plant uptake of nuclides from soil implicitly accounts for the deposition of gaseous ^{129}I originating from the soil, but not from other sources (Section 8.5.1.1). Accordingly, the air concentration, C_a^i, used in Equations (7.35) and (7.36) for ^{129}I reflects the contributions from these sources.

The equations used to calculate the flux of the various nuclides from the air to soil and vegetation are summarized below:

1. Deposition to soil, D_s^i ($mol \cdot m^{-2}$ $soil \cdot d^{-1}$):

   For ^{39}Ar, ^{3}H, ^{81}Kr, ^{85}Kr and ^{222}Rn,

   $$D_s^i = 0 \quad . \tag{7.37}$$

   For ^{14}C and ^{129}I,

   $$D_s^i = [(C_a^i)_{AP} + (C_a^i)_{AG}] \cdot [Vd + P \cdot Wr] \quad . \tag{7.38}$$

   For all the other nuclides,

   $$D_s^i = (C_a^i)_{AP} \cdot [Vd + P \cdot Wr] \quad . \tag{7.39}$$

2. Deposition to vegetation, D_b^i ($mol \cdot m^{-2}$ $soil \cdot d^{-1}$):

   For ^{3}H and ^{222}Rn,

   $$D_b^i = 0 \quad . \tag{7.40}$$

   For ^{129}I,

   $$\begin{aligned} D_b^i = {} & [(C_a^i)_{AP} + (C_a^i)_{AG} + (C_a^i)_{TP} + (C_a^i)_{AF} \\ & + (C_a^i)_{EF} + (C_a^i)_{LF}] \cdot [Vd + P \cdot Wr] \quad . \end{aligned} \tag{7.41}$$

   For ^{79}Se,

   $$\begin{aligned} D_b^i = {} & [(C_a^i)_{AP} + (C_a^i)_{TP} + (C_a^i)_{TG} + (C_a^i)_{AF} \\ & + (C_a^i)_{EF} + (C_a^i)_{LF}] \cdot [Vd + P \cdot Wr] \quad . \end{aligned} \tag{7.42}$$

   For all the other nuclides,

   $$\begin{aligned} D_b^i = {} & [(C_a^i)_{TP} + (C_a^i)_{AP} + (C_a^i)_{AF} + (C_a^i)_{EF} \\ & + (C_a^i)_{LF}] \cdot [Vd + P \cdot Wr] \quad . \end{aligned} \tag{7.43}$$

7.4 INTERFACES

The atmosphere submodel requires nuclide flows from the geosphere and nuclide concentrations in other compartments of the biosphere as inputs (Figure 7-4). Geosphere flows are required only for ^{39}Ar, ^{81}Kr and ^{85}Kr, which are assumed to pass directly through the lake to the atmosphere (Section 2.5.5). Lake-water concentrations are needed for all the nuclides except ^{39}Ar, ^{3}H, ^{81}Kr and ^{85}Kr to provide source terms for the aquatic suspension mechanisms. When a well is chosen as the domestic water source in BIOTRAC (Section 9.1.2), well-water concentrations of ^{14}C, ^{129}I and

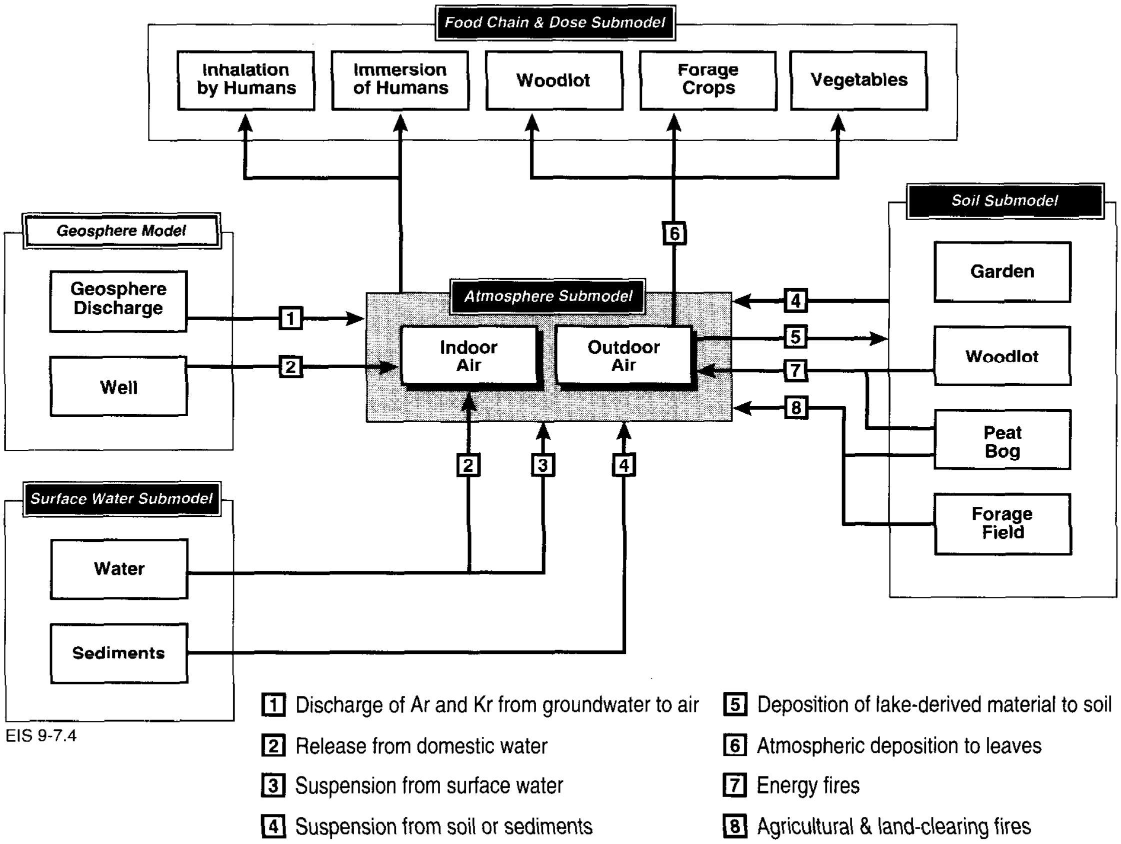


FIGURE 7-4: Interfaces (Related to Human Dose Prediction) Between the Atmosphere Submodel (Shaded), the Geosphere Model, and the Other BIOTRAC Submodels

^{222}Rn are needed to model gaseous releases to indoor air. Soil concentrations are required for all nuclides except ^{39}Ar, ^{3}H, ^{81}Kr and ^{85}Kr to provide source terms for evaluating the terrestrial suspension processes. For simulations in which the critical group is assumed to farm lake sediments, sediment concentrations are needed to calculate effective soil concentrations (Section 6.3.7.3). Nuclide concentrations in crops and trees are also needed to calculate suspension rates following fires. The required biomass concentrations can be obtained from the soil concentrations since the biomass and soil concentrations are directly proportional in this context (Equation 7.25). No inputs from the food-chain and dose submodel are needed. All of the inputs required to drive the atmosphere submodel are therefore available as outputs of the geosphere model, and the surface water and soil submodels.

The primary outputs of the atmosphere submodel are the annual average nuclide concentrations in both indoor and outdoor air for all nuclides except tritium. These concentrations are passed to the food-chain and dose submodel (Chapter 8), where they are used directly to calculate inhalation and immersion doses, taking into account the fraction of the time that members of the critical group spend indoors and outdoors (Sections 8.3.1.10 and 8.3.2.1). The air concentrations are also used to calculate the rate at which airborne nuclides are deposited to the soil and vegetation. The flux to the soil is passed to the soil submodel, where it is used to calculate nuclide concentrations in soil (Section 6.3.1.2). Similarly, the flux to vegetation is passed to the food-chain and dose submodel, and is used to calculate concentrations in crops and trees (Sections 8.3.1.3, 8.3.1.4 and 8.3.2.4). Airborne nuclides therefore contribute indirectly to doses that result from exposure to contaminated soils or vegetation.

Air concentrations do not play a prominent role in evaluating environmental effects (Chapter 13). However, nuclides suspended in air may be deposited on vegetation and soil, which play a prominent role in evaluating environmental effects. Furthermore, non-human biota are subject to air immersion, and we calculate the resulting doses (Section 13.3.3.2).

7.5 ATMOSPHERE PARAMETERS

The atmosphere model contains a large number of parameters. The quantity and quality of the data available for deriving values and distributions for these parameters are varied. Where possible, we have used observational data from the Ontario portion of the Canadian Shield to define parameter values. However, in a number of cases it was necessary to use other sources to supplement these data. Values for most of the parameters are distributed to reflect spatial and temporal variability, and uncertainty in the data and in the model formulation (Section 1.5.7). Fixed values are adopted for parameters that are well defined and show little variability.

For convenience, we have grouped the parameters according to the processes with which they are associated. For each parameter we show how the available data have been used to establish a value or to construct a PDF suitable for the postclosure assessment.

7.5.1 Transfer Parameters

7.5.1.1 Atmospheric Dust Load, ADL
(kg dry soil$\cdot$m^{-3} air)

The atmospheric dust load is the mass per unit volume of suspended particulate matter in the atmosphere. It is used in Equation (7.12) to estimate the air concentration of nuclides suspended from the soil surface in particulate form. Atmospheric dust loads have been measured at numerous sites across Canada since 1970 through the National Air Pollution Surveillance network organized by Environment Canada (NAPS 1970 to 1983). The sites are generally located near large population centres where anthropogenic emissions make up a sizeable portion of the dust load. The data reflect the total atmospheric dust load, and include both mineral and organic particulates from local and remote sources.

We calculated annual mean dust loads for the period 1970 to 1983 inclusive for 149 locations across Canada. The distribution of values, shown in Figure 7-5, reflects the variation of ADL in space. But it probably includes most long-term temporal variability as well, since year-to-year variability at any one site is generally less than site-to-site variability

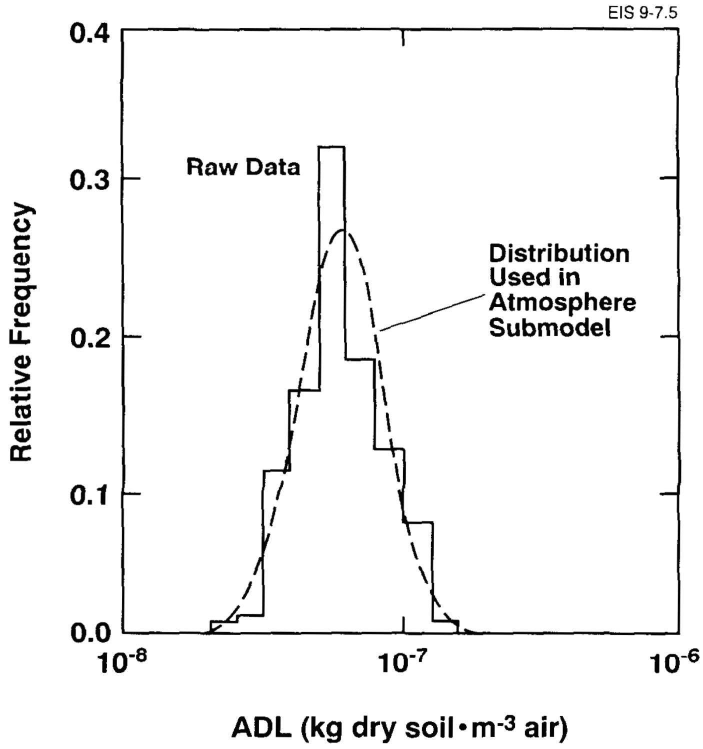


FIGURE 7-5: Distributions of the Atmospheric Dust Load, ADL. The histogram represents data from 149 Canadian sites, and the smooth curve is the distribution used in BIOTRAC.

in the long-term average values. The values are distributed lognormally, with a GM of 5.9 x 10^{-8} kg dry soil·m^{-3} air and a GSD of 1.41. We adopted these values for BIOTRAC. The value of 5.9 x 10^{-8} kg dry soil·m^{-3} air is about a factor of three higher than the average dust load at isolated Canadian Shield sites where anthropogenic emissions are low (Amiro 1992b).

7.5.1.2 Aquatic Atmospheric Dust Load, AADL (m^3 water·m^{-3} air)

The aquatic atmospheric dust load is the ratio of the nuclide concentration in aerosol above a water surface to the nuclide concentration in the water body. It is used in Equation (7.13) to estimate the air concentration of nuclides suspended as particles from the lake. Data for aerosol production above freshwater bodies are not available; instead, we have derived our AADL values from data for the suspension of salt particles over oceans. The marine data likely overestimate values relevant to the Canadian Shield because wind speeds and wave heights are larger over oceans than over lakes. Thus, marine data would lead to conservatively high nuclide concentrations in air.

Using the data on salt concentrations above and within oceans, Amiro (1992b) deduced an average AADL value of 2.9 x 10^{-10} m^3 water·m^{-3} air. In the absence of any information on the variability of this parameter, we assume it is distributed in the same way as ADL (Section 7.5.1.1). Accordingly, for our assessment, AADL is lognormally distributed with GM = 2.9 x 10^{-10} m^3 water·m^{-3} air and GSD = 1.41.

7.5.1.3 Radon Emission Rate from Soil, q^{Rn} ((mol $^{222}Rn·m^{-2}$ soil·s^{-1})/(mol $^{226}Ra·kg^{-1}$ dry soil))

The radon emission rate is used in Equation (7.15) to calculate the radon flux to the atmosphere from a soil with a known ^{226}Ra concentration. A distribution for q^{Rn} was constructed using a model that depends on the physical properties of the soil and on radon (Rogers et al. 1980, UNSCEAR 1982). We defined distributions for each of these properties (Amiro 1992b) and drew 1000 sets of values from them at random. These sets were used in turn to generate 1000 q^{Rn} values, which were found to be distributed approximately lognormally (Figure 7-6). Therefore, we assume q^{Rn} is distributed lognormally, with a GM of 2.7 x 10^{-9} ((mol $^{222}Rn·m^{-2}·s^{-1}$)/(mol $^{222}Ra·kg^{-1}$ dry soil)) and a GSD of 2.16.

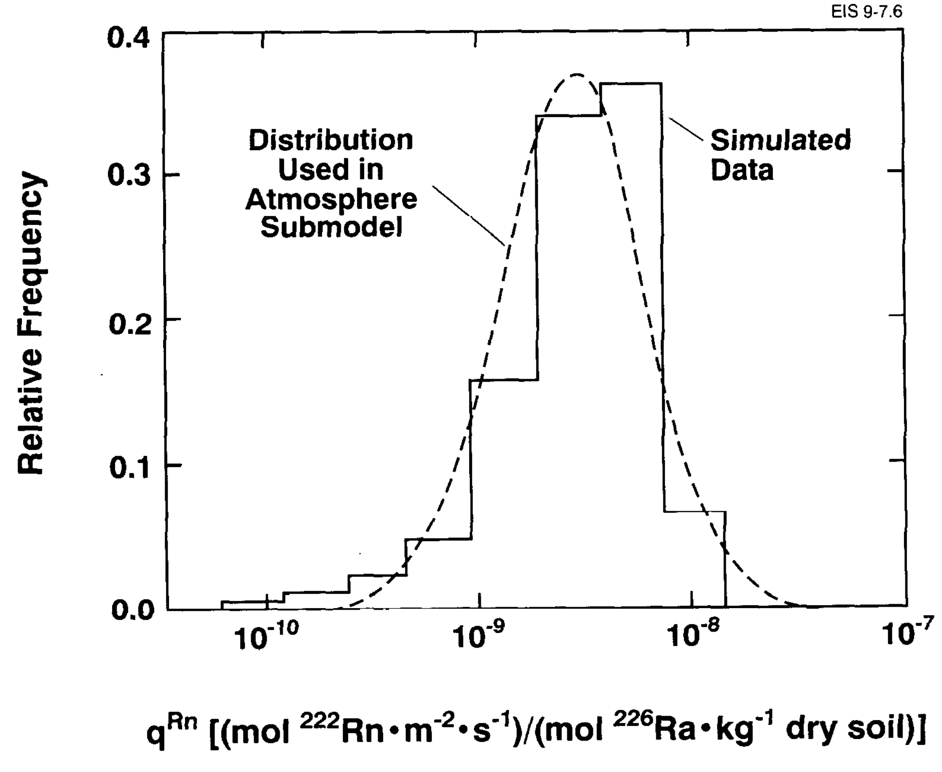


FIGURE 7-6: Distributions of the Radon Emission Rates From Soil, q^{Rn}. The histogram is based on a simulation and the smooth curve is the distribution used in BIOTRAC.

7.5.1.4 Radon Aquatic Transfer Coefficient, ATC^{Rn} ($m \cdot s^{-1}$)

The aquatic transfer coefficient is used in Equation (7.17) to calculate the flux of radon from water bodies to the atmosphere. A few measurements of ATC^{Rn} are available from the Experimental Lakes Area (ELA) in northwestern Ontario (Emerson et al. 1973, Emerson 1975) from lakes representative of the Canadian Shield. Amiro (1992b) synthesized these observations and recommended a normal distribution for ATC^{Rn} with a mean of 4.6×10^{-6} $m \cdot s^{-1}$ and an SD of 2.3×10^{-6} $m \cdot s^{-1}$. This distribution is truncated at its low end at 0 $m \cdot s^{-1}$ because negative values are impossible.

7.5.1.5 Aquatic Iodine Mass Loading Parameter, AIML (m^3 water$\cdot m^{-3}$ air)

AIML is the ratio of the concentration of gaseous iodine in air above a water surface to the iodine concentration in the water itself. It is used in Equation (7.20) to calculate air concentrations of ^{129}I resulting from gaseous emissions from water bodies. As was the case for AADL (Section 7.5.1.2), data from freshwater bodies are unavailable and AIML values must be deduced from studies over the ocean. For the reasons given in Section 7.5.1.2, the marine values should be conservative when applied to Canadian Shield lakes.

Fuge and Johnson (1986) published data on iodine concentrations in ocean water and in the air above it. Amiro (1992b) used these data to extract values of AIML. Since AIML is calculated as a ratio of concentrations, he assumed that it is distributed lognormally, and derived a GM of 1.3×10^{-5} m^3 water$\cdot m^{-3}$ air and a GSD of 6.3. These values are used for the postclosure assessment.

7.5.1.6 Indoor Radon Transfer Coefficient, INDRN ((mol $^{222}Rn \cdot m^{-3}$ air)/(mol $^{226}Ra \cdot kg^{-1}$ dry soil))

The indoor radon transfer coefficient is used in Equation (7.31) to calculate the radon concentration in indoor air resulting from diffusion into the building from soil with a given ^{226}Ra concentration. Soil ^{226}Ra concentrations and indoor radon concentrations are both well known. The former varies little worldwide, and a value of 3×10^{-12} $mol \cdot kg^{-1}$ dry soil is representative of Canadian Shield conditions (DSMA Atcon 1978, Keith Consulting 1978, Sheppard M.I. et al. 1981). Indoor radon concentrations are lognormally distributed and show much greater variability. Studies carried out at several locations on the Canadian Shield produced GM values ranging from 1.6×10^{-17} to 1.3×10^{-16} $mol \cdot m^{-3}$ air, and GSD values exceeding 3 (DSMA Atcon 1978, Keith Consulting 1978, Amiro 1992b). To ensure that BIOTRAC predictions are conservative, we adopt values at the upper end of the observed ranges and assume that indoor radon concentrations on the Canadian Shield are distributed lognormally, with a GM of 1.3×10^{-16} $mol \cdot m^{-3}$ air and a GSD of 4.3. The distribution for INDRN was then derived using Equation (7.31) with $C_s^{Ra} = 3 \times 10^{-12}$ $mol \cdot kg^{-1}$ dry soil. The result is a lognormal distribution for INDRN with a GM of 4.3×10^{-5} (mol $^{222}Rn \cdot m^{-3}$ air)/(mol $^{226}Ra \cdot kg^{-1}$ dry soil) and a GSD of 4.3.

7.5.1.7 Release Fraction, $RELFRAC^i$
(unitless)

$RELFRAC^i$ defines the fraction of the nuclide inventory in domestic water supplies that is released to indoor air (Equation 7.32). As noted in Section 7.3.6.2, we assume that significant air concentrations could result only for ^{14}C, ^{129}I and radon. In the absence of other data, we assume conservatively that the entire inventory of these radionuclides is released so that $RELFRAC^i$ is assigned a fixed value of 1.0 for ^{14}C, ^{129}I and radon. For all other nuclides, $RELFRAC^i = 0$.

7.5.2 Fire Pathway Parameters

7.5.2.1 Domestic Heating Need, FUELUS
($MJ \cdot s^{-1}$)

An estimate of FUELUS is required to calculate the amount of wood or peat burned by the critical group for home heating or other domestic purposes (Equations (7.27) and (7.28)). Long-term averages for total annual fuel consumption for Northern Ontario range between 2.9×10^{-3} and 3.5×10^{-3} $MJ \cdot s^{-1}$ (Amiro 1992a), of which about 60% goes toward space heating. Because energy consumption shows little long-term variability, we set FUELUS equal to a constant of 3.5×10^{-3} $MJ \cdot s^{-1}$. This value is conservative because it reflects maximum consumption rates and includes energy used for purposes other than space heating.

FUELUS is also used in Section 9.1.1.3 to calculate the areas of the woodlot and peat bog required to heat the home of the critical group. For this calculation, FUELUS is expressed in units of $MJ \cdot a^{-1}$, in which case it has a value of 1.1×10^5 $MJ \cdot a^{-1}$.

7.5.2.2 Energy Content of Wood, EW
($MJ \cdot kg^{-1}$ wet biomass)

The convertible energy content of wood fuel is required to calculate the amount of wood burned by the critical group for home heating or other purposes (Equation (7.27)). The average energy content of typical Canadian Shield forest species is about 11 $MJ \cdot kg^{-1}$ on a wet weight basis (Tillman 1978). We assume that wood stoves are 50% efficient (Sexton et al. 1984) so that the convertible energy content, EW, is 5.5 $MJ \cdot kg^{-1}$ wet biomass. We adopt this value for the postclosure assessment and assume it is constant.

7.5.2.3 Energy Content of Peat, EP
($MJ \cdot kg^{-1}$ dry peat)

The convertible energy content of peat is required to calculate the amount of peat burned by the critical group for space heating or other purposes (Equation (7.28)). The average net energy content of dry, milled peat is about 10 $MJ \cdot kg^{-1}$ (Mustonen 1984). Assuming that stoves perform as efficiently for peat as for wood (Section 7.5.2.2), we derive a convertible energy content of 5 $MJ \cdot kg^{-1}$ dry peat. As in the case of wood, we do not distribute EP values.

7.5.2.4 Forest Yield, FY
(kg wet biomass$\cdot$m^{-2} land)

Forest yield refers to the biomass actually burned in a fire, and is required to estimate the nuclide flux to the atmosphere caused by a forest or land-clearing fire (Equation (7.29)). Fires on the Canadian Shield typically engulf about 2.2 kg wet biomass$\cdot$m^{-2} land, with a range from 1.0 to 10 kg$\cdot$m^{-2} (Van Wagner 1983). Assuming that FY is lognormally distributed, we set its GM equal to 2.2 kg wet biomass$\cdot$m^{-2} land, and cover the expected range of values by choosing a GSD of 1.6.

7.5.2.5 Peat Yield, PY
(kg dry peat$\cdot$m^{-2} land)

Peat yield refers to the mass of peat burned in an outdoor fire, and is required to estimate the nuclide flux to the atmosphere caused by a forest or land-clearing fire (Equation (7.29)). We calculate PY from

$$PY = Z_s \cdot \rho_s \qquad (7.44)$$

where Z_s (m) is the depth of the peat burned and ρ_s (kg dry peat$\cdot$m^{-3} peat) is its bulk density. In a land-clearing fire, it is likely that a depth of peat equal to the root-zone depth would be burned. We assume that the same depth of peat would be burned in a forest fire. Accordingly, we set $Z_s = 0.3$ m (Section 6.1). In Section 6.5.1.3, the bulk density of organic soil or peat was given as 150 kg dry soil$\cdot$m^{-3} soil. Substituting these values in Equation (7.44) gives PY = 45 kg dry peat$\cdot$m^{-2} land. PY values are not distributed in BIOTRAC.

7.5.2.6 Emission Fraction, $EMFRAC^i$
(unitless)

$EMFRAC^i$ defines the fraction of nuclide i that is released from biomass by burning. Three different values of $EMFRAC^i$ are needed for each nuclide to calculate fluxes to the atmosphere from agricultural fires ($(EMFRAC^i)_{AF}$, (Equation (7.24)), energy fires ($(EMFRAC^i)_{EF}$, Equations (7.27) and (7.28)), and forest and land-clearing fires ($(EMFRAC^i)_{LF}$, Equation (7.29)). The emission fractions for all three types of fires are set to zero for the argon and krypton radionuclides, which do not accumulate in plants because they are inert gases. Values for tritium are not required because tritium is handled in a different manner (Section 7.3.2.1).

For all three types of fires, we assume that ^{14}C, ^{129}I and radon are released as gases and that their entire inventory reaches the atmosphere. Therefore, $(EMFRAC^i)_{AF}$, $(EMFRAC^i)_{EF}$, $(EMFRAC^i)_{LF}$ have a value of 1 for these radionuclides.

For agricultural fires, the remaining nuclides that are released are likely attached to smoke particles. Few data are available for now to define $EMFRAC^i$ values for this pathway. The most appropriate data are from studies of nutrient losses from burning heather (Evans C.C. and Allen 1971), which indicate that 10% to 20% of the initial inventory of many elements is lost. We assume these values apply to our nuclides and to fires involving other types of vegetation, and set $(EMFRAC^i)_{AF} = 0.2$ for all the radionuclides except ^{39}Ar, ^{14}C, ^{3}H, ^{129}I, ^{81}Kr, ^{85}Kr and ^{222}Rn. $(EMFRAC^i)_{AF}$ values are not distributed.

Data are also scarce for defining emission fractions from energy fires. Information on nuclide emissions from commercial peat power plants and woodburning fireplaces suggest that about 20% of the contaminant inventory of the fuel is lost to the atmosphere (Amiro 1992b). We assume that this value applies to both wood and peat, and set $(EMFRAC^i)_{EF} = 0.2$ for all nuclides except ^{39}Ar, ^{14}C, ^{3}H, ^{129}I, ^{81}Kr, ^{85}Kr and ^{222}Rn. $(EMFRAC^i)_{EF}$ values are not distributed.

Forest fires burn much hotter than controlled agricultural fires and may have surface temperatures in excess of 1000°C (Smith D.W. and Sparling 1966). At these temperatures, many nuclides may volatilize and be released in large amounts (Grier 1975). However, it is difficult to account for this effect quantitatively because of the limited amount of data on fire temperatures. Moreover, the chemical form of the nuclides in plants is generally unknown, so that boiling points cannot be specified. Given this uncertainty, we conservatively assume that the total inventory of all the nuclides (except ^{39}Ar, ^{3}H, ^{81}Kr and ^{85}Kr) is released in land fires, and give $(EMFRAC^i)_{LF}$ a value of 1. $(EMFRAC^i)_{LF}$ values are not distributed.

7.5.2.7 Probability of Peat Fuel Use, PT
(unitless)

We assume that the home of the critical group is heated either by wood or peat. We are not aware of any available data on peat use for heating on the Canadian Shield, but peat use for heating is likely not very common. Accordingly, we assume that peat is used for home heating 1% of the time when organic soil has been chosen as the soil type, which is 14% of the time (Table 6-7). When organic soil is not chosen, no peat is available to the critical group. In BIOTRAC, peat use as fuel is expressed by a switch, PT, which has a value of 0.01 subject to the restriction indicated above. This means that only few BIOTRAC simulations will involve peat fuel.

7.5.3 Dispersion Parameters

7.5.3.1 Wind Speed, UCAV
$(m \cdot s^{-1})$

The annual average wind speed, UCAV, at a generic Canadian Shield site is used in Equations (7.6) and (7.7) to calculate the dispersion factor for elevated point sources. Wind speed data for the Canadian Shield are readily available from the AES, which has made routine measurements at several locations for many years (Environment Canada 1982a). We calculated annual average speeds for 16 sites widely spread across the Ontario portion of the Shield (Amiro 1992b). Equation (7.7) requires a wind speed measured at a height of about 3 m, which corresponds to the top of a chimney on a typical one-storey house (Section 7.5.3.4). Accordingly, the mean speeds for each site, which were obtained at a nominal height of 10 m, were extrapolated to 3 m (Amiro 1992b). The speeds at a height of 3 m were distributed approximately normally, with a mean of 2.36 $m \cdot s^{-1}$ and an SD of 0.64 $m \cdot s^{-1}$. This value is not only a measure of spatial variability, but also accounts for most temporal variability because year-to-year variations at any one site are generally much less than site-to-site variations.

For postclosure assessment, we adopt a normal distribution for UCAV, with a mean of 2.36 $m \cdot s^{-1}$ and an SD 0.64 $m \cdot s^{-1}$. The distribution was truncated at its low end at three SDs (UCAV = 0.44 $m \cdot s^{-1}$) because lower annual average wind speeds are not probable.

7.5.3.2 Wind Speed Weighting Factor, UWGHT (unitless)

The wind speed weighting factor is an annual average wind speed for a generic Canadian Shield site normalized by the wind speed averaged spatially across the Shield. This involved a comparison of long-term data from WL and data from other sites on the Canadian Shield (Amiro 1992c). It is used in Equations (7.4) and (7.5) to calculate the dispersion factors for ground-level area sources. UWGHT is simply a non-dimensional form of UCAV (Section 7.5.3.1). A value of UWGHT for each BIOTRAC simulation is calculated by dividing the sampled value of UCAV by the mean value of UCAV. Thus,

$$UWGHT = UCAV/2.36\ m \cdot s^{-1} \quad . \qquad (7.45)$$

7.5.3.3 Building Width, BW (m)

The building width and building height, BH (Section 7.5.3.4), define the size of the cavity created by the building into which contaminants released from a chimney become entrained (Section 7.3.1.2). These dimensions also define the building volume (Equation (7.9)) and the volume of air that is available for diluting contaminants released inside the building occupied by the critical group. Ultimately, BW and BH are used in Equations (7.7) and (7.8) to calculate dispersion factors for elevated point sources and indoor releases.

The available data on building dimensions are given in terms of floor area. Floor areas for new dwellings in Canada are distributed lognormally, with a GM of 95 m^2 and a GSD of 1.42 (CMHC 1987). We assume a square building, so that BW is distributed lognormally with a GM of 9.7 m and a GSD of 1.2. This distribution is truncated at the lower end at 8.4 m because houses with floor areas smaller than 70 m^2 are very rarely built.

7.5.3.4 Building Height, BH (m)

Building height is also required in Equations (7.7) and (7.8) to calculate the dispersion factors for elevated point sources and indoor releases. We assume that the critical group lives in a single-storey house, and set BH equal to a fixed value of 2.4 m. This is a minimum reasonable value for both ceiling and building heights, and leads to conservative estimates for the dispersion factors, which are inversely proportional to BH.

7.5.3.5 Entrainment Parameter, KK (unitless)

The building wake entrainment parameter is required in Equation (7.7) to calculate the dispersion factor for elevated point sources. KK is an empirical constant that varies between 0.2 and 2.0 (Hanna et al. 1982). Its exact value depends on the geometry of the building and its orientation with respect to wind direction. We conservatively set KK at its maximum possible value of 2.0 because these factors are unknown at a generic site.

7.5.3.6 Infiltration Rate, INFILT (s^{-1})

Building infiltration rates are required to estimate the volume of indoor air into which nuclides released through domestic water use become mixed. INFILT is used in Equation (7.8) to calculate the dispersion factor for indoor releases. Infiltration rates are variable and have decreased in recent times as homes have become more energy-efficient. However, there is likely a practical lower limit to INFILT below which gases such as CO_2 would build up to unacceptably high levels. The Canadian Standards Association (CSA 1989) recommends a minimum exchange rate of 0.35 h^{-1}, which is substantially lower than the rates characteristic of today's homes. Since indoor air concentrations are inversely proportional to the infiltration rate, we conservatively adopt this minimum value and set INFILT at 0.35 h^{-1}, which corresponds to 0.0058 s^{-1}.

7.5.4 Deposition Parameters

7.5.4.1 Dry Deposition Velocity, Vd ($m \cdot d^{-1}$)

The deposition velocity is a transfer coefficient used in Equation (7.35) to calculate the flux of nuclides from the atmosphere to the underlying surface as a result of dry deposition. It is a bulk parameter that incorporates many factors and processes, including the physical and chemical form of the contaminant, properties of the underlying surface (soil characteristics, vegetation cover and surface roughness), meteorological conditions (wind speed and atmospheric stability), and the measurement height. As a result, most of the variability in Vd is caused by local temporal or spatial variability rather than by variability in the long-term average conditions across the Canadian Shield.

Using a model that takes into account particle size, particle density, surface roughness and meteorological conditions (Sehmel 1980), Amiro (1992b) calculated an average deposition velocity for Canadian Shield conditions of 0.006 $m \cdot s^{-1}$. We assume that Vd values are distributed lognormally, and adopt this average value as the GM. A GSD of 2.0 results in a distribution that covers most of the range of measured values reported by Sehmel (1980). Therefore, our PDF for nuclides subject to deposition is lognormal with a GM of 0.006 $m \cdot s^{-1}$ and a GSD of 2.0. In units of $m \cdot d^{-1}$, as required by Equation (7.35), Vd is distributed lognormally, with a GM of 518 $m \cdot d^{-1}$ and a GSD of 2.0. We assume that this PDF applies to both particles and gases, and to deposition to both soil and vegetation. We set Vd = 0 for ^{39}Ar, ^{3}H, ^{81}Kr, ^{85}Kr, and ^{222}Rn because these nuclides are not deposited (Section 7.3.8). For ^{14}C deposition to vegetation, Vd = 0 because deposition is already accounted for by the plant/soil concentration ratio (Section 8.5.1.1).

Deposition fluxes are used in the soil model in units of $mol \cdot m^{-2} \cdot a^{-1}$ (Equation (6.16)), in which case Vd must be expressed in units of $m \cdot a^{-1}$. In these units, our PDF for Vd is lognormal with a GM of 1.89×10^5 $m \cdot a^{-1}$ and a GSD of 2.0.

7.5.4.2 Washout Ratio, Wr
(unitless)

The washout ratio is the ratio of the nuclide concentration in precipitation reaching the ground to the air concentration of the nuclide. It is used in Equation (7.36) to estimate the nuclide flux from the atmosphere to the underlying surface as a result of wet deposition. Most of the available data apply to particulate matter, and were obtained in studies involving natural aerosols and fallout from nuclear weapons tests (Slinn 1978). Washout ratios for gases are generally smaller than for particulates. We assume Wr is lognormally distributed (Barrie and Nuestadter 1983), and choose a GM of 2.5×10^5 and a GSD of 1.58 to cover the range of reported values (Slinn 1978). This distribution applies to all nuclides except ^{39}Ar, ^{14}C, ^{3}H, ^{81}Kr, ^{85}Kr and ^{222}Rn for which Wr = 0. As noted in Section 7.5.4.1, the deposition of ^{14}C to vegetation is accounted for by the plant/soil concentration ratio.

The median value of 0.01 $m \cdot s^{-1}$ for our combined wet and dry deposition velocity (Vd + Wr·P) is about a factor of three higher than the value of 0.003 $m \cdot s^{-1}$ recommended by the Canadian Standards Association (CSA 1987) for particulate iodine, and is therefore conservative. Our median value is comparable to the CSA value recommended for iodine gas (I_2).

7.5.5 Parameters Documented Elsewhere

The atmosphere submodel contains a number of parameters that also appear in other parts of BIOTRAC. The parameters related to the surface water and soil submodels are documented in Chapters 5 and 6, and those pertaining to the food-chain and dose submodel are documented in Chapters 8 and 9. All these parameters are listed in Table 7-3 together with the sections in which they are discussed in detail and documented.

7.6 MODEL VALIDATION

In the case of the atmosphere model, validation means showing that the processes of suspension, dispersion and deposition are properly modelled, and that the predicted air concentrations and deposition fluxes are realistic. Only a few components of the model have been validated experimentally.

Full-scale field testing of suspension models is extremely difficult, as the scarcity of published, validated models attests. The strengths of natural analog sources are invariably too weak to allow air concentrations to be measured with any certainty. Tracers must then be employed, but the use of radioactive tracers is often unacceptable. There are practical difficulties in applying a tracer uniformly over a large area. Costs can

TABLE 7-3

ATMOSPHERE SUBMODEL PARAMETERS COMMON TO OTHER SUBMODELS AND THEIR SECTIONS OF DOCUMENTATION

Parameter	Equation	Section Where Documented
Area of terrestrial contamination, A_T (m^2)	7.4	9.1.1.3
Area of the lake, A_l (m^2)	7.5, 7.10, 7.11	5.5.3
Mean depth of the lake, Z_l (m)	7.22	5.5.2
Soil bulk density, ρ_s (kg dry soil$\cdot m^{-3}$ soil)	7.18, 7.21, 7.23	6.6.1.3
Evasion rate from soil for C, I and Se, η_s^i (s^{-1})	7.18, 7.21, 7.23	6.5.4
Evasion rate from water for C, η_l^c (s^{-1})	7.22	5.5.9
Crop yield, Y_j (kg wet biomass$\cdot m^{-2}$ soil)	7.24	8.5.8.1
Plant/soil concentration ratio, Bv^i ((mol$\cdot kg^{-1}$ wet biomass)/(mol$\cdot kg^{-1}$ dry soil))	7.25	8.5.1.1
Water demand per person, Uwc (m^3 water$\cdot a^{-1}\cdot p^{-1}$)	7.32	9.1.1.4
Number of people per household, Nph	7.32	9.1.1.1
Precipitation rate, P (m water$\cdot d^{-1}$)	7.36	9.1.3

become prohibitive as large amounts of tracer are required to achieve detectable air concentrations. A very extensive program would be required to validate models for all of the potential suspension mechanisms.

Of the many suspension mechanisms that we model, only one has been validated experimentally. This is the model for q^{Rn}, the terrestrial radon emission rate (Section 7.3.4.1), which was developed using data from radium-rich mine tailings (Rogers et al. 1980). Values derived from this model agree well with experimental measurements reported by Pearson (1967). Furthermore, when combined with soil radium concentrations, the model predicts a range of radon fluxes comparable to observed values (UNSCEAR 1982).

Dispersion models have been validated better than suspension models because field studies of atmospheric dispersion are relatively easy to carry out. The model that we use to derive the dispersion factors for ground-level area sources (Section 7.3.1.1) has received extensive validation. Its predictions agree well with experimental data, and with analytical solutions (Wilson et al. 1981). The model also reflects the understanding that was gained in a series of atmospheric diffusion trials that we conducted over the Canadian Shield (Davis et al. 1986).

Deposition experiments suffer from many of the difficulties that beset suspension work. For full-scale field studies, it is often necessary to resort to tracers, with their attendant challenges. The models for both wet and dry deposition are usually expressed through bulk parameters that incorporate many processes and are difficult to generalize and apply to specific conditions. On the other hand, we derived our dry deposition velocities (Section 7.5.4.1) from a model that has been validated experimentally using a wide range of measurements involving both particulate and gaseous deposition to a variety of surfaces (Sehmel 1980).

In the absence of full validation with experimental or field data, confidence in the atmosphere model must be demonstrated in other ways. These other approaches to validation are discussed in Section 7.7.2 and in Chapter 11.

7.7 MODEL DISCUSSION

7.7.1 Assumptions

Several assumptions were made in deriving the atmosphere model. In this section, we explain and review the main assumptions and discuss their effect on the predicted air concentrations and deposition rates.

1. *The physical layout of the garden, forage field, woodlot and peat bog, and their positions relative to the building occupied by the critical group are not known for our generic site.* This makes it difficult to calculate appropriate dispersion factors or to account for the contribution of nuclides suspended from one field to the air concentration over another. We have compensated for this uncertainty by adopting a field geometry and a receptor location that maximize the air concentration. Moreover, we have assumed that air concentrations arising from suspension over one field apply over all fields with no reduction from dispersion. These are conservative assumptions, ensuring that air concentrations and deposition rates are not underestimated.

2. *Complex suspension and deposition processes are represented by very simple transfer models.* We believe that this is appropriate, given the large number of relevant processes and our current understanding of how they operate. The transfer coefficients that we have employed are empirically based, and incorporate the effects of many processes without the need for a complete theoretical model. This includes physical, chemical and biological forces that act together to determine nuclide transport rates. Mass loading parameters and deposition velocities are used in many other models of nuclide transport in the atmosphere (Healy 1980, USNRC 1977, Napier et al. 1980).

3. *The model accounts for all the suspension processes that could contribute significantly to air concentrations.* We have not explicitly included all the processes, particularly when they originate from cultural practices. Many human activities such as sanding a wall in an enclosed space would increase the particulate load briefly, but would not contribute substantially to long-term air concentrations. We have not considered large-scale industrial operations such as mining or the commercial production of energy from biomass. We assume that the area of contaminated land is not large enough to support large-scale industry, or if it is, the facility would be located too far from the region occupied by the critical group and other biota to contribute significantly to the air concentration of nuclides from the vault.

   Many other anthropogenic suspension mechanisms are included implicitly through bulk transfer parameters, particularly the atmospheric dust load, ADL (Section 7.5.1.1). Furthermore, where we have modelled a cultural pathway explicitly, we did so conservatively. For example, the critical group burns the stubble from its forage field every year; when peat is used as a source of fuel, it is always available, regardless of how previous generations have used the peat bog; releases from the chimney are always caught in the wake cavity of the home; and the dispersion factors derived for passive sources are applied to fires. Throughout the model, we have assumed that each nuclide can be in a variety of chemical forms, consistent with the physical, chemical and biological environment.

   Finally, we calculate the total air concentration by summing the contributions from all pathways (Equation (7.34)). This will result in double accounting for some nuclides. For example, potentially volatile nuclides contribute in both particulate and gaseous forms, and particles originating from lake water are added separately, although they would be included in the parameter ADL. By these means, we have ensured that air concentrations are not underestimated, even though not all suspension mechanisms have been considered explicitly.

4. *Our model of particle suspension from terrestrial sources assumes that all sizes of soil particles are equally contaminated.* In fact, finer particles could be more highly contaminated. Such particles are frequently composed of clay, and nuclides tend to sorb to clay. It is the fine particles that become truly suspended in the atmosphere, and so our nuclide concentrations in air could be underestimates (Equation (7.12)). However, any underpredictions are likely offset by the inclusion in ADL of particles from distant sources that would be uncontaminated.

5. *Air concentrations are not reduced when contaminants are lost from the plume through deposition.* The critical group and other biota live very close to the suspension sources at the discharge zones so that there is little time for deposition to occur, and little reduction in air concentration. This assumption is therefore conservative, but not overly so. It allows us to choose high values

for the deposition velocity, Vd (Section 7.5.4.1), and so predict conservative deposition rates without underestimating air concentrations.

6. *The model does not account for the effects of variations in parameter values throughout the year.* In reality, meteorological parameters show strong daily and seasonal variations, and suspension rates for many natural processes decrease in winter when the ground and lakes are frozen and snow-covered. In addition, suspension mechanisms such as wind erosion and fires occur episodically. However, we assume that all the processes operate continuously and drive the model with parameter values that are annual averages, at least nominally. In fact, data for many transfer parameters, including AADL, AIML, ATC^{Rn} and q^{Rn}, represent summer conditions only. Since winter values are generally lower, use of the data as annual averages results in an overprediction of air concentrations and deposition rates. Only ADL, INDRN and the parameters incorporating meteorological data have values that represent true annual averages.

   Seasonal effects can be safely ignored when calculating air concentrations for predicting inhalation and immersion doses for humans. Since the atmosphere model is linear, the concentration calculated using annual average parameter values equals the average concentration of the concentration values made at various times throughout the year. The situation is somewhat different for deposition to vegetation, which occurs mainly under summer conditions. Annual average values may be inappropriate in this case if they differ significantly from summer values. For example, the summer ADL value is slightly higher than its annual average. Summer air concentrations and deposition fluxes to crops are therefore slightly underestimated for nuclides originating from terrestrial sources. On the other hand, deposition rates will be accurately predicted where air concentrations depend on parameters whose annual average values are based on summer conditions only. This is the case for gaseous nuclides such as ^{129}I because the data were derived for summer conditions only.

   The use of annual average parameter values in the fire pathways raises similar points. In the models, nuclides released via fires enter the atmosphere at a uniform rate throughout the year. However, they are deposited to crops over a two- to three-month growing season only (Section 8.5.3.2). The total amount deposited is therefore less than it would have been had we assumed that the fire event occurred entirely within the growing season. Although this treatment appears to be non-conservative, it allows for fires that occur in the spring before the crops come up, or in the fall after harvest. Similarly, wood or peat burning for energy occurs primarily in winter, and in reality could not contribute to crop contamination because contaminants would have dispersed before the crop emerges.

To summarize, the use of annual average parameter values can result in either conservative or non-conservative predictions, depending on the pathway and the available data. Close examination of our models, parameter values and arguments can only lead to the conclusion that this practice will not underestimate the total air concentration or the total deposition fluxes predicted by BIOTRAC.

7. *Long-term changes to the climate, or to the parameters that control suspension, dispersion and deposition, are modelled implicitly.* We assume that our parameter distributions are wide enough to describe the relevant biosphere as long as current interglacial conditions persist. This is likely to be the case for some time (Chapter 12). Our PDFs reflect primarily spatial variability across the Canadian Shield, which will probably exceed the temporal changes experienced at any one site over the next 10 000 a. As far as the atmosphere model is concerned, we assume that processes such as acid rain and global warming are included in our parameter distributions. It is impossible to predict future cultural practices, and we assume that they will not involve significant new suspension or deposition pathways. With these restrictions, the model is suitable for predicting air concentrations and deposition fluxes over at least the next 10 000 a.

7.7.2 Evaluation

The atmosphere submodel involves a large number of processes and pathways, many of which are related to cultural activities. The complexity of the model chosen to simulate each process reflects our level of understanding and the amount and quality of the available data. Occasionally, a detailed, mechanistic treatment is warranted, but more commonly a lack of information dictates a simpler approach using transfer coefficients. The latter approach is appropriate for assessment models (USNRC 1977, Healy 1980, Napier et al. 1980), and is used internationally in waste management assessments (Bergström et al. 1982, Korhonen and Savolainen 1982). To offset the uncertainty that exists in most of the processes, we have incorporated many conservative assumptions into the models and selected conservative parameter values. Where possible, we use models that have been validated experimentally. The atmosphere submodel was improved through comments and criticisms of an interdisciplinary team of scientists from the International Institute of Applied Systems Analysis (IIASA) in Vienna, Austria (Amiro 1992b).

The submodel was specifically designed for the postclosure assessment. In particular, it accounts for the underground location of the vault by treating contaminated water, soil and vegetation as the source of airborne nuclides. It was formulated to provide annual average air concentrations at a generic Canadian Shield site. Its simplified, efficient structure, together with its distributed parameter values make it suitable for a probabilistic postclosure assessment. It was designed to interface smoothly with the geosphere model and the other three submodels of BIOTRAC.

The parameter values and distributions used in the atmosphere submodel were derived from the best available data measured under Canadian Shield conditions. Most of the information was extracted from the literature, but some was supplied by our own studies (Davis and Reimer 1980). The data available for defining some parameters were quite limited. In these cases, we have chosen what we believe are conservative values.

We conclude that the atmosphere submodel and its associated database provide a reasonable description of the atmospheric pathways relevant to the postclosure assessment of the concept for disposal of Canada's nuclear fuel waste. They will not underestimate air concentrations or deposition rates, and, therefore, consequences to humans and other biota.

8. THE FOOD-CHAIN AND DOSE SUBMODEL

8.1 THE FOOD-CHAIN COMPARTMENT

Nuclides in the physical compartments of the biosphere (surface water, sediment, soil and the atmosphere) may be taken up by living organisms, both plant and animal. These nuclides may affect the organisms and may move along the food chain and eventually be ingested by humans. All biota, including humans, that live in the contaminated environment may be exposed internally and externally to radiation fields. This chapter describes the model developed to trace nuclides through the food chain to humans and other organisms, and to calculate doses from both internal and external exposure pathways. The information was extracted from the food-chain and dose submodel report (Zach and Sheppard 1992), which contains further details. The model has also been published in the open literature (Zach and Sheppard 1991). For convenience, the food-chain and dose submodel has been given the acronym CALDOS (CALculation of DOSe).

The human individual of concern in CALDOS is a member of the critical group (Section 1.5.4), and is represented by ICRP reference man (ICRP 1975). This individual lives his entire life at the discharge zone. He experiences a generic Canadian Shield environment, and has a lifestyle similar to that of present-day residents of the Canadian Shield in Ontario. However, he is entirely self-sufficient, drawing all of his resources, including food, water, building materials and heating fuel, from the local potentially contaminated environment. He grows the types of crops and raises the types of livestock that are found on family farms on the Shield today. He may also eat foods native to the Shield, including wild plants (berries, wild rice and mushrooms), wild game (moose, deer, beaver, game birds and waterfowl), fish and other natural foodstuffs such as honey and maple syrup (Section 2.1). He stores his summer produce for winter consumption, and so eats potentially contaminated food throughout the year.

For modelling purposes, the diet of the critical group is made up of five very general food types: terrestrial plant foods (TE PLANT), milk and dairy products (TE MILK), mammalian meats (TE MEAT), poultry and eggs (TE BIRD) and freshwater fish (FW FISH). Food-chain transfer coefficients of nuclides are not known well enough to consider other food types explicitly in the model. However, the distributions of transfer coefficients for our five food types are wide enough to encompass all the likely food sources, both domestic and wild, of a Canadian Shield resident. TE PLANT consists primarily of grains, vegetables and fruit, but also covers berries, mushrooms and wild rice. TE MILK refers mainly to milk and dairy products derived from cows, but also includes goats. The parameter values for TE MEAT apply mainly to beef, but are distributed widely enough to include pork, mutton and venison as well. Transfer coefficients for TE BIRD are based primarily on data from chickens, but our distributions include turkeys, ducks, geese and various wild fowl. FW FISH refers mainly to Canadian Shield species such as lake trout, lake whitefish, northern pike and walleye.

Human diet is treated probabilistically in CALDOS, so that members of the critical group draw different proportions of their food needs from the five food types in each simulation. In this way, we include diets ranging from almost purely vegetarian or purely meat-eating to diets consisting of 50% fish. The assumption that the critical group is represented by ICRP reference man (Section 1.5.4) places some restrictions on diet, particularly with regard to total caloric intake (Section 8.5.6.1).

All of the domestic plant foods that humans consume are assumed to be grown in the garden. Wild plant species eaten by humans are assumed to be grown on soils contaminated to the same extent as the garden. Similarly, all plants eaten by animals that are in turn consumed by humans are assumed to be grown on the forage field, or on soils with a nuclide concentration equal to that of the forage field. Water for both man and animals comes from either a lake or a well, depending on which is chosen as the source of the domestic water supply (Section 9.1.2).

CALDOS can also be used to calculate radiation doses to non-human organisms because it provides radionuclide concentrations in several kinds of organisms through the food types TE PLANT, TE MEAT, TE BIRD and FW FISH. These concentrations are based on several broadly distributed parameters (Section 8.5), making the food types very general. It also makes the food types representative of a wide variety of organisms on the Canadian Shield. Thus, the food types are suitable for defining generic target organisms for calculating radiation doses to non-human biota, as explained in Section 13.3. To enhance clarity, Chapter 8 focuses on human dose prediction and Chapter 13 on dose prediction for other biota.

The objective of the food-chain and dose submodel is to trace nuclide movement through the food chain, and to estimate concentrations in Canadian Shield biota and doses to members of the critical group from all the important exposure pathways, given the nuclide concentrations in surface water, well water, soil and the atmosphere. Doses to humans are one of the primary end points of the assessment for determining the acceptability of the disposal concept (AECB 1987, Federal Environmental Assessment Review Panel 1992).

8.2 QUALITATIVE DESCRIPTION OF CALDOS

Concern for human safety has prompted many studies of nuclide transport through the food chain. Work in this area began in the late 1940s in response to the development of nuclear weapons programs. It received further impetus from the growth of the nuclear power industry in the 1970s and the establishment of national nuclear fuel waste management programs in the 1980s.

The knowledge accumulated in these studies has given rise to a multitude of food-chain and dose models, most of which were developed to assess conventional nuclear power installations (Hoffman et al. 1977, Kaye et al. 1982, Till and Meyer 1983). Most of these models are very similar in concept, structure and formulation (Fletcher and Dotson 1971, Baker 1977, Moore et al. 1979, Shaeffer and Etnier 1979, Napier et al. 1980). They have been broadly used and accepted by scientists and in regulations (USNRC 1977, 1983b, IAEA 1982, NCRP 1984). They form the basis for the guidelines put forward by the Canadian Standards Association for calculating dose to humans from routine releases from nuclear reactors for electricity generation (CSA 1987). They also provide the basis for the food-chain models developed by a number of countries for assessing the geological disposal of nuclear fuel waste (Bergström et al. 1982, Smith J.M. et al. 1985, USEPA 1985).

In most of these food-chain models, the dose, Dp ($Sv \cdot a^{-1}$), resulting from a particular exposure pathway is calculated using a simple multiplicative chain of the form

$$Dp = C \cdot TC \cdot U \cdot DCF \quad . \qquad (8.1)$$

Here, C is the nuclide concentration in the environmental compartment that acts as the source of contamination for the food chain, TC is a transfer coefficient that predicts the nuclide concentration in the components of the food chain between the source compartment and man, U is a use factor that describes man's utilization rate of the food-chain components, and DCF is a dose conversion factor. For example, for man's plant ingestion pathway, C would be the nuclide concentration in soil ($Bq \cdot kg^{-1}$ dry soil), TC would be the plant/soil concentration ratio (($Bq \cdot kg^{-1}$ wet biomass)/($Bq \cdot kg^{-1}$ dry soil)), U would be man's ingestion rate of TE PLANT (kg wet biomass$\cdot a^{-1}$), and DCF would be the ingestion dose conversion factor ($Sv \cdot Bq^{-1}$).

Most food-chain models assume steady-state conditions. Concentrations in plants, animals and humans adjust very rapidly to changes in concentration in the physical environmental compartments. The transient aspects of food-chain transfer are therefore not important when annual average doses are being calculated. Dynamic food-chain models do exist (McDowell-Boyer et al. 1980, Simmonds and Linsley 1981, Whicker and Kirchner 1987), but they are complex and suffer from a limited database.

We have based CALDOS largely on the food-chain models developed to assess nuclear power stations to take advantage of the accumulated body of knowledge that these models represent. The concepts that apply to nuclide releases from power installations carry over to waste disposal because transport through the food chain is largely independent of the mechanisms through which the environment becomes contaminated. Accordingly, CALDOS is essentially a steady-state, multiplicative chain model. However, it does differ from the traditional models (and in particular from the guidelines promulgated by the Canadian Standards Association (CSA 1987)) in a number of ways (Section 9.5.2, Zach and Sheppard 1992). It treats more exposure pathways than is traditional to allow for the uncertainties associated with the long time frame of the postclosure assessment. It is dynamic to the extent that it allows for the ingrowth of daughter radionuclides in plants and animals. It treats a number of nuclides with unique properties in special ways. Finally, it reflects the knowledge gained from recent studies on food-chain transfer conducted under the auspices of various nuclear fuel waste management programs around the world, including the Canadian program.

CALDOS includes all the internal and external pathways that could contribute significantly to human exposure. It includes all the pathways that appear in the most commonly used food-chain models, as well as several additional pathways of potential importance in the waste management context (Zach and Sheppard 1991).

The internal pathways considered in CALDOS are the ingestion of contaminated plants, water and soil by humans; the ingestion of animals and fish that have consumed contaminated plants, water and soil; and the inhalation of contaminated air by humans (Figure 8-1). We consider a number of different processes in modelling these pathways. Nuclides are transferred to plants, animals and fish from the physical environment. The plants eaten by humans and animals are allowed to become contaminated in three different ways: by root uptake from contaminated soil, by deposition to leaves from contaminated air, and by application of contaminated water to leaves during aerial irrigation. Nuclides deposited onto leaf surfaces are subject to removal by environmental processes such as wind and water action, and by plant growth. By considering holdup times, activity is lost through radioactive decay between the time that a plant or animal is removed from the contaminated source through harvest or slaughter and the time it is ingested by humans. The decay of precursor radionuclides throughout the life of a plant or animal is allowed to contribute to the buildup of daughters. Doses arising from inhalation are calculated from both indoor and outdoor air concentrations (Section 7.3), taking into account the amount of time that members of the critical group spend in the two locations.

The external pathways considered in CALDOS are immersion in contaminated air, immersion in contaminated water, exposure to contaminated soil and exposure to contaminated building materials (Figure 8-2). Doses from the first three of these pathways and from inorganic building materials are calculated very simply from predicted nuclide concentrations in water, soil and air (Chapters 5, 6 and 7), and from assumptions about exposure times. Where building materials are made from wood, the processes discussed above relating to crop contamination apply.

Many of the nuclides reaching the food chain would be returned to the physical environment with waste products or through the death and decay of the organisms involved. Recycling is a complex process and has not been modelled explicitly in CALDOS. Instead, it has been accounted for implicitly by assuming that uptake and transfer in most pathways do not deplete the concentrations or inventories of the source compartments. Where source concentrations are reduced (e.g., when gaseous nuclides evade the soil),

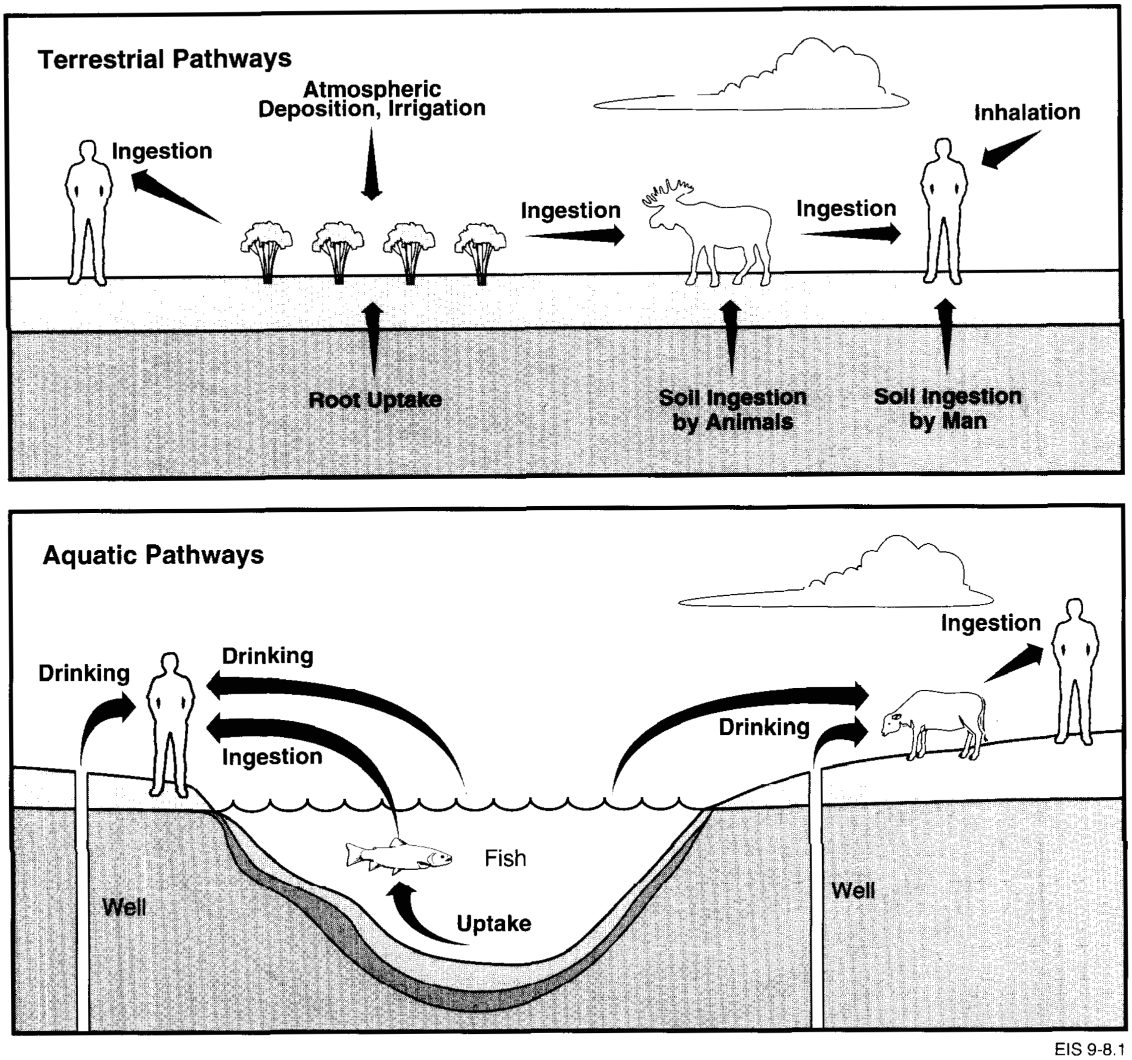


FIGURE 8-1: Internal Human Exposure Pathways Considered in CALDOS

the transfer parameters generally describe the net effect of the process. In this way, the consequences of recycling can be accounted for simply and conservatively. However, this implies that nuclides are not redistributed among source compartments by the action of recycling.

CALDOS calculates concentrations in plants and animals before they have been processed for human consumption. These concentrations could be altered during food preparation. Cooking may drive off volatile nuclides (Danfors 1986, Lofti et al. 1989), but can also concentrate others (Buma and Meerstra 1964). In CALDOS, we assumed that preparation neither dilutes nor concentrates nuclides. We further assumed that most soil particles attached to plant surfaces are removed by washing or peeling prior to ingestion. Particles that may remain are accounted for in man's soil ingestion pathway, which considers a soil ingestion component related to plant ingestion.

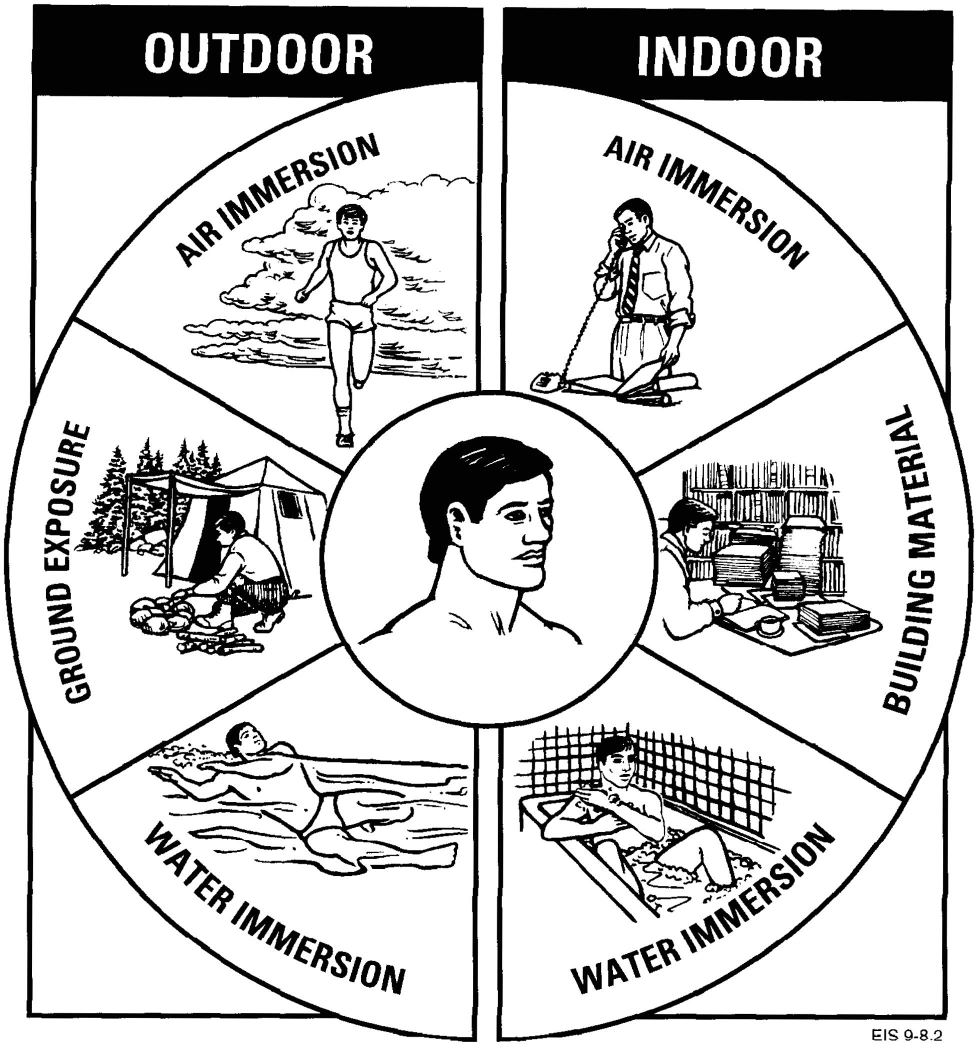


FIGURE 8-2: External Human Exposure Pathways Considered in CALDOS

Doses from the internal exposure pathways depend on the amount of contaminated food, water and air taken into the body. Food and water ingestion rates and the inhalation rate are highly correlated for a given individual, and in CALDOS are calculated in a consistent manner using a simplified version of the Energy Water Air Model (EWAM) developed by Zach and Barnard (1985, 1987). This model translates the total energy needs of the individual into ingestion rates of each food type in the diet, given the proportion of the diet made up by each food type and the nutritional contents of the foods. These rates are then used to predict the associated water ingestion and inhalation rates. In general, the total energy need is variable, and for an individual can be calculated from a consideration of body mass, activity level, age, sex, and ambient temperature. However, for the postclosure assessment we specify a fixed energy need consistent with ICRP reference man, for whom doses are calculated.

Internal doses for a number of nuclides with special properties are calculated without tracing them through the food chain. We assume that the specific activity of tritium is constant throughout the biosphere, so that tritium concentrations in humans and the total internal dose can be calculated using a single DCF and the predicted tritium concentrations in lake or well water (Section 2.5.1). A limited specific-activity model is also used for ^{129}I, based on man's expected intake of stable iodine, and restricted by the predicted ratio of ^{129}I to stable iodine in groundwater at the WRA (Section 2.5.3). The maximum ^{14}C dose is also limited by the predicted specific activity of ^{14}C in groundwater at the WRA. For radon, the only internal exposure pathway of significance is inhalation (Section 2.5.5); internal doses can therefore be calculated from the predicted radon air concentrations (Section 7.3) without considering food-chain transfer. Similarly, the noble gases argon and krypton cause no significant internal doses because they are not taken up and deposited in the body and so do not have to be traced through the food chain.

Radioactive daughters with half-lives less than 1 d are accounted for indirectly in the internal DCFs of their precursors. Similarly, these nuclides are not considered explicitly in the external exposure pathways; rather they are accounted for indirectly by adding their DCFs to those of their precursors.

CALDOS assumes that food-chain transfer and accumulation are independent of the contaminant concentration in plants and animals. In reality, very high concentrations might cause radiological or chemical toxicity effects to plants that would interrupt the normal transfer process (Chapter 13). By neglecting such effects, we would overestimate doses to humans and other biota whenever concentrations in the food chain become high. However, the very low dose criteria for humans by the AECB (1987) preclude such food-chain disruptions.

8.3 MATHEMATICAL FORMULATION OF CALDOS

Doses from each exposure pathway considered in CALDOS are calculated using separate equations, each of which has the general form of Equation (8.1). The various pathways and equations are discussed in turn below.

CALDOS is set up to work with radioactive decay constants, λ^i, expressed in units of d^{-1} rather than a^{-1}. Values of λ^i in d^{-1} were obtained by dividing the values in Table 1-1 by 365. Similarly, when working with radionuclides, CALDOS is formulated to operate in becquerels (Bq) per unit volume, mass or time, rather than in moles (mol). Accordingly, before CALDOS is implemented, the deposition rates and the predicted water, sediment, soil and air concentrations from the other biosphere submodels and the geosphere model are converted from moles to becquerels. This is achieved by multiplying the molar concentrations for radionuclide i by a factor $N_A \cdot \lambda^i$ ($Bq \cdot mol^{-1}$), where $N_A = 6.02 \times 10^{23}$ $atoms \cdot mol^{-1}$, Avogadro's number, and λ^i is the decay constant in units of s^{-1}.

In the equations of CALDOS, soil concentrations of nuclides are designated by C^i_s, which relates to the soil submodel (Table 6-6). However, soil concentrations may also be based on sediment concentrations, C^i_{se}, as calculated in Equation (5.15). The probability of sediment use as soil is presented in Section 6.5.5.4. Several equations in CALDOS involve domestic water and the nuclide concentration of it is designated by C_w^i. Domestic water may be derived from the bedrock well, C_{ww}^i (Equation (4.18)), or the lake, C_l^i (Equation (5.6)). The choice of the water source is explained in Section 9.1.2.

Values for the various parameters appearing in the mathematical expressions of CALDOS are discussed in Section 8.5 and in much greater detail by Zach and Sheppard (1992).

8.3.1 Internal Ingestion and Inhalation Exposure Pathways

8.3.1.1 Soil/Plant/Man

Plants grown on contaminated soil may themselves become contaminated by drawing up through their roots nuclides dissolved in soil pore water. Man's intake rate of nuclide i through ingestion of terrestrial plant food contaminated by root uptake is given by

$$(E_j^i)_{RP} = \left[C_s^i \cdot Bv^i \cdot \exp(-\lambda^i \cdot th)\right] \cdot U_j \qquad (8.2)$$

where $(E_j^i)_{RP}$ is man's intake of nuclide i via food type j = TE PLANT ($Bq \cdot a^{-1}$),

C_s^i is the annual average soil concentration of nuclide i ($Bq \cdot kg^{-1}$ dry soil),

Bv^i is the plant/soil concentration ratio for nuclide i (($Bq \cdot kg^{-1}$ wet biomass)/($Bq \cdot kg^{-1}$ dry soil)),

λ^i is the radioactive decay constant of nuclide i (d^{-1}),

th is the holdup time for TE PLANT (d), and

U_j is man's ingestion rate of food type j (kg wet biomass$\cdot a^{-1}$).

The term $C_s^i \cdot Bv^i$ in Equation (8.2) describes the concentration in the growing plant before it is harvested; the exponential term accounts for activity lost through radioactive decay during the time, th, defined by the times of harvest and consumption. For daughter radionuclides with half-lives between 1 d and 20 a, decay of the precursor also contributes to the activity in the plant through ingrowth. As noted in Section 2.5.4, this contribution is estimated assuming secular equilibrium between the daughter and the precursor. Accordingly, man's intake of nuclide i through ingestion of terrestrial plant foods contaminated by root uptake of nuclide i - 1 is

$$(E_j^i)_{IRP} = (E_j^{i-1})_{RP} = \left[C_s^{i-1} \cdot Bv^{i-1} \cdot \exp(-\lambda^{i-1} \cdot th)\right] \cdot U_j \qquad (8.3)$$

where $(E_j^i)_{IRP}$ is man's intake of nuclide i due to ingrowth following root uptake of nuclide i - 1 ($Bq \cdot a^{-1}$).

In practice, nuclide i - 1 is taken to be the immediate precursor to nuclide i with a half-life greater than 1 d.

Food ingestion rates, U_j, are calculated in an integrated way from the total energy requirements of man as shown in Section 8.3.4.1.

The dose to man from ingestion of terrestrial plant foods contaminated by root uptake is then given by

$$(D_j^i)_{RP} = \left[(E_j^i)_{RP} + (E_j^i)_{IRP}\right] \cdot DFe^i \tag{8.4}$$

where $(D_j^i)_{RP}$ is man's ingestion dose from nuclide i in food type j = TE PLANT ($Sv \cdot a^{-1}$), and

DFe^i is man's ingestion dose conversion factor for nuclide i ($Sv \cdot Bq^{-1}$).

The term $(E_j^i)_{IRP}$ is included in Equation (8.4) only when nuclide i is a daughter with a half-life between 1 d and 20 a. The concentrations, C_s^i, appearing in Equations (8.2) and (8.3) are set equal to the soil concentrations of the garden (Section 6.3.7). The ingestion rates, U_j, are calculated from the diet and total energy requirements of members of the critical group (Section 8.3.4.1).

In using the concentration ratio concept, we assume that nuclide concentrations in the growing plant are directly proportional to soil nuclide concentrations. No account is taken of the nuclide inventory, as opposed to concentration, in the soil that is actually available for uptake. For nuclides with a high Bv^i value, the amount transferred in the model could thus exceed the amount present in the soil. To prevent this from happening, we define a maximum nuclide plant concentration, CM_p^i ($Bq \cdot kg^{-1}$ wet biomass), equal to the concentration that would occur if the entire inventory of a nuclide were taken up by the plants (Zach and Sheppard 1992):

$$CM_p^i = C_s^i \cdot Z_r \cdot \rho_s / Y_j \tag{8.5}$$

where Z_r (m) is the depth of the soil root zone (Section 6.1), ρ_s (kg dry soil$\cdot m^{-3}$ soil) is the soil bulk density (Section 6.5.1.3), and Y_j (kg wet biomass$\cdot m^{-2}$ soil) is the appropriate plant yield (Section 8.5.8.1). Whenever the plant concentration calculated using the concentration ratio approach exceeds CM_p^i, the term $C_s^i \cdot Bv^i$ in Equations (8.2) is replaced by CM_p^i. The same is true for Equation (8.3), but with nuclide i - 1.

8.3.1.2 Soil/Plant/Animal/Man

Animals may become contaminated by eating plants that have taken up nuclides from the soil. Man's intake of nuclide i through ingestion of terrestrial animal products contaminated through root uptake is given by

$$(E_j^i)_{RPA} = \left[C_s^i \cdot Bv^i \cdot F_j^i \cdot Qf_j \cdot \exp(-\lambda^i \cdot thf_j)\right] \cdot U_j \tag{8.6}$$

where $(E^i_j)_{RPA}$ is man's intake of nuclide i via food types j = TE MEAT, TE MILK or TE BIRD contaminated by root uptake ($Bq \cdot a^{-1}$),

F^i_j is the terrestrial animal transfer coefficient for nuclide i and food type j ($d \cdot L^{-1}$ or $d \cdot kg^{-1}$ wet biomass),

Qf_j is the rate of feed or forage ingestion for food type j (kg wet biomass$\cdot d^{-1}$), and

thf_j is the terrestrial animal feed holdup time (d).

The product $C^i_s \cdot Bv^i$ appearing in Equation (8.6) is the concentration of nuclide i in the feed or forage consumed by the animals, and is evaluated using the soil concentration predicted for the forage field (Section 6.3.7). This quantity is not allowed to exceed the maximum plant concentration, CM_p (Section 8.3.1.1). The transfer factor F^i_j is the portion of a nuclide ingested daily that is secreted per litre of milk (TE MILK), or incorporated into one kilogram of meat or eggs (TE MEAT and TE BIRD). The quantity $C^i_s \cdot Bv^i \cdot F^i_j \cdot Qf_j$ represents the nuclide concentration in the animal product before the animal is slaughtered (TE MEAT and TE BIRD), or before the dairy cow is milked (TE MILK). The holdup time, thf_j, allows for losses in the forage crop between the time the crop is harvested and the time it is consumed by the animals, and for losses in the animal products between the time the animal is slaughtered or milked and the time the product is consumed by man.

The dose to man, $(D^i_j)_{RPA}$ ($Sv \cdot a^{-1}$), from ingestion of animal products contaminated through the root-uptake pathway is given by

$$(D^i_j)_{RPA} = \left[(E^i_j)_{RPA} + (E^{i-1}_j)_{RPA}\right] \cdot DFe^i \quad . \tag{8.7}$$

The term $(E^{i-1}_j)_{RPA}$ accounts for ingrowth, and is included only when nuclide i is a daughter with a half-life between 1 d and 20 a, as discussed in Section 8.3.1.1.

8.3.1.3 Air/Plant/Man

Plants may also become contaminated through their leaves and other exposed surfaces following deposition of nuclides from contaminated air or interception of contaminated irrigation water. Deposited radionuclides may be absorbed or remain attached externally. Man's intake rate of nuclide i through ingestion of terrestrial plant foods contaminated by deposition to leaves is given by

$$(E^i_j)_{LP} = \left\{ D^i_b \cdot (r_j/Y_j) \cdot \left[\exp(-\lambda^i \cdot th)\right] \cdot \left[1 - \exp(-\lambda^i_E \cdot te_j)\right] / \lambda^i_E \right\} \cdot U_j \tag{8.8}$$

where $(E^i_j)_{LP}$ is man's intake of nuclide i via food type j = TE PLANT ($Bq \cdot a^{-1}$),

D_b^i	is the rate of deposition of nuclide i to vegetation ($Bq \cdot m^{-2}$ soil$\cdot d^{-1}$),
r_j	is the plant interception fraction for food type j = TE PLANT (unitless),
λ_E^i	is the effective removal constant of nuclide i from vegetation (d^{-1}),
te_j	is the time of above-ground exposure for terrestrial food type j = TE PLANT during the growing season (d), and
Y_j	is the yield of food type j = TE PLANT (kg wet biomass$\cdot m^{-2}$ soil).

The term $D_b^i \cdot (r_j/Y_j)$ in Equation (8.8) provides the amount of nuclide i intercepted by the leaves per unit time and unit mass of standing vegetation. The deposition rate, D_b^i, may be based on deposition from the atmosphere (Section 7.3.8), in which case it is given by Equations (7.40) to (7.43). If the field is irrigated (Section 6.3.7.2), nuclides may be deposited with the irrigation water, in which case an additional contribution to D_b^i is represented by $(D_b^i)_I$ and calculated from the amount of irrigation water applied, I_w (m^3 water$\cdot d^{-1}$) (Equation (6.1)), and the nuclide concentration in the water, C_w^i (mol$\cdot m^{-3}$ water),

$$(D_b^i)_I = C_w^i \cdot I_w \quad . \tag{8.9}$$

Because the atmospheric deposition of ^{14}C is accounted for in the parameters of its root-uptake pathway (Section 8.5.1.1), Equation (8.8) is applied to ^{14}C only when it deals with the interception of irrigation water.

The term $\exp(-\lambda^i \cdot th)$ in Equation (8.8) accounts for activity lost from the plant through radioactive decay in the interval between the time the plant is harvested and the time it is consumed. The term $[1 - \exp(-\lambda_E^i \cdot te_j)]/\lambda_E^i$ describes the loss of activity from radiological decay and environmental factors such as wind and water action while the plant is in the field. The effective removal constant, λ_E^i, is given by

$$\lambda_E^i = \lambda^i + (\ln 2)/tp \tag{8.10}$$

where tp is the plant environmental halftime (d). Nuclides deposited to vegetation are assumed not to be removed by food processing and preparation.

The dose to man, $(D_j^i)_{LP}$ ($Sv \cdot a^{-1}$), from ingestion of TE PLANT contaminated by atmospheric and irrigation water depositions is given by

$$(D_j^i)_{LP} = \left[(E_j^i)_{LP} + (E_j^{i-1})_{LP}\right] \cdot DFe^i \quad . \tag{8.11}$$

The term $(E_j^{i-1})_{LP}$ accounts for ingrowth, and is included only when nuclide i is a daughter with a half-life between 1 d and 20 a (Section 8.3.1.1).

8.3.1.4 Air/Plant/Animal/Man

Animals may become contaminated by eating plants that have taken up nuclides through their leaves. Man's intake of nuclide i through ingestion of terrestrial animal products contaminated through leaf deposition, $(E^i_j)_{LPA}$ ($Bq \cdot a^{-1}$), is given by

$$(E^i_j)_{LPA} = \left\{ D^i_b \cdot (r_j/Y_j) \cdot F^i_j \cdot Qf_j \cdot \exp\left[-\lambda^i \cdot thf_j\right] \cdot \left[1 - \exp(-\lambda^i_E \cdot te_j)\right]/\lambda^i_E \right\} \cdot U_j \quad . \tag{8.12}$$

The parameters in this equation and the physical interpretation of the equation itself have been discussed above. The equation is used to calculate intakes of nuclide i in food types j = TE MEAT, TE MILK and TE BIRD for vegetation contaminated both by atmospheric deposition and irrigation water.

The dose to man, $(D^i_j)_{LPA}$ ($Sv \cdot a^{-1}$), from ingestion of animal products contaminated through leaf deposition is given by

$$(D^i_j)_{LPA} = \left[(E^i_j)_{LPA} + (E^{i-1}_j)_{LPA}\right] \cdot DFe^i \quad . \tag{8.13}$$

The term $(E^{i-1}_j)_{LPA}$ accounts for ingrowth, and is included only when nuclide i is a daughter with a half-life between 1 d and 20 a (Section 8.3.1.1). As explained in Section 8.3.1.3, Equation (8.12) is applied to ^{14}C only when it deals with the interception of irrigation water.

8.3.1.5 Water/Animal/Man

Animals may become contaminated by drinking water that contains nuclides. Man's intake of nuclide i through ingestion of terrestrial animal products contaminated through water, $(E^i_j)_{WA}$ ($Bq \cdot a^{-1}$), is given by

$$(E^i_j)_{WA} = \left[C^i_w \cdot F^i_j \cdot Qdw_j \cdot \exp(-\lambda^i \cdot thw_j)\right] \cdot U_j \tag{8.14}$$

where C^i_w is the annual average concentration of nuclide i in animal's drinking water ($Bq \cdot m^{-3}$ water),

Qdw_j is the drinking water ingestion rate for food types j = TE MEAT, TE MILK and TE BIRD (m^3 water$\cdot d^{-1}$), and

thw_j is the terrestrial animal drinking water holdup time (d).

In applying Equation (8.14), C^i_w is set equal to the nuclide concentration in well water or lake water, whichever has been chosen as the domestic water source in a given simulation (Section 9.1.2). The same transfer factor, F^i_j, is assumed to govern the transfer of nuclides from water to animals as from feed or forage to animals (Equation (8.6)).

The dose to man, $(D^i_j)_{WA}$ ($Sv \cdot a^{-1}$), from ingestion of animal products contaminated through drinking water is given by

$$(D^i_j)_{WA} = \left[(E^i_j)_{WA} + (E^{i-1}_j)_{WA}\right] \cdot DFe^i \quad . \tag{8.15}$$

The term $(E^{i-1}_j)_{WA}$ accounts for ingrowth, and is included only when nuclide i is a daughter with a half-life between 1 d and 20 a (Section 8.3.1.1).

8.3.1.6 Soil/Animal/Man

Grazing animals may ingest considerable amounts of soil, either inadvertently with feed or forage, during inhalation, or deliberately in response to dietary mineral deficiencies. If the soil is contaminated, this pathway may contribute to the nuclide load in the animals. Man's intake of nuclide i through ingestion of terrestrial animal products contaminated through soil ingestion is given by

$$(E^i_j)_{SA} = \left[C^i_s \cdot F^i_j \cdot Qs_j \cdot \exp(-\lambda^i \cdot ths_j)\right] \cdot U_j \tag{8.16}$$

where $(E^i_j)_{SA}$ is man's intake of nuclide i via food types j = TE MEAT, TE MILK and TE BIRD ($Bq \cdot a^{-1}$),

Qs_j is the rate of soil ingestion by food type j (kg dry $soil \cdot d^{-1}$), and

ths_j is the terrestrial animal soil holdup time (d).

In applying Equation (8.16), C^i_s is set equal to the soil concentration in the forage field (Section 6.3.7). Transfer from soil ingestion is assumed to be governed by the same factor, F^i_j, as transfer from forage or water (Equations (8.6) and (8.14)).

The dose to man, $(D^i_j)_{SA}$ ($Sv \cdot a^{-1}$), from ingestion of animal products contaminated through soil ingestion is given by

$$(D^i_j)_{SA} = \left[(E^i_j)_{SA} + (E^{i-1}_j)_{SA}\right] \cdot DFe^i \quad . \tag{8.17}$$

The term $(E^{i-1}_j)_{SA}$ accounts for ingrowth, and is included only when nuclide i is a daughter with a half-life between 1 d and 20 a (Section 8.3.1.1).

8.3.1.7 Water/Fish/Man

Fish inhabiting the discharge lake may become contaminated through ingestion of food and sediment, and through the osmotic exchange of fluids. Man's intake of nuclide i through fish ingestion is given by

$$(E^i_j)_{WF} = \left[C^i_l \cdot B^i_j \cdot \exp(-\lambda^i \cdot thp)\right] \cdot U_j \tag{8.18}$$

where $(E^i_j)_{WF}$ is man's intake of nuclide i via food type j = FW FISH ($Bq \cdot a^{-1}$),

C^i_l is the concentration of nuclide i in lake water ($Bq \cdot m^{-3}$ water),

B^i_j is the aquatic concentration ratio for nuclide i and food type j = FW FISH (m^3 water$\cdot kg^{-1}$ wet biomass), and

thp is the holdup time for FW FISH (d).

The dose to man, $(D^i_j)_{WF}$ ($Sv \cdot a^{-1}$), from ingestion of contaminated fish is given by

$$(D^i_j)_{WF} = \left[(E^i_j)_{WF} + (E^{i-1}_j)_{WF}\right] \cdot DFe^i \quad . \tag{8.19}$$

The term $(E^{i-1}_j)_{WF}$ accounts for ingrowth, and is included only when nuclide i is a daughter with a half-life between 1 d and 20 a (Section 8.3.1.1).

8.3.1.8 Water/Man

Man's intake of nuclide i from the ingestion of contaminated drinking water is given by

$$(E^i)_w = C^i_w \cdot \exp(-\lambda^i \cdot thdw) \cdot Udw \tag{8.20}$$

where $(E^i)_w$ is man's intake of nuclide i via drinking water ($Bq \cdot a^{-1}$),

Udw is man's ingestion rate of drinking water (m^3 water$\cdot a^{-1}$), and

thdw is the holdup time for man's drinking water (d).

In applying Equation (8.20), C^i_w is set equal to the nuclide concentration in well water or lake water, whichever has been chosen as the water source in BIOTRAC (Section 9.1.2). We assume conservatively that nuclides are not removed from the water by sedimentation, community treatment plants or home filtration systems prior to ingestion. The water ingestion rate, Udw, is calculated from the diet and total energy requirements of man (Section 8.3.4.2).

Man's dose from water ingestion, $(D^i)_w$ ($Sv \cdot a^{-1}$), is given by

$$(D^i)_w = (E^i)_w \cdot DFe^i \quad . \tag{8.21}$$

There is no need to consider ingrowth in this pathway. Ingrowth is accounted for explicitly in calculating C^i_w, and holdup times are too short to allow daughters to build up significantly because water may be taken directly out of the lake and ingested immediately.

8.3.1.9 Soil/Man

Humans may ingest soil that has become attached to their hands or to crops. Man's intake of nuclide i from the ingestion of contaminated soil, $(E^i)_S$ ($Bq \cdot a^{-1}$), is given by

$$(E^i)_S = C_s^i \cdot Us \quad . \tag{8.22}$$

Here, Us is man's soil ingestion rate (kg dry soil$\cdot a^{-1}$), which includes a component for hand-to-mouth transfer, and another for food ingestion. The latter is linked to the ingestion rate, U_j, of TE PLANT (Section 8.3.4.1), the main food type involved in soil ingestion. The soil ingestion rate is given by

$$Us = Hs + Ps \cdot U_j \tag{8.23}$$

where Hs is the soil ingestion rate from hands (kg dry soil$\cdot a^{-1}$), and

Ps is the mass of soil adhering to a unit mass of TE PLANT (kg dry soil$\cdot kg^{-1}$ wet biomass).

In applying Equation (8.22), C_s^i is set equal to the soil concentration in the garden (Section 6.3.7). We assume that nuclides associated with soil are absorbed via the gastrointestinal tract in the same way as from food and water, and use the DCF for man's food and drinking water ingestion. We have not included a holdup time in Equation (8.22), although a delay in soil ingestion might occur in the case of soil contamination of plants.

The dose to man from soil ingestion, $(D^i)_S$ ($Sv \cdot a^{-1}$), is given by

$$(D^i)_S = (E^i)_S \cdot DFe^i \quad . \tag{8.24}$$

There is no need to consider ingrowth for this pathway for reasons similar to those discussed for the drinking-water pathway (Section 8.3.1.8).

8.3.1.10 Air/Man

Man's intake of nuclide i from inhaling contaminated air, $(E^i)_A$ ($Bq \cdot a^{-1}$), is given by

$$(E^i)_A = \left[(C_a^i)_O \cdot Og + (C_a^i)_I \cdot Ob\right] \cdot I_A \tag{8.25}$$

where $(C_a^i)_O$ and $(C_a^i)_I$ are the annual average outdoor and indoor air concentrations of nuclide i ($Bq \cdot m^{-3}$ air),

Og and Ob are man's ground and building occupancy factors respectively (unitless), and

I_A is man's inhalation rate (m^3 air$\cdot a^{-1}$).

Man's inhalation dose, $(D^i)_A$ $(Sv \cdot a^{-1})$, is then given by

$$(D^i)_A = (E^i)_A \cdot DFi^i \qquad (8.26)$$

where DFi^i is man's inhalation dose conversion factor for nuclide i $(Sv \cdot Bq^{-1})$.

Inhalation doses are calculated for all nuclides except ^{39}Ar, ^{81}Kr and ^{85}Kr for which inhalation is an insignificant pathway (Section 2.5.5). The specific-activity model for tritium considers inhalation implicitly (Section 8.3.1.13). The sum Og + Ob, which is the air occupancy factor, must have a value of 1.0. The inhalation rate, I_A, is calculated from the diet and total energy requirements of man (Section 8.3.4.3).

8.3.1.11 Model for ^{129}I

A special model, discussed in detail by Zach and Sheppard (1992), has been developed to calculate internal ^{129}I doses to man to reflect the unique behaviour of iodine in the body and in the biosphere. Internal ^{129}I doses are dominated by the thyroid dose, and the iodine content of the thyroid gland is regulated metabolically (Section 2.5.3). Because stable iodine (^{127}I) is ubiquitous in the environment, members of the critical group will always ingest stable iodine along with any ^{129}I that might reach the biosphere from the vault.

Let the ratio I_R of ^{129}I to total (^{129}I plus ^{127}I) iodine ingested be

$$I_R = \frac{(E^I)_T \cdot gb}{(E^I)_T \cdot gb + E^{Is}} \qquad (8.27)$$

where $(E^I)_T$ is man's total annual intake of ^{129}I $(Bq \cdot a^{-1})$,

E^{Is} is man's total intake of stable iodine (kg $^{127}I \cdot a^{-1}$), and

gb is the mass/activity conversion factor for ^{129}I (kg iodine$\cdot Bq^{-1}$).

We adopt a limited specific-activity model and assume that I_R also describes the ratio of ^{129}I to total iodine in the thyroid. The total activity of ^{129}I per unit mass of the thyroid, A_I (Bq$\cdot kg^{-1}$ thyroid), is then

$$A_I = I_R \cdot \frac{Thi/gb}{Thm} \qquad (8.28)$$

where Thi is the iodine content of man's thyroid (kg iodine), and

Thm is the mass of man's thyroid (kg thyroid).

Man's total internal dose from ^{129}I, D^I $(Sv \cdot a^{-1})$, is then given by

$$D^I = A_I \cdot DF^I \qquad (8.29)$$

where DF^I is man's internal dose conversion factor for ^{129}I $((Sv \cdot a^{-1})/(Bq \cdot kg^{-1}$ thyroid$))$.

The total intake of ^{129}I, $(E^I)_T$ $(Bq \cdot a^{-1})$, is calculated by summing the intakes over all pathways using Equations (8.2), (8.6), (8.8), (8.12), (8.14), (8.16), (8.18), (8.20), (8.22) and (8.25):

$$(E^I)_T = (E^I_{TE\ PLANT})_{RP} + (E^I_{TE\ PLANT})_{LP} + (E^I_{FW\ FISH})_{WF} + (E^I)_W + (E^I)_S + (E^I)_A + \sum_j \sum_k (E^I_j)_K \tag{8.30}$$

where the summation in the last term extends over the food types j = TE MEAT, TE MILK and TE BIRD, and the pathways k = RPA, LPA, WA and SA (Appendix B). D^I (Equation (8.29)) therefore represents the total internal dose, including inhalation and all the ingestion pathways.

The stable iodine present in the biosphere arises from a number of sources, including groundwater transport, deposition from the atmosphere, and the weathering of surficial material. We do not have a good understanding of the exact contribution that each of these sources makes to local stable iodine concentrations, or to the intake of stable iodine by humans, although the latter is well known (Zach and Sheppard 1992). In applying Equation (8.27), we set E^{IS} equal to a conservatively low value (Section 8.5.9.1) and do not specify its source. If stable iodine concentrations in the biosphere are high, our value for E^{IS} would underestimate the ^{127}I intake rate, and doses from ^{129}I would be overpredicted. It is difficult to correct for this in general, given our present understanding of the sources of environmental stable iodine.

We can, however, say that the critical group's intake of stable iodine should be at least as great as that implied by the stable iodine concentration in groundwater discharging to their locality. Put another way, the ratio, I_R, cannot exceed the ratio of ^{129}I to total iodine in groundwater carrying nuclides from the vault. This imposes a second, or groundwater, limit on the internal ^{129}I dose, which is implemented in the following way. The ratio I^G_R of ^{129}I to total iodine in groundwater is calculated from

$$I^G_R = \frac{C^I_{gw} \cdot gb}{C^I_{gw} \cdot gb + C^{sI}_{gw}} \tag{8.31}$$

where C^I_{gw} is the ^{129}I concentration in groundwater $(Bq \cdot m^{-3}$ water$)$, and C^{sI}_{gw} is the stable iodine concentration in groundwater $(kg\ ^{127}I \cdot m^{-3}$ water$)$. If the source of domestic water is the lake, C^I_{gw} is set equal to the concentration in groundwater discharging to the lake (Equation (4.3)) at the most contaminated discharge zone. If the water source is the well, C^I_{gw} is set equal to the higher of the concentrations in the well water (Equation (4.18)) or water discharging to the lake using the most contaminated discharge. C^{sI}_{gw} is treated as a distributed parameter, with values based on observed stable iodine concentrations in near-surface groundwater at the WRA (Section 8.5.9.5).

The value of I_R^G is then used in Equations (8.28) and Equation (8.29) to calculate an upper groundwater limit to the internal dose from ^{129}I, $(D^I)_U$ ($Sv \cdot a^{-1}$). If D^I as calculated from Equation (8.29) with I_R exceeds $(D^I)_U$, the internal dose from ^{129}I is set equal to $(D^I)_U$.

In writing Equation (8.31), we have assumed that man's entire stable iodine intake is from groundwater. In principle, it would be possible to reduce the maximum ^{129}I dose further by including other sources of stable iodine uncontaminated with ^{129}I from the vault. These large sources are not known well enough to formulate such a model at this time (Zach and Sheppard 1992). An upper limit based on stable iodine derived from the geosphere alone provides a conservative estimate of the internal ^{129}I dose.

8.3.1.12 Limit to Man's ^{14}C Internal Dose

Groundwater discharging into the biosphere contains appreciable amounts of stable carbon. The ratio of radioactive ^{14}C to stable carbon will be higher in the groundwater than elsewhere in the biosphere, which contains additional large pools of stable carbon. The presence of stable carbon in the groundwater therefore imposes an upper limit on the internal dose to humans in the same way that the existence of stable iodine limits the ^{129}I dose (Section 2.5.2 and 8.3.1.11) even though carbon does not accumulate in a specific organ such as the thyroid gland. Doses calculated using our transport model for ^{14}C can exceed this groundwater limit for some combinations of extreme parameter values because the model does not allow for the large amounts of stable carbon in groundwater that would accompany and dilute ^{14}C from the vault. In each simulation of BIOTRAC, we compare the ^{14}C dose predicted by the transport model with the dose based on the groundwater limit and use the smaller in subsequent calculations.

We establish the groundwater limit for the ^{14}C dose similarly to how we did for ^{129}I, assuming that the ^{14}C specific activity in the human body equals the specific activity in the groundwater. The ratio C_R^G of ^{14}C to total carbon in groundwater is calculated from

$$C_R^G = \frac{C_{gw}^c \cdot gc}{C_{gw}^c \cdot gc + C_{gw}^{sc}} \tag{8.32}$$

where C_{gw}^c is the ^{14}C concentration in groundwater ($Bq \cdot m^{-3}$ water),

gc is the mass/activity conversion factor for ^{14}C (kg carbon$\cdot Bq^{-1}$), and

C_{gw}^{sc} is the concentration of stable carbon in groundwater (kg $^{12}C \cdot m^{-3}$ water).

If the lake is the source of domestic water, C_{gw}^c is set equal to the concentration in groundwater discharging to the lake (Equation (4.3)) at the most contaminated discharge zone. If the water source is the well (Equation (4.18)), C_{gw}^c is set equal to the higher of the concentrations in well water or water discharging to the lake using the most contaminated discharge. C_{gw}^{sc} is treated as a distributed parameter, with values based on observed stable carbon concentrations in near-surface groundwater at the

WRA (Section 8.5.10.4). The upper groundwater limit to the dose is then calculated from

$$(D^{c})_{U} = DF^{c\prime} \cdot C_{R}^{G} \cdot Bc/(gc \cdot Bs) \tag{8.33}$$

where $(D^{c})_{U}$ is man's maximum total internal dose from ^{14}C ($Sv \cdot a^{-1}$),

$DF^{c\prime}$ is man's internal dose conversion factor for ^{14}C based on the specific-activity model (($Sv \cdot a^{-1}$)/($Bq \cdot kg^{-1}$ soft tissue)),

Bc is the carbon content of soft tissue in man's body (kg carbon), and

Bs is the mass of soft tissue in man's body (kg soft tissue).

In writing Equation (8.33), we assume that the internal ^{14}C dose results from the irradiation of soft tissue. This is conservative because bone has a lower carbon content than soft tissue (ICRP 1975) and so inclusion of bone would reduce dose estimates. Carbon-14 decays through the emission of β radiation, which is assumed to be completely absorbed by the soft tissue. The dose conversion factor, $DF^{c\prime}$, differs from that used in the transport model (Section 8.5.2.1) because it is calculated from the amount of ^{14}C in soft tissue rather than from the intake rate of ^{14}C.

If the total internal ^{14}C dose in a given BIOTRAC simulation exceeds $(D^{c})_{U}$, the internal dose from ^{14}C is set equal to $(D^{c})_{U}$. This overall approach is conservative because members of the critical group would derive almost all of their stable carbon from sources other than groundwater, i.e., photosynthetically fixed atmospheric carbon (CO_2). Inclusion of these sources of stable carbon would further reduce internal dose predictions.

8.3.1.13 Model for Tritium

Internal doses from tritium are calculated on the assumption that the specific activity of tritium in humans is the same as the specific activity in lake or well water. This assumption is appropriate as explained in Section 2.5.1. Accordingly, man's total internal dose from tritium, D^{H3} ($Sv \cdot a^{-1}$), is given by

$$D^{H3} = (C_{w}^{H3}/C_{w}^{H}) \cdot MC^{H} \cdot DF^{H3} \tag{8.34}$$

where C_{w}^{H3} is the annual average tritium concentration in domestic water ($Bq \cdot m^{-3}$ water),

C_{w}^{H} is the hydrogen concentration in water (g hydrogen$\cdot m^{-3}$ water),

MC^{H} is the average hydrogen concentration in man's body (g hydrogen$\cdot kg^{-1}$ soft tissue), and

DF^{H3} is man's internal dose conversion factor for tritium (($Sv \cdot a^{-1}$)/($Bq \cdot kg^{-1}$ soft tissue)).

If the lake is the source of domestic water, C_w^{H3} is set equal to the tritium concentration in the lake water; if the well is the water source, C_w^{H3} is set equal to the well-water concentration (Section 9.1.2). Our specific-activity model accounts for all mechanisms of internal tritium contamination, including absorption through the skin (Section 2.5.1).

8.3.2 External Exposure Pathways

8.3.2.1 Immersion in Air

The external dose to man from immersion in air contaminated by nuclide i, $(D^i)_{AI}$ $(Sv \cdot a^{-1})$, is given by

$$(D^i)_{AI} = \left[(C_a^i)_O \cdot Og + (C_a^i)_I \cdot Ob\right] \cdot DFa^i \qquad (8.35)$$

where DFa^i is man's air immersion dose conversion factor for nuclide i $((Sv \cdot a^{-1})/(Bq \cdot m^{-3}$ air)).

Although in principle this equation applies to all nuclides, DCFs for ^3H, ^{63}Ni, ^{107}Pd, ^{187}Re and ^{93}Zr are very small because of the type and energy level of radiation emitted during decay, and so air immersion doses are not included (Holford 1989, Zach and Sheppard 1992).

8.3.2.2 Immersion in Water

The external dose to man from swimming and bathing in contaminated water is given by

$$(D^i)_{WI} = C_w^i \cdot Oe \cdot DFh^i \qquad (8.36)$$

where $(D^i)_{WI}$ is man's water immersion dose from nuclide i $(Sv \cdot a^{-1})$,

Oe is man's water occupancy factor (unitless), and

DFh^i is man's water immersion dose conversion factor for nuclide i $((Sv \cdot a^{-1})/(Bq \cdot m^{-3}$ water)).

Since most people spend far more time bathing than swimming, C_w^i is set equal to the nuclide concentration in the lake or well, whichever is chosen as the domestic water source (Section 9.1.2). Although Equation (8.36) applies to all the nuclides, water immersion doses for ^{39}Ar, ^3H, ^{81}Kr, ^{85}Kr, ^{63}Ni, ^{107}Pd, ^{187}Re, ^{222}Rn and ^{93}Zr are all very small (Holford 1989) and are not included.

8.3.2.3 Ground Exposure

The external dose to man from ground contaminated by nuclide i, $(D^i)_G$ $(Sv \cdot a^{-1})$, is given by

$$(D^i)_G = C_s^i \cdot dws \cdot Og \cdot DFg^i \qquad (8.37)$$

where dws is a dry/wet soil conversion factor (kg dry soil$\cdot$kg^{-1} wet soil), and

DFg^i is man's ground exposure dose conversion factor for nuclide i ((Sv$\cdot$a^{-1})/(Bq$\cdot$kg^{-1} wet soil)).

In applying Equation (8.37), C_s^i is set equal to the nuclide concentration in the garden soil. The conversion factor dws is required because the soil model predicts concentrations in dry soil, whereas the DCFs are traditionally given in terms of wet soil weight. We have conservatively excluded the shielding effects of snow cover, which would be less contaminated than the soil itself. Ground exposure doses for ^{39}Ar, ^{41}Ca, ^{3}H, ^{81}Kr, ^{85}Kr, ^{63}Ni, ^{107}Pd, ^{187}Re, ^{222}Rn and ^{93}Zr are very small (Holford 1989) and are not included. Because garden soils may consist of sediment (Section 6.3.7.3), we account indirectly for external doses from contaminated shorelines or beaches.

8.3.2.4 Exposure to Building Materials

Members of the critical group may use local resources to build their homes. The external dose to man from building materials is given by

$$(D_j^i)_{BM} = C_j^i \cdot Ob \cdot DFb^i \tag{8.38}$$

where $(D_j^i)_{BM}$ is man's dose from exposure to building material j contaminated by nuclide i (Sv$\cdot$a^{-1}),

C_j^i is the annual average concentration of nuclide i in building material j (Bq$\cdot$kg^{-1} dry weight), and

DFb^i is man's building material exposure dose conversion factor for nuclide i ((Sv$\cdot$a^{-1})/(Bq$\cdot$kg^{-1} dry material)).

The building materials may be either organic (wood) or inorganic (sand, gravel, etc.), derived from contaminated soils. Both types would be used in most homes, but the proportion and distribution throughout the home is unknown at our generic site. Accordingly, we calculate doses for both materials in each simulation, and use the larger value in computing the total dose to man.

Nuclide concentrations in inorganic building materials, C_{ibm}^i (Bq$\cdot$kg^{-1} dry weight), are calculated from the soil concentrations

$$C_{ibm}^i = C_s^i \cdot sbc \cdot \exp(-\lambda^i \cdot thb_{ibm}) \tag{8.39}$$

where C_s^i is the concentration of nuclide i in soil (Bq$\cdot$kg^{-1} dry soil),

sbc is an inorganic building material/soil conversion factor (unitless), and

thb_{ibm} is the holdup time for inorganic building materials (d).

In applying Equation (8.39), C_s^i is set equal to the soil concentration in the most contaminated layer of the forage field (Section 6.3.7). Since the soil is contaminated from above or below, the most contaminated layer in practice is either the root zone or layer 4 at the bottom of the soil profile. The critical group is not likely to derive their building materials from the garden, a more valuable resource for growing food. The forage field is usually the next most highly contaminated field, and is used conservatively. The conversion factor, sbc, allows for the loss or concentration of nuclides during the processing of the raw resources into finished building materials.

Nuclide concentrations in wooden building materials are calculated taking into account root uptake, atmospheric deposition to leaves and, where appropriate, ingrowth. The concentration of nuclide i resulting from root uptake and atmospheric deposition, C_{wbm}^i ($Bq \cdot kg^{-1}$ dry biomass), is given by

$$\begin{aligned} C_{wbm}^i = {} & wdw \cdot \exp(-\lambda^i \cdot thb_{wbm}) \\ & \cdot \left\{ C_s^i \cdot Bv^i + D_b^i \cdot rb/Yb \cdot \left[1 - \exp(-\lambda_E^i \cdot teb) \right] / \lambda_E^i \right\} \end{aligned} \tag{8.40}$$

where wdw is a wet/dry wood conversion factor (kg wet biomass$\cdot kg^{-1}$ dry wood),

thb_{wbm} is the holdup time for wooden building materials (d),

rb is the plant interception fraction for wooden building materials (unitless),

Yb is the plant yield for wooden building materials (kg wet biomass$\cdot m^{-2}$ land), and

teb is the time of above-ground exposure for wooden building materials (d).

For daughters with half-lives between 1 d and 20 a, the concentration in the wood is given by

$$C_{wbm}^i + C_{wbm}^{i-1} \tag{8.41}$$

where the second term accounts for ingrowth. This is analogous to the situation for man's intake of nuclides discussed in Section 8.3.1.1.

The conversion factor wdw in Equation (8.40) is required to provide concentrations in dry wood to match the units of the dose conversion factor DFb^i (Equation (8.38)). The soil concentration C_s^i is set equal to the concentration in the woodlot (Section 6.3.7). For each nuclide, we assume that the same processes and the same Bv^i values govern the transfer from soil to wood as from soil to plant and forage crops. In Equation (8.40), the leaf pathway, characterized by the deposition flux D_b^i ($Bq \cdot m^{-2}$ soil$\cdot d^{-1}$), applies only to deposition from the atmosphere because we assume that the woodlot is not irrigated. Building material exposure doses for ^{39}Ar, ^{10}Be, ^{14}C, ^{41}Ca, ^{113m}Cd, ^{135}Cs, ^{3}H, ^{81}Kr, ^{85}Kr, ^{63}Ni, ^{107}Pd, ^{87}Rb, ^{187}Re, ^{222}Rn, ^{79}Se, ^{32}Si, ^{90}Sr, ^{99}Tc and ^{93}Zr are very small (Holford 1989) and are not included.

8.3.3 Total Dose to Man

The total dose to members of the critical group is found by summing the doses over all pathways, nuclides, and, where appropriate, food types (Zach and Sheppard 1992). The total dose for each pathway is given by the sum of the contributions from each nuclide. The total dose for each nuclide is found by summing the doses over all pathways. The total internal dose is found by considering the pathways discussed in Section 8.3.1, and the total external dose is found by considering the pathways in Section 8.3.2. Note that the internal doses for ^{129}I and tritium (Equations (8.29) and (8.34) respectively) are already expressed as total doses summed over all internal pathways. The total internal dose from ^{129}I, calculated by Equation (8.29), can be broken down into its contributions via the different pathways summed in Equation (8.30). However, this is not possible when the total internal dose is calculated by Equation (8.31). Similarly, the total internal dose from ^{14}C calculated by Equation (8.33) cannot be broken down into individual pathway constituents.

8.3.4 Man's Ingestion and Inhalation Rates

Man's food ingestion rates, U_j, drinking water ingestion rate, Udw, and inhalation rate, I_A, are calculated in an integrated way in CALDOS. A simplified version of the EWAM model (Zach and Barnard 1987) is used to infer the rates given man's total energy need, his diet and the nutritional content of the foods in his diet.

8.3.4.1 Man's Food Ingestion Rates, U_j (kg wet biomass·a^{-1})

Man's food ingestion rates of TE PLANT, TE MEAT, TE MILK, TE BIRD and FW FISH are given by

$$U_j = \frac{En \cdot Ycfs_j}{Cym_j \cdot Cec + Fym_j \cdot Fec + Pym_j \cdot Pec} \tag{8.42}$$

where En is man's total energy need (kJ·a^{-1}),

$Ycfs_j$ is the energy fraction for food type j (unitless),

Cym_j, Fym_j and Pym_j are the carbohydrate, fat and protein contents of food type j respectively (g carbohydrate, fat or protein·kg^{-1} wet biomass), and

Cec, Fec and Pec are the carbohydrate, fat and protein fuel values respectively (kJ·g^{-1} carbohydrate, fat or protein).

The numerator in Equation (8.42) defines the annual amount of energy derived by man from food type j. The denominator converts this value to an ingestion rate, taking into account the energy derived from carbohydrate, fat and protein in the food types. The Cym, Pym and Pym values for TE MILK are derived assuming the density of milk is 1.0 $kg \cdot L^{-1}$.

The $Ycfs_j$ values in Equation (8.42) are given by

$$Ycfs_j = Ycf_j / \sum_{j=1}^{n} Ycf_j \tag{8.43}$$

where Ycf_j is a weighting factor (unitless) describing the contribution of food type j to man's total energy need, En, and n is the number of food types. Because the Ycf_j distributions are sampled independently for each food type (Section 8.5.6.4), the sum of the Ycf_j values may not equal one in a given model simulation. Use of the normalized fractions $Ycfs_j$ ensures that the sum of the energies contributed by the various food types adds up to man's total energy need.

8.3.4.2 Man's Drinking Water Ingestion Rate, Udw (m^3 water$\cdot a^{-1}$)

Man's drinking water ingestion rate is given by

$$\begin{aligned} Udw = En \cdot ewc - \sum_{j=1}^{n} (U_j \cdot Ywc_j) \\ + \sum_{j=1}^{n} U_j \cdot \left[Cym_j \cdot Cmw + Fym_j \cdot Fmw + Pym_j \cdot Pmw \right] \end{aligned} \quad . \tag{8.44}$$

where ewc is man's water/energy conversion ratio (m^3 water$\cdot kJ^{-1}$),

Ywc_j is the water content of food type j (m^3 water$\cdot kg^{-1}$ wet biomass), and

Cmw, Fmw and Pmw are the carbohydrate, fat and protein metabolic water yields respectively (m^3 water$\cdot g^{-1}$ carbohydrate, fat or protein).

The first term on the right in Equation (8.44) defines man's annual water requirements, given his total energy need. The two remaining terms define man's indirect intake of water made up by water ingested as part of his food and metabolic water from the oxidation of ingested carbohydrate, fat and protein. The difference between the water need and the indirect water input defines the amount of drinking water, Udw, that man must ingest.

8.3.4.3 Man's Inhalation Rate, I_A (m^3 air$\cdot a^{-1}$)

Man's inhalation rate is given by

$$I_A = oac \cdot ov \cdot \sum_{j=1}^{n} U_j \cdot \left[Cym_j \cdot Co + Fym_j \cdot Fo + Pym_j \cdot Po \right] \tag{8.45}$$

where oac is man's air/oxygen conversion factor (unitless),

ov is man's oxygen utilization factor (unitless), and

Co, Fo and Po are the carbohydrate, fat and protein STP (standard temperature and pressure) oxygen combustion values respectively ($m^3\ O_2 \cdot g^{-1}$ carbohydrate, fat or protein).

Equation (8.45) conservatively predicts an inhalation rate sufficient to allow complete oxidation of all the ingested carbohydrate, fat and protein.

8.3.4.4 Magnitudes of Man's Ingestion and Inhalation Rates

Table 8-1 shows man's ingestion and inhalation rates calculated from Equations (8.42), (8.44) and (8.45) using GM values of the parameters (Section 8.5). These rates are in broad agreement with published values, but tend to be on the high side (Zach and Sheppard 1992). This is partly because our value for man's total energy need is set conservatively high (Section 8.5.6.1), and partly because literature values tend to be too low (Zach and Barnard 1987). Since high rates lead to greater nuclide intake by man, they are appropriate for the postclosure assessment.

TABLE 8-1

MAN'S INGESTION AND INHALATION RATES CALCULATED USING GEOMETRIC MEAN PARAMETER VALUES

Rate	Units	Value
Food Ingestion, U_j		
TE PLANT	kg wet$\cdot a^{-1}$	375.7
TE MILK	$L\cdot a^{-1}$ or kg$\cdot a^{-1}$	199.4
TE MEAT	kg wet$\cdot a^{-1}$	130.9
TE BIRD	kg wet$\cdot a^{-1}$	53.2
FW FISH	kg wet$\cdot a^{-1}$	10.0
Water Ingestion, Udw	L water$\cdot a^{-1}$	641
Inhalation, I_A	m^3 air$\cdot a^{-1}$	8617

8.4 INTERFACES

CALDOS is driven by nuclide concentrations in lake water, well water, soil (including, where appropriate, contributions from sediment) and air (Figure 8-3). Deposition rates from the atmosphere and from irrigation water to plant surfaces are also required. All these inputs are needed for each nuclide, with the exception of tritium, ^{39}Ar, ^{81}Kr, ^{85}Kr and radon.

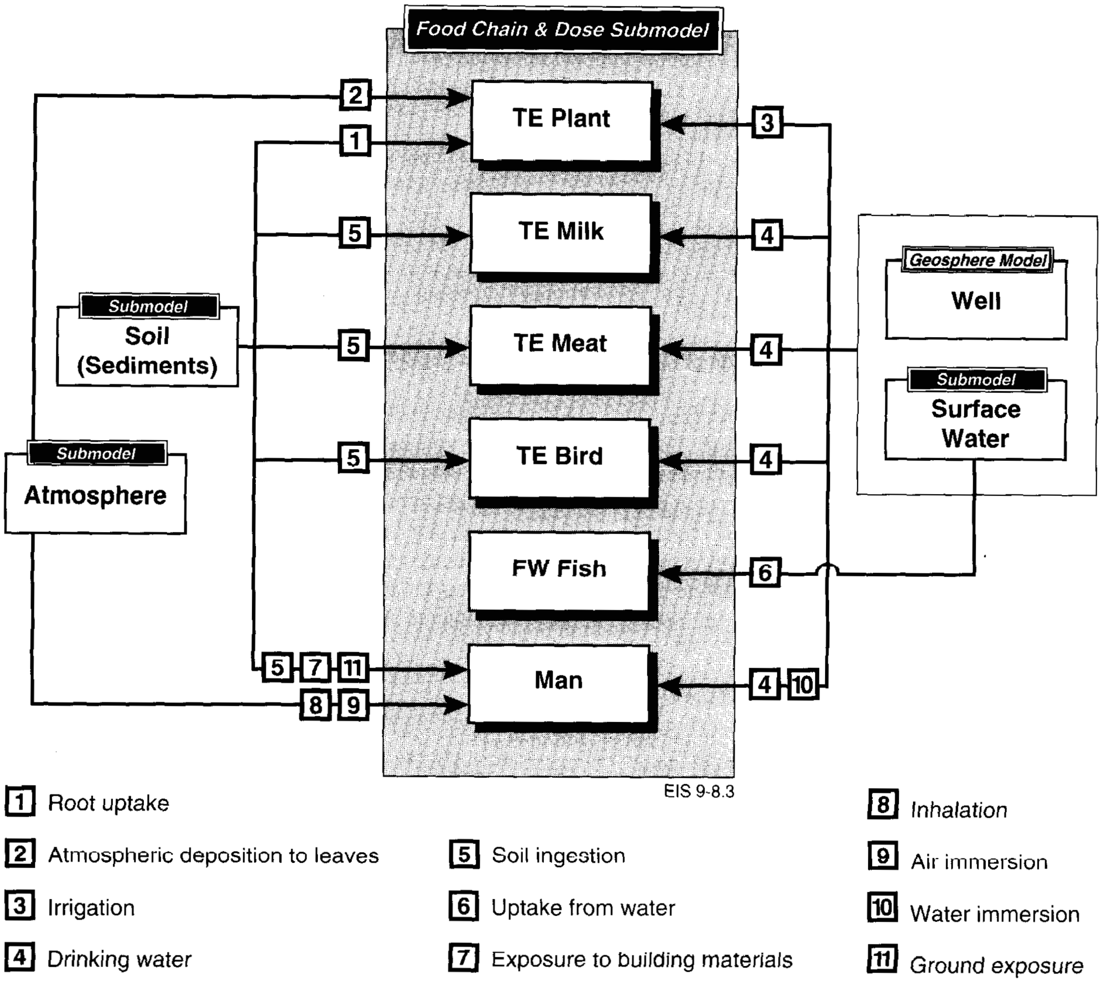


FIGURE 8-3: Interfaces (Related to Human Dose Prediction) Between the Food-Chain and Dose Submodel (Shaded) and the Other BIOTRAC Submodels

For tritium, only water concentrations are required; for the noble gases, only air concentrations are required. All inputs are readily available from the other three submodels of BIOTRAC (Chapters 5 to 7) or from the geosphere model (Chapter 4). The primary outputs of CALDOS are doses to man from all the important exposure pathways and from many minor pathways. These doses constitute one of the main end points of the postclosure assessment, and are compared with regulatory criteria to judge the acceptability of the disposal concept in terms of human safety (AECB 1987, Federal Environmental Assessment Review Panel 1992, Goodwin et al. 1994).

With its food types that can be used as generic organisms, output from CALDOS also plays a pivotal role in evaluating radiation doses to non-human biota, as explained in Chapter 13.

8.5 CALDOS PARAMETERS

CALDOS contains a large number of parameters. The data available for deriving appropriate values and distributions for these parameters are generally fairly good because of the work that has gone into assessing nuclear power installations and waste disposal concepts around the world. For the postclosure assessment, some parameters have been distributed, and some have been fixed. Distributions were assigned to parameters that show significant variability in space or time, or that describe processes that are not fully understood. Fixed values were specified for parameters that exhibit little variability and are reasonably well known. Where possible, values and distributions were based on data characteristic of the Canadian Shield, although data from other sources have also been used. The values generally represent annual averages, and were chosen to be conservative.

For convenience, we discuss the parameters in groups arranged according to the processes they describe. For each parameter, we show how appropriate values and distributions were derived from the available data. Many of the finer points relating to the interpretation of the data and to the derivation of parameter values from them are discussed only briefly here; further details are available in Zach and Sheppard (1991, 1992).

8.5.1 Transfer Coefficients

Transfer coefficients allow the nuclide concentration in a receptor compartment to be calculated from the given concentration in a source compartment (Equation (8.1)). Transfer coefficients are highly aggregated empirical parameters that quantify nuclide transfer through the food chain in a holistic way, accounting for diverse physical, chemical and biological processes. Values for given transfer coefficients are usually highly variable (Zach 1980a), reflecting differences in biological systems and uncertainty in measurement technique.

Transfer coefficient values vary widely among elements (Zach and Sheppard 1992), although certain groups of elements tend to behave similarly because of comparable physical and chemical properties. Such similarities can be used to assist in determining transfer coefficient values when data are inadequate. Although there is some evidence that isotopes of the same element exhibit different environmental behaviour, such effects are very small and we follow the accepted practice of using element-specific values for all the transfer coefficients. The use of transfer coefficients also implies that food-chain transfer of nuclides occurs linearly, regardless of concentration, the presence of other elements, or the ability of an organism to regulate uptake.

There are few specific data from the Canadian Shield from which transfer coefficient values can be derived. The distributions we have chosen are representative of current agricultural practices and conditions, including those of the Shield. The measurement of transfer coefficients is an ongoing area of research, but the more recently determined values tend to confirm older ones.

8.5.1.1 Plant/Soil Concentration Ratio, Bv^i ((Bq·kg^{-1} wet biomass)/(Bq·kg^{-1} dry soil))

Plant/soil concentration ratios are defined as the ratio of nuclide concentration in plant material to that in soil. They quantify the transfer of nuclides from soil to plants under steady-state conditions or at the time of harvest when plants are removed from contamination sources. Bv^i values are used in Equations (8.2), (8.3) and (8.6) to predict nuclide concentrations in plants eaten by humans or animals, and in Equation (8.40) to calculate concentrations in trees used as building materials. They are also used in Equation (7.25) in connection with agricultural fires. The uptake of elements by plants from soil is a complex phenomenon, highly dependent on a variety of plant, soil and environmental factors (Zach and Sheppard 1992). The observed plant/soil concentration ratios are highly variable for this reason and because of different measurement techniques. The most relevant data for waste disposal assessment come from field studies or from long-term experiments in outdoor settings where soil concentrations have reached steady state. Because the distribution of nuclides within plants can vary considerably, concentrations used for determining Bv^i values must be based on the plant parts actually used by humans or animals.

Several major compendia of average generic Bv^i values for use in environmental and safety assessments have been published (Ng et al. 1982a, Baes et al. 1984, IUR 1984, Coughtrey et al. 1985). The compendium by Baes et al. (1984) is outstanding for several reasons. It contains Bv^i values for most elements of interest in nuclear fuel waste management for both vegetative and reproductive plant parts. The values are of high quality because the data from which they were calculated were put through strict quality control procedures. Physical, chemical and biological similarities were used to support questionable values and to predict missing ones. We have adopted the Bv^i values by Baes et al. (1984) for vegetative plant parts. These values were originally expressed on a dry plant weight basis; to convert to a wet biomass basis, as required in CALDOS, we divided the published values by a factor of four, assuming a plant water content of 75% (Ng et al. 1968, Garten 1978). The resulting values are listed in Table 8-2. The values for vegetative plant parts are conservative because they are slightly higher than those for reproductive parts.

Although the values by Baes et al. (1984) are mainly based on agricultural crops, they are probably also appropriate for wild plant species on the Canadian Shield (Zach et al. 1989).

Plant/soil concentration ratios tend to be lognormally distributed (Sheppard M.I. and Thibault 1983, Zach et al. 1989, Sheppard S.C. and Evenden 1990). The values listed in Table 8-2 were therefore taken as the GMs of our distributions. The variation in Bv^i values has not been reported extensively. The studies that have been done (e.g., Ng et al. 1982a, Baes et al. 1984, IUR 1984, Sheppard S.C. and Evenden 1990) have produced varied results. Variation appears to be mainly a function of the number of samples and the range of conditions under which Bv^i was measured rather than of the physical or chemical properties of the system studied. Zach and Sheppard (1992) reviewed the available information and concluded that there was no justification for defining element-specific GSDs. They recommended a GSD of 10 for all elements, which is adopted here.

TABLE 8-2

ELEMENT-SPECIFIC GEOMETRIC MEANS OF THE LOGNORMAL DISTRIBUTIONS FOR THE PLANT/SOIL CONCENTRATION RATIO, Bv^i, TERRESTRIAL ANIMAL TRANSFER COEFFICIENT, F^i_j, AND AQUATIC CONCENTRATION RATIO, B^i_j

	Bv^i	F^i_j			B^i_j
Element	Plant/Soil ((Bq·kg^{-1} wet)/ (Bq·kg^{-1} dry))	TE MILK (d·L^{-1})	TE MEAT (d·kg^{-1} wet)	TE BIRD (d·kg^{-1} wet)	FW FISH (L·kg^{-1} wet)
Ac	8.8×10^{-4}	2.0×10^{-5}	2.5×10^{-5}	2.5×10^{-3}*	2.5×10^{1}
Am	1.4×10^{-3}	4.1×10^{-7}	3.5×10^{-6}	8.5×10^{-3}	1.0×10^{2}
Ar	0.0	0.0	0.0	0.0	0.0
Be	2.5×10^{-3}	9.1×10^{-7}	1.0×10^{-3}	1.0×10^{-1}*	2.0×10^{0}
Bi	8.8×10^{-3}	5.0×10^{-4}	4.0×10^{-4}	4.0×10^{-2}*	1.5×10^{1}
Br	3.8×10^{-1}	2.0×10^{-2}	2.5×10^{-2}	2.5×10^{0}*	4.2×10^{2}
C	5.5×10^{0}**	1.5×10^{-2}	6.4×10^{-2}	6.4×10^{0}*	5.0×10^{4}
Ca	8.8×10^{-1}	1.1×10^{-2}	1.6×10^{-3}	4.4×10^{-1}	4.0×10^{1}
Cd	1.4×10^{-1}	1.5×10^{-3}	3.5×10^{-4}	8.4×10^{-1}	2.0×10^{2}
Cr	1.9×10^{-3}	1.1×10^{-3}	9.2×10^{-3}	9.2×10^{-1}*	2.0×10^{2}
Cs	2.0×10^{-2}	7.1×10^{-3}	2.6×10^{-2}	4.4×10^{0}	1.0×10^{4}
H++					
Hf	8.8×10^{-4}	5.0×10^{-6}	1.0×10^{-3}	1.0×10^{-1}*	3.3×10^{0}
I	3.8×10^{-2}+	9.9×10^{-3}	7.0×10^{-3}	2.8×10^{0}	5.0×10^{1}
K	2.5×10^{-1}	7.2×10^{-3}	1.8×10^{-2}	1.8×10^{0}*	1.0×10^{3}
Kr	0.0	0.0	0.0	0.0	0.0
Mo	6.3×10^{-2}	1.4×10^{-3}	6.8×10^{-3}	5.0×10^{-1}	1.0×10^{1}
Nb	5.0×10^{-3}	2.0×10^{-2}	2.5×10^{-1}	3.0×10^{-3}	1.0×10^{2}
Ni	1.5×10^{-2}	1.0×10^{-3}	2.0×10^{-3}	2.0×10^{-1}*	1.0×10^{2}
Np	2.5×10^{-2}	5.0×10^{-6}	5.5×10^{-5}	5.5×10^{-3}*	2.5×10^{3}
P	8.8×10^{-1}	1.6×10^{-2}	4.9×10^{-2}	4.9×10^{0}*	6.7×10^{4}
Pa	6.3×10^{-4}	5.0×10^{-6}	1.0×10^{-5}	1.0×10^{-3}*	1.1×10^{1}
Pb	1.1×10^{-2}	2.6×10^{-4}	4.0×10^{-4}	4.0×10^{-2}*	3.0×10^{2}
Pd	3.8×10^{-2}	1.0×10^{-2}	4.0×10^{-3}	4.0×10^{-1}*	1.0×10^{1}
Po	6.3×10^{-4}	3.4×10^{-4}	4.5×10^{-3}	4.5×10^{-1}*	5.0×10^{2}
Pu	1.1×10^{-4}	1.0×10^{-7}	2.0×10^{-6}	7.6×10^{-3}	2.5×10^{2}
Ra	3.3×10^{-3}	4.0×10^{-4}	9.0×10^{-4}	9.0×10^{-2}*	5.0×10^{1}
Rb	3.8×10^{-2}	1.2×10^{-2}	1.1×10^{-2}	1.1×10^{0}*	2.0×10^{3}
Re	3.8×10^{-1}	1.3×10^{-3}	8.0×10^{-3}	8.0×10^{-1}*	1.2×10^{2}
Rn	0.0	0.0	0.0	0.0	0.0
Sb	5.0×10^{-2}	1.1×10^{-4}	1.0×10^{-3}	1.0×10^{-1}*	2.0×10^{2}
Se	6.3×10^{-3}	4.0×10^{-3}	1.5×10^{-2}	9.3×10^{0}	1.7×10^{2}
Si	8.8×10^{-2}	2.5×10^{-5}	4.0×10^{-5}	4.0×10^{-3}*	2.5×10^{0}

continued...

TABLE 8-2 (concluded)

	Bv^i	F^i_j			B^i_j
Element	Plant/Soil ((Bq·kg^{-1} wet)/ (Bq·kg^{-1} dry))	TE MILK (d·L^{-1})	TE MEAT (d·kg^{-1} wet)	TE BIRD (d·kg^{-1} wet)	FW FISH (L·kg^{-1} wet)
Sm	2.5×10^{-3}	2.0×10^{-5}	5.0×10^{-3}	5.0×10^{-1}*	3.0×10^{1}
Sn	7.5×10^{-3}	1.2×10^{-3}	8.0×10^{-2}	8.0×10^{0}*	3.0×10^{3}
Sr	6.3×10^{-1}	1.4×10^{-3}	8.1×10^{-4}	3.0×10^{-1}	1.0×10^{2}
Ta	2.5×10^{-3}	2.8×10^{-6}	6.0×10^{-4}	6.0×10^{-2}*	3.0×10^{4}
Tc	2.4×10^{0}	9.9×10^{-4}	8.5×10^{-3}	1.9×10^{0}	1.5×10^{1}
Te	6.3×10^{-3}	2.0×10^{-4}	1.5×10^{-2}	1.5×10^{0}*	4.0×10^{2}
Th	2.1×10^{-4}	5.0×10^{-6}	6.0×10^{-6}	6.0×10^{-4}*	1.0×10^{3}
U	2.1×10^{-3}	3.7×10^{-4}	2.0×10^{-4}	1.2×10^{0}	5.0×10^{1}
Y	3.8×10^{-3}	2.0×10^{-5}	3.0×10^{-4}	3.0×10^{-2}*	1.0×10^{2}
Zr	5.0×10^{-4}	3.0×10^{-5}	2.0×10^{-2}	2.0×10^{0}*	2.0×10^{2}

* Value set at 100 times the corresponding F^i_j value for TE MEAT.

** Includes atmospheric deposition of ^{14}C from all sources except irrigation water.

\+ Includes atmospheric deposition of ^{129}I suspended from the soil as a gas.

++ No transfer coefficients required because of specific-activity model.

Bv^i values depend on the mobility of the nuclide in soil, as described by the soil solid/liquid partition coefficient, Kd^i (Section 6.5.3). A nuclide with a high Kd^i value will be tightly bound to soil solids; little will be available in the aqueous phase for root uptake, and its Bv^i value will be low. Bv^i is therefore negatively correlated with Kd^i. Correlation coefficients, r, ranging from -0.47 to -0.88 have been reported in the literature (Baes 1982, Sheppard S.C. 1985, 1986). The data are not numerous enough to allow element-specific coefficients to be derived. Therefore, we chose a value of -0.7 for all elements for the postclosure assessment.

Since Kd^i values in the soil model are classified according to soil type, the negative correlation effectively categorizes the Bv^i values according to soil type.

For most nuclides, plant/soil concentration ratios account for root uptake only. However, for ^{14}C and ^{129}I, Bv^i also accounts for some nuclide deposition to leaves originating from contaminated soil (Zach and Sheppard 1992). As gases, both carbon and iodine may evade the soil and immerse the vegetation above. The plant concentrations used to calculate Bv^i will therefore reflect both root uptake and leaf deposition from such emissions.

For ^{129}I, we do not separately model deposition of the airborne gaseous nuclide from soil to leaves (Section 7.3.8). However, we do model leaf deposition of ^{129}I from other sources, including irrigation water, lake-derived nuclide, and particulates from the soil.

Baes et al. (1984) do not list a Bv^i value for ^{14}C. Since plants absorb most of the carbon required for photosynthesis from the atmosphere, we derived a Bv^i value for carbon by assuming that the specific activity of ^{14}C in air and vegetation is the same as in the soil (USNRC 1977, Zach and Sheppard 1992). Root uptake and leaf deposition are considered together, and the Bv^i value accounts for both. It is therefore not necessary to model the leaf-deposition pathway separately (Section 7.3.8) for any ^{14}C source except irrigation water.

Using the specific-activity model, we calculated 500 values of Bv^i for carbon, taking into account variations in the dry organic matter content of soil and the frequency of occurrence of the four soil types considered in our assessment (Section 6.5.1.1). The resulting distribution was bimodal, with a GM of 5.5 ((Bq·kg^{-1} wet biomass)/(Bq·kg^{-1} dry soil)) and extreme values of 0.07 and 219; the GSD was 9.7 (Sheppard S.C. 1989). The GM is identical to the value recommended by the Nuclear Regulatory Commission (USNRC 1977), and the GSD close to the value of 10 specified for the other elements. On the basis of these results, we have established a lognormal distribution for the Bv^i values for carbon with a GM value of 5.5 ((Bq·kg^{-1} wet biomass)/(Bq·kg^{-1} dry soil)) and a GSD of 10.0. The distribution has lower and upper truncation values of 0.07 and 220 ((Bq·kg^{-1} wet biomass)/(Bq·kg·dry soil)) respectively, which correspond to extremes in the organic matter content of soils.

In summary, our Bv^i PDFs are element-specific and lognormal, with GM values corresponding to those calculated by Baes et al. (1984) for vegetative plant parts (Table 8-2). The exception is the value for carbon, which is based on the USNRC (USNRC 1977) and our own calculations. All the distributions have a GSD of 10.0, and the correlation coefficient between log-transformed soil Kd^i and Bv^i values is -0.7 in all cases. None of the distributions is truncated, except that for carbon. Although our distributions are mainly based on data from agricultural crops, we have used them for calculating nuclide concentrations in wooden building material as well.

8.5.1.2 Terrestrial Animal Transfer Coefficients, F_j^i (d·L^{-1} or d·kg^{-1} wet biomass)

This transfer coefficient quantifies the transfer of nuclides to the terrestrial animal food types TE MILK, TE MEAT and TE BIRD from feed or forage (Equations (8.6) and (8.12)), drinking water (Equation (8.14)), or soil (Equation (8.16)) under steady-state conditions or at the time of slaughter. For milk, F_j^i is defined as the portion of an element ingested daily that is secreted per litre of milk. For the other two coefficients, F_j^i refers to the portion of an element ingested daily that is incorporated into 1 kg of meat (TE MEAT and TE BIRD) or 1 kg of the edible parts of eggs (TE BIRD).

The uptake and metabolism of elements in mammals and birds are complex and element-specific (Zach and Sheppard 1992). F_j^i values vary considerably, depending on factors such as the metabolic function of the element, the chemical and physical forms of the nuclide, the level of stable elements in an animal's diet, and the type and ingestion rate of food and forage. Variation can also arise from the experimental techniques used to measure F_j^i, which generally involve the use of tracers (Ng 1982).

Physical, chemical and biological similarities of elements have been used extensively to supplement missing data for F_j^i. This has also involved extrapolation between animal species (Ng and Hoffman 1983), so that the database includes a variety of terrestrial animals, making transfer coefficient values broadly representative.

F_j^i values have typically been determined for biologically assimilated nuclides taken up from feed or forage. Such nuclides tend to be more readily adsorbed than those in water or soil. Our assumption that the same F_j^i values can be used for uptake from feed or forage, soil or water is therefore conservative.

Many compendia of generic F_j^i values are in existence (Ng et al. 1977, 1982b, USNRC 1977, 1983b, Baes et al. 1984, NCRP 1984, CSA 1987). Most of the values are similar from compendium to compendium because they are based largely on the same research data.

TE MILK

More data are available for deriving F_j^i values for milk than for any of the other terrestrial animal transfer coefficients. Most of the data pertain to cow's milk; transfer coefficients for goat's milk tend to be somewhat higher. With the exception of technetium, we have taken our F_j^i values for TE MILK from the compilation of Ng et al. (1977), taking into account subsequent revisions made to accommodate new information (Ng 1982). These values, which are listed in Table 8-2, are of high quality because they are based on a thorough literature review and rigorous screening of the available data. The compendia of Baes et al. (1984) and of the CSA (CSA 1987) borrow heavily from Ng's work.

Ng's F_j^i value for technetium is equal to that for iodine, assuming that the two elements have identical metabolic behaviour. However, recent studies have shown that this analogy may be inappropriate and that technetium is much less readily transferred to milk (Wiechen et al. 1983, Bondietti and Garten 1986). Experimentally determined F_j^i values for technetium are two to three orders of magnitude below those for iodine (Wiechen et al. 1983, Voigt et al. 1987, Johnson J.E. et al. 1988). Because of the uncertainty that still exists, we have adopted a conservative intermediate value that is one order of magnitude lower than the commonly accepted iodine value.

F_j^i values for milk are distributed lognormally (Hoffman 1979, Ng and Hoffman 1983). The values listed in Table 8-2 were therefore taken to be the GMs of the distributions. The variation in F_j^i values has not been reported extensively. The available information supports a GSD of 3.2 for all nuclides, a value that we have adopted for BIOTRAC.

TE MEAT

The database for F_j^i values for meat is limited and not fully documented. Most of the values pertain to domestic mammals, particularly cattle, although information is also available for pork, lamb and venison from white-tailed deer, caribou and moose. The values for cattle tend to be slightly lower than those for other animals. For most elements, we have drawn our F_j^i values for TE MEAT from the compendium of Baes et al. (1984). For the remaining elements, we adopted the revised values published by Ng et al. (1982b). The value for carbon was taken from the CSA (CSA 1987). Baes et al. (1984) based their values for all the elements with an atomic number exceeding 82 on systematic trends in the periodic table for the plant/soil concentration ratio and the milk transfer coefficient. This was thought to be preferable to relying on incomplete observational data. Many of the values are very conservative when considered in terms of feed or forage consumption and efficiency of transfer. Many of the CSA (1987) values are similar to ours because of common sources and selection procedures. Our F_j^i values for TE MEAT are listed in Table 8-2.

Few data are available on the distribution of F_j^i values for meat. We have assumed that F_j^i is lognormally distributed, taking into account the observed data (e.g., Ward and Johnson 1965, Zach et al. 1989) and the distribution of F_j^i values for milk. For each nuclide, the GM of the distribution was set equal to the value given in Table 8-2. We chose a conservative GSD of 3.2, which is higher than almost all of the observed values. It is also large enough to cover most of the relatively large F_j^i values reported for wild game.

TE BIRD

The database for F_j^i values for poultry and eggs is not extensive. Several compendia do exist (Fletcher and Dotson 1971, Baker 1977, USNRC 1977, CSA 1987), but their values are largely undocumented. Most of the available data pertain to chickens, and indicate that transfer coefficients for eggs tend to be one to two orders of magnitude higher than those for poultry. The few values available for game birds (ducks and grouse) are somewhat lower than those for chickens.

The type of PDF for the F_j^i values of poultry and eggs has not been statistically documented for any element. However, it is reasonable to assume that all the terrestrial transfer coefficients have similar distributions because of common underlying physical, chemical and biological processes (Zach and Sheppard 1992). Thus, given lognormal distributions for milk and meat, we have assumed lognormal distributions for TE BIRD for our assessment.

The best supported F_j^i values for TE BIRD are found in the compendium of Ng et al. (1982b). However, this compendium covers only about one third of the relevant elements. We adopted these values for the postclosure assessment, picking the larger if there was a choice between poultry and eggs. These values were assigned to the GMs of the distributions for each element listed in Ng's report. For the remaining elements, we believe the published data are too unreliable to be used to establish F_j^i values. Instead, we derived F_j^i values for TE BIRD from the corresponding value for TE MEAT. The data for those elements with measured transfer coefficient values for both beef and poultry and eggs suggest that values for TE BIRD are about two orders of magnitude larger than the values for meat. Therefore, for the elements missing from Ng's compendium, we have set the GMs for F_j^i for TE BIRD equal to those for TE MEAT increased by a factor of 100. The final values are listed in Table 8-2.

Data for poultry and eggs are too limited to establish GSDs of the F_j^i values for TE BIRD. We adopted a GSD of 3.2 for all elements. This value is consistent with the available data and with the values chosen for F_j^i for TE MILK and TE MEAT.

8.5.1.3 Aquatic Concentration Ratio, B_j^i (m^3 water·kg^{-1} wet biomass)

The aquatic concentration ratio quantifies the transfer of nuclides from the lake environment to fish (Equation (8.18)). It is the ratio of the nuclide concentration in the edible portion of fish to the concentration in the water.

Nuclide uptake by freshwater fish is a complex phenomenon. Elements can be directly absorbed from the water or via the gastrointestinal tract following ingestion of contaminated sediment or aquatic organisms (Zach and Sheppard 1992). The parameter B_j^i does not directly account for food-chain transfer of nuclides from water or sediments. However, concentration ratios measured in natural situations include food-chain transfer implicitly, and this is being investigated furhter.

Because of the number and complexity of uptake mechanisms involved, the variability in B_j^i values is quite large. The variability is further increased by variations in water chemistry, the chemical form of the nuclide, the fish species and the nuclide distribution throughout the fish, as well as by experimental methods, and analytical and sampling errors. The total variability in B_j^i values is therefore closer to that of Bv^i than of F_j^i. The most reliable estimates of B_j^i come from tracer experiments in natural aquatic systems.

The average generic B_j^i values listed by Thompson et al. (1972) are the most commonly used assessment values (Blaylock 1982, NCRP 1984). The limited CSA (1987) compendium is largely undocumented, but it does represent a useful source of values for Canadian conditions. Recently, Poston and Klopfer (1986) have thoroughly reviewed the uptake of 26 elements by fish, and established generic concentration ratio values for environmental assessments. For the postclosure assessment, we preferentially used the B_j^i values from Posten and Klopfer (1986). Missing values were then taken from the CSA (1987) and finally from Thompson et al. (1972). Given a choice between values for fish with different feeding habits, or for water with different mineral contents, we used the higher, more conservative values in all cases. The final values are listed in Table 8-2. The values for cesium and strontium reflect the average concentrations of potassium and calcium respectively in Canadian Shield lakes (Zach and Sheppard 1992).

The available data suggest that B_j^i values are lognormally distributed (Hoffman and Baes 1979, Swanson and Richert 1987). We therefore interpret the values in Table 8-2 as GMs of lognormal PDFs. As noted above, the variability in B_j^i values tends to be large. We have conservatively adopted a GSD of 12.0 for all aquatic concentration ratios to cover the entire range of reported values.

The B_j^i values in Table 8-2 have units of L water$\cdot$kg^{-1} wet biomass, the units in which aquatic concentration rates are usually reported. However, units of m^3 water$\cdot$kg^{-1} wet biomass are required for Equation (8.18). Values in m$^3\cdot$kg^{-1} can be obtained by dividing the tabulated values by a factor of 10^3 L$\cdot$m^{-3}. The distribution type and GSD are unaffected by this unit conversion.

8.5.2 Dose Conversion Factors for Man

Here a DCF is the dose received by a human per unit radiation from internal or external exposure. Unlike transfer coefficients, which are element-specific (Section 8.5.1), DCFs are radionuclide-specific. Dose conversion factors are used to predict annual effective dose equivalents, or simply doses, to man following the intake of radionuclides into the body or exposure to external radiation fields (Section 1.2.3). Internal DCFs are difficult to measure experimentally, and are generally calculated using physical and biological models of radionuclide behaviour in the body and of the interaction of radiation with biological tissue. The requirements of these models for metabolic and dosimetric data are large. Dose conversion factors are usually calculated using the methodologies and databases recommended by the ICRP. These recommendations are based largely on scientific work compiled by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) and by the Committee on the Biological Effects of Ionizing Radiation (BEIR) of the U.S. National Academy of Sciences (NAS). The DCFs used in our assessment were calculated following the recommendations in ICRP 26 (ICRP 1977) and ICRP 30 (ICRP 1979), which have been formally accepted by many countries. As discussed in Section 1.2.3, there have been several recent changes in DCFs, but these changes have not been fully accepted or implemented, and would likely have relatively minor effects on our dose predictions (Goodwin et al. 1994).

The calculation of DCFs is a complex procedure (Myers 1989, Zach and Sheppard 1992). The models must take into account a number of factors, including the differential sensitivity of various human tissues and organs to radiation (organ factor, W_T); the distribution and turnover of radionuclides in the body (for internal exposure); the spatial distribution of radionuclides in the environment (for external exposure); and the type, energy and biological effectiveness of the radiation emitted by the radionuclide in question (radiation quality factor, Q).

Data for determining DCFs come from a variety of sources (ICRP 1975, 1979). These include studies on the uptake, distribution and excretion of radionuclides by laboratory animals and by humans deliberately or accidentally contaminated. Data on human and animal anatomy and physiological processes, and on the response of organs, tissues and cells to ionizing radiation are also required. Little is known about the exact variability of DCFs. Anatomical and physiological differences between individuals likely cause less than an order of magnitude variation in dose. Internationally accepted DCFs are available mainly for ICRP reference man (Section 1.5.4). The DCFs used for our assessment are therefore represented by single values rather than by probability density functions.

8.5.2.1 Internal Dose Conversion Factors

Nuclides that enter the human body through ingestion or inhalation may be absorbed and distributed to various tissues and organs from which they are eventually excreted (Camner et al. 1979). Standard, commonly accepted gastrointestinal tract and lung models exist to predict the absorption of ingested and inhaled radionuclides (ICRP 1979). These can be coupled with standard organ models to calculate radionuclide concentration in tissues and organs. If the decay rate of the radionuclide and the type of radiation emitted are known, absorbed doses can be calculated for each target organ, taking into account radiation from the target organ itself, and from other source organs nearby. The absorbed doses can be translated into DCFs by applying appropriate Q and W_T factors and by summing over all tissues and organs.

Most of our internal DCFs were calculated by Johnson J.R. and Dunford (1983). Missing values were supplied by Linauskas (1989a,b,c), who used the same or similar models and data as Johnson J.R. and Dunford. The DCFs include the effects from the decay of in vivo-produced daughters, which are assumed to stay with their precursors and behave metabolically in the same way. Accordingly, radionuclides with half-lives less than one day need not be modelled separately (Section 2.5.4). Special models have been used to calculate internal DCFs for tritium and ^{129}I in recognition of their unique biological properties (Section 2.5). This is also true for the geosphere limit to the internal ^{14}C dose (Section 8.3.1.12). Our DCFs agree well with ICRP 30 values and values published thereafter (Zach and Sheppard 1991, 1992).

<u>Ingestion Dose Conversion Factor, DFe^i</u>
($Sv \cdot Bq^{-1}$)

Ingestion DCFs convert radionuclides ingested by man with food, drinking water and soil into radiological doses (Equations (8.4), (8.7), (8.11), (8.13), (8.15), (8.17), (8.19), (8.21) and (8.24)). Most of the values used in BIOTRAC are from Johnson J.R. and Dunford (1983). However, values for americium, neptunium, plutonium and thorium radionuclides were taken from Johnson J.R. (1986), who incorporated new metabolic information on absorption from the gastrointestinal tract. Values for ^{10}Be, ^{208}Bi, ^{210m}Bi, ^{182}Hf, ^{40}K, ^{93m}Nb, ^{32}P, ^{87}Rb, ^{187}Re and ^{32}Si, which were not considered by Johnson J.R. and Dunford, were calculated by Linauskas (1989a,c), using the same methodology. The radionuclide-specific ingestion DCFs used for the postclosure assessment are listed in Table 8-3.

Johnson J.R. and Dunford (1983) used standard gastrointestinal tract and organ models to calculate DFe^i values for most radionuclides. Special models were used for the alkali earth elements and for technetium. The organ model for the alkali earth elements (calcium, strontium and radium) was revised to handle nuclide recycling and bone retention more realistically and to incorporate age dependence. The organ model for technetium uses a special retention function for the thyroid gland based on recent experimental data. Otherwise, dosimetric data from ICRP 30 (ICRP 1979) and methodologies from ICRP 26 (ICRP 1977) were used to compute our DFe^i values.

For some radionuclides, Johnson J.R. and Dunford (1983) calculated more than one value using different absorption fractions from the gastrointestinal tract. This can account for different chemical forms of the ingested radionuclides. In such instances, we have adopted the highest and most conservative DFe_i value for use in BIOTRAC.

Inhalation Dose Conversion Factor, DFi^i
($Sv \cdot Bq^{-1}$)

Inhalation DCFs convert radionuclides inhaled by man into radiological doses (Equation (8.26)). For most radionuclides, we adopted the DFi^i values of Johnson J.R. and Dunford (1983); the remainder were calculated by Linauskas (1989a,b,c). The values are listed in Table 8-3.

Johnson J.R. and Dunford (1983) calculated values for DFi^i using the lung model documented by the ICRP (1966). Inhaled radionuclides may be deposited and trapped in the lungs. Alternatively, they may be cleared from the lungs and transferred to the gastrointestinal tract through swallowing. Other inhaled material may be cleared into the blood or the lymphatic system deep within the lungs, and then passed into various tissues and organs. The lung, gastrointestinal tract and organ models are therefore closely linked.

The DFi^i values for the alkali earth elements and for technetium were calculated using the revised organ models discussed in the previous section. The DFi^i value for radon was also calculated in a special way. Although radon is inert, does not bind to body tissues and so gives only a small dose, its daughters are highly reactive and can give rise to a significant inhalation dose. Our DFi^i for radon was calculated by Linauskas (1989b) using ICRP (1986) methodology.

The dosimetric data required to implement the models for calculating DFi^i values were taken from ICRP 30 (ICRP 1979); Q and W_T factors were drawn from ICRP 26 (ICRP 1977). For some radionuclides, Johnson J.R. and Dunford (1983) calculated DFi^i values for more than one pulmonary clearance class. In all such cases, we adopted the highest value for our assessment to be conservative.

Internal Dose Conversion Factor for ^{129}I, DF^I
($(Sv \cdot a^{-1})/(Bq \cdot kg^{-1}$ thyroid))

This conversion factor is used to predict man's total internal dose arising from ingestion and inhalation of ^{129}I (Equation (8.29)). Because our ^{129}I model is based on a specific activity approach (Section 2.5.3), DF^I is calculated from the amount of ^{129}I in man's thyroid rather than from the rate of intake (Zach and Sheppard 1992). There is thus no need to use the gastrointestinal tract, lung and organ models to calculate a concentration per unit intake. Our value of 9.7×10^{-9} ($(Sv \cdot a^{-1})/(Bq \cdot kg^{-1}$ thyroid)) (Table 8-3) was obtained assuming the thyroid is the only surce and target organ.

TABLE 8-3

HUMAN INGESTION AND INHALATION DOSE CONVERSION FACTOR VALUES

Radionuclide	Ingestion DFe^i (Sv·Bq^{-1})	Inhalation DFi^i (Sv·Bq^{-1})
^{225}Ac	3.1×10^{-8}	2.7×10^{-6}
^{227}Ac	3.8×10^{-6}	1.6×10^{-3}
^{241}Am	1.2×10^{-6}	1.3×10^{-4}
^{39}Ar	0.0*	0.0
^{10}Be	1.3×10^{-9}	6.1×10^{-9}
^{208}Bi	1.4×10^{-9}	6.2×10^{-9}
^{210}Bi	1.7×10^{-9}	6.3×10^{-8}
^{210m}Bi	2.6×10^{-8}	2.1×10^{-6}
^{14}C	5.2×10^{-10}	5.2×10^{-12}
^{41}Ca	3.6×10^{-10}	3.3×10^{-10}
^{113m}Cd	6.9×10^{-7}	3.7×10^{-7}
^{135}Cs	1.9×10^{-9}	9.4×10^{-10}
^{3}H	2.9×10^{-8}	**
^{182}Hf	7.5×10^{-10}	3.9×10^{-8}
^{129}I	9.7×10^{-9}	**
^{40}K	5.0×10^{-9}	3.3×10^{-9}
^{81}Kr	0.0	0.0
^{85}Kr	0.0	0.0
^{93}Mo	3.8×10^{-10}	8.4×10^{-9}
^{93m}Nb	3.8×10^{-10}	7.9×10^{-9}
^{94}Nb	1.3×10^{-8}	1.3×10^{-7}
^{59}Ni	3.5×10^{-11}	2.0×10^{-10}
^{63}Ni	1.1×10^{-10}	5.2×10^{-10}
^{237}Np	1.1×10^{-6}	1.2×10^{-4}
^{32}P	2.3×10^{-9}	4.4×10^{-9}
^{231}Pa	2.9×10^{-6}	3.2×10^{-4}
^{233}Pa	9.4×10^{-10}	2.9×10^{-9}
^{205}Pb	4.7×10^{-10}	9.8×10^{-10}
^{210}Pb	1.6×10^{-6}	3.5×10^{-6}
^{107}Pd	4.1×10^{-11}	4.2×10^{-9}
^{210}Po	5.3×10^{-7}	2.5×10^{-6}
^{238}Pu	1.0×10^{-6}	1.2×10^{-4}
^{239}Pu	1.2×10^{-6}	1.3×10^{-4}
^{240}Pu	1.2×10^{-6}	1.3×10^{-4}
^{241}Pu	2.1×10^{-8}	2.3×10^{-6}
^{242}Pu	1.1×10^{-6}	1.4×10^{-4}

continued...

TABLE 8-3 (concluded)

Radionuclide	Ingestion DFe^{i} $(Sv \cdot Bq^{-1})$	Inhalation DFi^{i} $(Sv \cdot Bq^{-1})$
^{223}Ra	1.5×10^{-7}	2.5×10^{-6}
^{224}Ra	9.3×10^{-8}	9.9×10^{-7}
^{225}Ra	5.0×10^{-8}	2.5×10^{-6}
^{226}Ra	3.2×10^{-7}	2.6×10^{-6}
^{228}Ra	2.6×10^{-7}	1.3×10^{-6}
^{87}Rb	1.1×10^{-9}	7.0×10^{-10}
^{187}Re	2.6×10^{-12}	1.5×10^{-11}
^{222}Rn	0.0	1.4×10^{-8}
^{125}Sb	8.2×10^{-10}	4.9×10^{-9}
^{126}Sb	2.3×10^{-9}	3.2×10^{-9}
^{79}Se	2.3×10^{-9}	2.5×10^{-9}
^{32}Si	7.5×10^{-10}	3.6×10^{-9}
^{126}Sn	9.1×10^{-10}	5.3×10^{-9}
^{90}Sr	3.4×10^{-8}	4.2×10^{-7}
^{182}Ta	4.5×10^{-9}	1.4×10^{-8}
^{99}Tc	6.5×10^{-10}	2.7×10^{-9}
^{125m}Te	2.6×10^{-9}	2.1×10^{-9}
^{227}Th	7.9×10^{-8}	5.1×10^{-6}
^{228}Th	2.5×10^{-6}	1.1×10^{-4}
^{229}Th	2.4×10^{-5}	5.6×10^{-4}
^{230}Th	3.5×10^{-6}	8.1×10^{-5}
^{231}Th	3.6×10^{-10}	2.2×10^{-10}
^{232}Th	1.9×10^{-5}	4.1×10^{-4}
^{234}Th	2.8×10^{-9}	8.4×10^{-9}
^{232}U	4.1×10^{-7}	2.1×10^{-4}
^{233}U	8.7×10^{-8}	4.4×10^{-5}
^{234}U	8.5×10^{-8}	4.3×10^{-5}
^{235}U	7.9×10^{-8}	4.0×10^{-5}
^{236}U	8.1×10^{-8}	4.1×10^{-5}
^{238}U	7.5×10^{-8}	3.8×10^{-5}
^{90}Y	2.9×10^{-9}	2.1×10^{-9}
^{93}Zr	4.5×10^{-10}	7.7×10^{-8}

* Values of 0.0 indicate doses are very low and need not be considered because radionuclides are not absorbed and deposited in the body.

** Internal dose conversion factors, DF^{H3} and DF^{I}, account for both ingestion and inhalation using a specific activity approach, and are expressed in units of $((Sv \cdot a^{-1})/(Bq \cdot kg^{-1}$ soft tissue or thyroid)).

Internal Dose Conversion Factor for ^{14}C, $DF^{c\prime}$
(($Sv \cdot a^{-1}$)/($Bq \cdot kg^{-1}$ soft tissue))

This conversion factor used to calculate an upper limit to man's total internal ^{14}C dose is based on the specific activity of ^{14}C in groundwater (Equation (8.33)). $DF^{c\prime}$ differs from the dose conversion factor used in the transport equations and listed in Table 8-3 because it is calculated from the amount of ^{14}C in soft tissue rather than from the intake rate. The value for $DF^{c\prime}$ was determined using the methodology applied to the dose conversion factor for ^{129}I, and was found to be 2.5 x 10^{-7} (($Sv \cdot a^{-1}$)/($Bq \cdot kg^{-1}$ soft tissue)) (Zach and Sheppard 1992). Carbon-14 is assumed to be uniformly distributed in the body's soft tissues, which absorb all the emitted radiation.

Internal Dose Conversion Factor for Tritium, DF^{H3}
(($Sv \cdot a^{-1}$)/($Bq \cdot kg^{-1}$ soft tissue))

DF^{H3} is used to predict man's total internal dose from the ingestion and inhalation of tritium (Equation (8.34)). Our value of 2.9 x 10^{-8} (($Sv \cdot a^{-1}$)/($Bq \cdot kg^{-1}$ soft tissue)) (Table 8-3) was calculated by Johnson J.R. (1988) and Zach and Sheppard (1992) using the same methodology that was applied to DF^{I} and $DF^{c\prime}$. Hydrogen (and the tritium associated with it through the specific activity approach) is assumed to be uniformly distributed in the body's soft tissues, which absorb all the emitted radiation.

8.5.2.2 External Dose Conversion Factors

Radionuclides in the environment set up radiation fields that can lead to exposure of humans and all the other biota. External DCFs are determined by first calculating the absorbed dose in the environmental medium (e.g., air), taking account of the geometric relationship between the exposed individual and the medium, and of the attenuation of radiation in the medium. The absorbed dose in the medium is then used to calculate the dose for various target organs, taking into consideration the energy of the radiation and factors that account for shielding by overlying tissues. Absorbed doses by the target organs are then translated into DCFs by applying appropriate Q and W_T factors (ICRP 1977, 1978). Because of its very limited ability to penetrate, α radiation contributes very little to external doses and is not considered in calculating external DCFs. Similarly, β radiation can only affect tissues or organs at or very near the body surface, and the skin is normally the only target organ considered. Shielding by clothing is ignored when calculating external DCFs.

The external DCFs used in our assessment were calculated by Holford (1989), with the exception of the value for ^{208}Bi, which comes from Holford (1988). The 1989 values are based on up-to-date information consistent with ICRP 38 (ICRP 1983), and with the data used to calculate our internal DCFs (Section 8.5.2.1). Our shielding factors were taken from Barnard and D'Arcy (1986), as demonstrated by Zach and Sheppard (1992). The resulting external DCFs are similar to those derived by Kocher (1983) and Barnard and D'Arcy (1986). The DCFs for radionuclides with half-lives less than one day, which are not explicitly considered in BIOTRAC, are added to those of their precursors.

We made a number of conservative assumptions in deriving our external DCFs. Accordingly, it is likely that they account for exposure to minor sources, such as household goods, clothing, cosmetics and so on, which are not explicitly included in BIOTRAC. Given the critical group concept (Section 1.5.4), such items cannot be assumed to remain uncontaminated.

Air Immersion Dose Conversion Factor, DFa^i
$((Sv \cdot a^{-1})/(Bq \cdot m^{-3}$ air))

Dose conversion factors are used to predict radiological doses to man from immersion in contaminated air (Equation (8.35)). Our DFa^i values, as calculated by Holford (1988, 1989), are listed in Table 8-4. They assume that reference man stands in a semi-infinite volume of uniformly contaminated air, with his major body organs located 1.0 m above the ground. This geometry is used to predict the absorbed dose in air, followed by the absorbed dose by the various target organs and the air immersion DCF, as described above (Section 8.5.2.2).

The atmosphere model (Chapter 7) predicts air concentrations at a height of 1.5 m above ground level, which is consistent with the height at which human inhalation occurs. For ground-level sources, the concentration would be slightly larger at 1.0 m, the height for which the immersion dose is calculated. However, the difference in concentration over the 0.5-m height is insignificant, and the 1.5-m concentration can be used reliably to predict the immersion dose. In general, inhalation is a much more important dose contributor than air immersion because it involves exposure from internal radionuclides.

Water Immersion Dose Conversion Factor, DFh^i
$((Sv \cdot a^{-1})/(Bq \cdot m^{-3}$ water))

To predict radiological doses to man arising from swimming or bathing in contaminated water DFh^i values are used (Equation (8.36)). These values, calculated by Holford (1988, 1989), are listed in Table 8-4. The values are based on the assumption that reference man is totally submerged in a semi-infinite volume of uniformly contaminated water. His body centroid is 0.1 m below the water surface. Contaminated sediments are assumed to be far enough beneath the surface that they contribute nothing to the dose. The DFh^i values are then calculated using the methods described above. Because water is much denser than air, it is more efficient at absorbing radiation, and, consequently, DFh^i values are about three orders of magnitude lower than the DFa^i value for the same radionuclide.

Ground Exposure Dose Conversion Factor, DFg^i
$((Sv \cdot a^{-1})/(Bq \cdot kg^{-1}$ wet soil))

The DFg^i values we use to calculate radiological doses to man from standing on contaminated ground (Equation (8.37)) were also calculated by Holford (1988, 1989), and are listed in Table 8-4. They assume that reference man stands on semi-infinite uniformly contaminated soil. His body centroid is assumed to be 0.8 m above the ground surface when calculating doses from γ radiation. For β radiation, doses are integrated over the body height. The soil concentrations used in applying Equation (8.37) are the predicted root-zone concentrations, which are uniform to a depth of 30 cm (Sections

TABLE 8-4

HUMAN AIR IMMERSION, WATER IMMERSION, GROUND EXPOSURE AND BUILDING EXPOSURE DOSE CONVERSION FACTOR VALUES

Radionuclide	Air Immersion DFa^i	Water Immersion DFh^i	Ground Exposure DFg^i	Building Exposure DFb^i
^{225}Ac*	3.4×10^{-7}	5.5×10^{-10}	3.9×10^{-7}	7.7×10^{-7}
^{227}Ac*	1.3×10^{-9}	2.3×10^{-12}	9.6×10^{-10}	2.3×10^{-9}
^{241}Am	2.8×10^{-8}	5.5×10^{-11}	7.9×10^{-9}	5.9×10^{-8}
^{39}Ar	3.3×10^{-9}	0.0**	0.0	0.0
^{10}Be	4.0×10^{-9}	4.4×10^{-12}	3.8×10^{-10}	0.0
^{208}Bi	4.4×10^{-6}	7.0×10^{-9}	5.6×10^{-6}	1.1×10^{-5}
^{210}Bi	5.4×10^{-9}	5.9×10^{-12}	9.4×10^{-10}	6.2×10^{-10}
^{210m}Bi*	3.8×10^{-7}	6.4×10^{-10}	4.5×10^{-7}	9.2×10^{-7}
^{14}C	6.0×10^{-11}	6.5×10^{-14}	2.3×10^{-13}	0.0
^{41}Ca	1.3×10^{-12}	1.5×10^{-15}	0.0	0.0
^{113m}Cd	2.6×10^{-9}	2.8×10^{-12}	2.2×10^{-10}	0.0
^{135}Cs	2.5×10^{-10}	2.7×10^{-13}	3.6×10^{-12}	0.0
^{3}H	0.0	0.0	0.0	0.0
^{182}Hf	3.4×10^{-7}	5.9×10^{-10}	4.1×10^{-7}	8.3×10^{-7}
^{129}I	1.3×10^{-8}	2.8×10^{-11}	2.2×10^{-9}	2.8×10^{-8}
^{40}K	2.4×10^{-7}	3.9×10^{-10}	3.0×10^{-7}	6.0×10^{-7}
^{81}Kr	8.4×10^{-9}	0.0	0.0	0.0
^{85}Kr	7.2×10^{-9}	0.0	0.0	0.0
^{93}Mo	1.6×10^{-9}	2.7×10^{-12}	2.1×10^{-10}	2.2×10^{-9}
^{93m}Nb	2.7×10^{-10}	4.8×10^{-13}	3.6×10^{-11}	3.9×10^{-10}
^{94}Nb	2.3×10^{-6}	3.9×10^{-9}	3.0×10^{-6}	5.9×10^{-6}
^{59}Ni	8.1×10^{-11}	9.5×10^{-14}	2.7×10^{-12}	3.8×10^{-12}
^{63}Ni	0.0	0.0	0.0	0.0
^{237}Np	3.4×10^{-8}	6.3×10^{-11}	2.2×10^{-8}	6.6×10^{-8}
^{32}P	1.4×10^{-8}	1.6×10^{-11}	4.4×10^{-9}	7.7×10^{-9}
^{231}Pa	5.3×10^{-8}	9.1×10^{-11}	6.2×10^{-8}	1.2×10^{-7}
^{233}Pa	2.8×10^{-7}	4.9×10^{-10}	3.3×10^{-7}	6.6×10^{-7}
^{205}Pb	1.7×10^{-10}	2.4×10^{-13}	1.5×10^{-11}	1.0×10^{-10}
^{210}Pb	2.1×10^{-9}	4.4×10^{-12}	4.3×10^{-10}	4.4×10^{-9}
^{107}Pd	0.0	0.0	0.0	0.0
^{210}Po	1.2×10^{-11}	2.1×10^{-14}	1.6×10^{-11}	3.2×10^{-11}
^{238}Pu	2.8×10^{-10}	5.0×10^{-13}	4.7×10^{-11}	4.1×10^{-10}
^{239}Pu	1.8×10^{-10}	3.2×10^{-13}	9.2×10^{-11}	2.9×10^{-10}
^{240}Pu	2.7×10^{-10}	4.8×10^{-13}	4.5×10^{-11}	4.0×10^{-10}
^{241}Pu	2.4×10^{-12}	4.3×10^{-15}	1.8×10^{-12}	4.6×10^{-12}
^{242}Pu	2.3×10^{-10}	4.0×10^{-13}	3.9×10^{-11}	3.4×10^{-10}

continued...

TABLE 8-4 (concluded)

Radionuclide	Air Immersion DFa^i	Water Immersion DFh^i	Ground Exposure DFg^i	Building Exposure DFb^i
^{223}Ra*	4.3×10^{-7}	7.3×10^{-10}	4.7×10^{-7}	9.4×10^{-7}
^{224}Ra*	2.5×10^{-6}	4.1×10^{-9}	3.1×10^{-6}	6.2×10^{-6}
^{225}Ra	1.0×10^{-8}	2.1×10^{-11}	1.8×10^{-9}	2.1×10^{-8}
^{226}Ra*	9.7×10^{-9}	4.4×10^{-9}	3.3×10^{-6}	6.8×10^{-6}
^{228}Ra*	1.4×10^{-6}	2.4×10^{-9}	1.8×10^{-6}	3.6×10^{-6}
^{87}Rb	9.1×10^{-10}	9.9×10^{-13}	2.8×10^{-11}	0.0
^{187}Re	0.0	0.0	0.0	0.0
^{222}Rn*	2.7×10^{-6}	0.0	0.0	0.0
^{125}Sb	6.0×10^{-7}	1.0×10^{-9}	7.5×10^{-7}	1.5×10^{-6}
^{126}Sb	4.1×10^{-6}	6.9×10^{-9}	5.2×10^{-6}	1.0×10^{-5}
^{79}Se	9.3×10^{-11}	1.0×10^{-13}	4.4×10^{-13}	0.0
^{32}Si	2.3×10^{-10}	2.4×10^{-13}	3.4×10^{-12}	0.0
^{126}Sn*	2.3×10^{-6}	3.4×10^{-9}	2.4×10^{-6}	5.0×10^{-6}
^{90}Sr	2.8×10^{-9}	3.1×10^{-12}	2.4×10^{-10}	0.0
^{182}Ta	1.9×10^{-6}	3.2×10^{-9}	2.4×10^{-6}	4.7×10^{-6}
^{99}Tc	8.0×10^{-10}	8.6×10^{-13}	2.6×10^{-11}	0.0
^{125m}Te	1.6×10^{-8}	3.4×10^{-11}	2.9×10^{-9}	7.5×10^{-8}
^{227}Th	1.5×10^{-7}	2.6×10^{-10}	1.7×10^{-7}	3.4×10^{-7}
^{228}Th	3.0×10^{-9}	5.4×10^{-12}	2.4×10^{-9}	5.6×10^{-9}
^{229}Th	1.2×10^{-7}	2.2×10^{-10}	9.5×10^{-8}	2.4×10^{-7}
^{230}Th	6.4×10^{-10}	1.2×10^{-12}	3.3×10^{-10}	1.2×10^{-9}
^{231}Th	1.8×10^{-8}	3.4×10^{-11}	9.5×10^{-9}	3.5×10^{-8}
^{232}Th	3.6×10^{-10}	6.6×10^{-13}	1.4×10^{-10}	6.4×10^{-10}
^{234}Th*	7.4×10^{-8}	1.2×10^{-10}	6.9×10^{-8}	1.5×10^{-7}
^{232}U	5.9×10^{-10}	1.1×10^{-12}	2.7×10^{-10}	1.0×10^{-9}
^{233}U	5.8×10^{-10}	1.0×10^{-12}	4.4×10^{-10}	1.0×10^{-9}
^{234}U	3.6×10^{-10}	6.5×10^{-13}	1.2×10^{-10}	5.8×10^{-10}
^{235}U	2.2×10^{-7}	3.9×10^{-10}	2.5×10^{-7}	5.0×10^{-7}
^{236}U	2.7×10^{-10}	4.8×10^{-13}	6.4×10^{-11}	4.1×10^{-10}
^{238}U	2.0×10^{-10}	3.6×10^{-13}	3.0×10^{-11}	3.0×10^{-10}
^{90}Y	2.0×10^{-8}	2.2×10^{-11}	7.3×10^{-9}	1.5×10^{-8}
^{93}Zr	0.0	0.0	0.0	0.0

Note: Units for DFa^i and DFh^i are ((Sv·a^{-1})/(Bq·m^{-3} air or water)); for DFg^i ((Sv·a^{-1})/(Bq·kg^{-1} wet soil)); and for DFb^i ((Sv·a^{-1})/(Bq·kg^{-1} dry material)).

* Precursor radionuclides include one or more short-lived daughters (Zach and Sheppard 1992)

** Values of 0.0 indicate DCFs less than 10^{-15} in the relevant units.

6.3.1.2 and 6.3.5). Concentrations at greater depths will in general be different, but will contribute little to the dose because of the rapid attenuation of the radiation field in soil.

The spatial configurations of man for air immersion and ground exposure are basically similar, as are the methods for calculating the DCFs. Because the body centroid and the source of radiation are not in contact for ground exposure, DFg^i tends to be smaller than DFa^i for a given radionuclide.

Building Material Dose Conversion Factor, DFb^i
$((Sv \cdot a^{-1})/(Bq \cdot kg^{-1}$ dry material$))$

In Equation (8.38) we use DFb^i values to determine radiological doses to man from living in buildings constructed of contaminated materials. Holford (1988, 1989) calculated three sets of DFb^i values using the same approach in each case, but assuming different materials (concrete, log and frame construction). For the postclosure assessment, we conservatively chose the highest of the three values for each radionuclide (Table 8-4).

In calculating DFb^i values, reference man is assumed to be at the centre of a spherical shell with a radius of 2.0 m and composed of uniformly contaminated building material. The spherical geometry greatly facilitates the calculations. The shell has the volume of an average room, and accounts for the walls, ceiling and floor. DFb^i values are calculated in much the same way as the DCFs for the other three external pathways. Our values do not allow for radiation exposure from adjoining rooms.

8.5.3 Holdup Times and Other Time Parameters

8.5.3.1 Holdup Times

Holdup times define the period between the removal of a product, such as food, from its contamination source and its use by humans; they allow for radioactive decay in this interval. Short holdup times lead to conservative dose estimates because there is less time for decay. Since doses are not strongly influenced by holdup times, except for the very short-lived nuclides, holdup times can be assigned fixed values rather than distributions for the postclosure assessment. We have implemented product-specific holdup time values to allow some variation.

Holdup Time for TE PLANT, th
(d)

The parameter th, denoting the holdup time for TE PLANT, defines the average length of time between the harvesting of terrestrial plants and their consumption by man (Equations (8.2), (8.3) and (8.8)). The value of th can be very short when local produce is eaten fresh (USNRC 1977, Rupp 1979), or can extend to months for canned and frozen foods. We assume conservatively that the critical group eats only local fresh produce, and set th for TE PLANT equal to 1.0 d.

Terrestrial Animal Feed Holdup Time, thf_j
(d)

This holdup time defines the average interval between removal of feed or forage from a field and the consumption of animal food types TE MILK, TE MEAT and TE BIRD by man (Equations (8.6) and (8.12)). It is made up of two components, which are additive: a feed or forage holdup time, which is the interval between harvesting of the feed and forage and consumption by the animal; and an animal holdup time, which is the interval between milking or slaughter and ingestion of the animal products by man. The first component is small when animals graze, but can amount to more than a year when they are fed stored foods such as hay (USNRC 1977). The second component is typically a few days when the animals are processed commercially, and one day for local milk and eggs (Rupp 1979). For our assessment, we adopted thf_j values of 1, 5 and 1 d for TE MILK, TE MEAT and TE BIRD respectively, assuming that the critical group consumes fresh milk, eggs and poultry, but that they age their meat slightly. These values are based on a holdup time of 1 d for feed and forage. This is conservative because it does not account for the long storage periods required on the Canadian Shield for much of the year.

Terrestrial Animal Drinking Water Holdup Time, thw_j
(d)

The parameter thw_j specifies the average period between water ingestion by animals and consumption of the respective food types TE MILK, TE MEAT and TE BIRD by man (Equation (8.14)). As in the case of thf_j, thw_j consists of two components, one for the water and one for the animal. Values for the latter are identical for thf_j and thw_j. We assume that farm and other animals drink directly from the lake, or have immediate access to well water, so that the first component is zero. Therefore, in our assessment, thw_j has values of 0, 4 and 0 d for TE MILK, TE MEAT and TE BIRD respectively.

Terrestrial Animal Soil Holdup Time, ths_j
(d)

This holdup time parameter defines the average interval between soil ingestion by animals and consumption of the animal food types by man (Equation (8.16)). It also consists of two components, one for the soil and the other for the animal. Values of the latter are identical to those for thf_j and thw_j. We set the first component to zero, assuming that the animals ingest soil while grazing, or deliberately to meet nutritional needs. Accordingly, we use ths_j values of 0, 4 and 0 d for TE MILK, TE MEAT and TE BIRD respectively.

Holdup Time for FW FISH, thp
(d)

The parameter thp defines the average period between the time a fish is caught and the time it is eaten by man (Equation (8.18)). Locally caught fish are generally consumed within one day (Rupp 1979), whereas commercially processed fish are stored for an average of about 10 d. For BIOTRAC, we have adopted a conservative thp value of 0.5 d.

Holdup Time for Man's Drinking Water, thdw

(d)

This holdup time parameter represents the average delay between the isolation of drinking water from its source (Section 9.1.2) and its consumption by man (Equation (8.20)). For the United States, Rupp (1979) determined average values of 0.5 and 1.0 d for local and commercial drinking water respectively. For our assessment, we conservatively assumed that drinking water has no holdup time, and set thdw = 0 d.

Holdup Time for Building Materials, thb

(d)

The average interval between removal of raw building materials from their contamination sources and human occupation of dwellings made from finished materials is defined by thb (Equations (8.39) and (8.40)). In normal commercial practice, wooden building materials are subject to holdup times of about half a year as a result of delays caused by harvesting the trees, processing, transporting, storing and building. Inorganic materials, such as clay, sand and gravel are generally handled more rapidly. We have adopted a thb_{wbm} value of 180 d for wooden and thb_{ibm} value of 30 d for inorganic building materials. These values are conservative because they do not allow for loss of radioactivity following completion of the dwellings.

8.5.3.2 Exposure Times

The time of exposure is the period during which various plant crops, trees and other wild plants are exposed to deposition from the atmosphere or from aerial irrigation water. Long exposure times are conservative because they increase the possibility of nuclide accumulation in plants. Exposure times show little variability and so can be represented by single values. We introduce some variability by specifying separate values for crops and trees.

Time of Exposure for Terrestrial Food Types, te_j

(d)

For most plant crops, the exposure time te_j is the period from emergence to harvesting, and so is related to the length of the growing season (Equations (8.8) and (8.12)). Most common food crops, such as radishes, tomatoes and cereals, have te_j values ranging from 20 to 100 d. In the case of forage crops, te_j is defined as the return time, the period between successive grazings of a given pasture area. The return time for cattle is largely determined by management practices, but is generally less than 30 d. Most conventional food-chain models use values of 30 and 60 d for grasses and all other crops respectively (USNRC 1977, Hoffman et al. 1984a, CSA 1987). For BIOTRAC, we adopted conservative values of 100, 50, 50 and 100 d for TE PLANT, TE MILK, TE MEAT and TE BIRD respectively. These values assume that cattle have a mixed diet of forage and stored full-season feed crops, whereas poultry feeds mainly on full-season grain crops.

Time of Exposure for Wood, teb
(d)

The time of exposure for wood, teb, is the interval between emergence of the seedling tree and its harvest many years later. It is used in Equation (8.40) to calculate nuclide concentrations in wood building materials. Many factors influence the time at which trees are harvested, including species, site productivity, method of harvesting and the occurrence of forest fires. Most of the trees harvested on the Canadian Shield are between 40 and 100 a old (McKee and Rowsell 1984). For the postclosure assessment, we chose a conservative teb value of 100 a, or 36 525 d. This value is somewhat at odds with our assumption that forest and land-clearing fires occur every 50 a (Section 7.3.5.3). Furthermore, the size of the woodlot is calculated on the assumption that the trees are harvested for fuel when they are 50 a old (Section 9.1.1.3). This inconsistency has no significant implications for the model predictions, and allows conservative values to be used in each part of BIOTRAC.

8.5.3.3 Plant Environmental Halftime, tp
(d)

The plant environmental halftime is the period during which half of the initial amount of deposited nuclide disappears from exposed plant parts through causes other than radioactive decay. Nuclides absorbed by the plant are not considered lost. This halftime is used in Equation (8.10) to calculate the effective removal constant from vegetation, λ_E^i. Nuclides may be lost from plant parts through several processes, including the action of wind and water, grazing, and growth of the plant itself. Environmental halftimes therefore depend upon factors such as leaf morphology, meteorological conditions, size of deposited particles, plant growth habits and method of measurement. Halftimes are therefore quite variable, and are treated as distributed parameters in CALDOS.

Values of tp are distributed lognormally (Miller and Hoffman 1979) and, with the possible exception of iodine, are not element-specific. On the basis of a review of the available data by Miller and Hoffman (1983), Zach and Sheppard (1992) recommended a GM of 12 d and a GSD of 2.0 for all elements. These values apply to all vegetation types, and to plant contamination via both atmospheric deposition and deposition from irrigation water.

8.5.4 Occupancy Factors for Man

Occupancy factors define the fractions of time humans are exposed to, and may receive a dose from, various environmental media. The different factors are not necessarily independent. For example, we assume that man is always exposed to either contaminated ground or contaminated building materials. The ground, Og, and building, Ob, occupancy factors must therefore add up to one; the sum defines the air occupancy factor, for which a separate value is not required. On the other hand, we assume that immersion in water does not preclude immersion in air or exposure to soil or building materials. In this case, the occupancy factors overlap. We have set occupancy factors for major activities only; exposure from standing on contaminated ice, for example, is assumed to be accounted for through the water immersion and ground exposure pathways, so that a separate ice occupancy factor is not required. Occupancy factors show little variability, and we have selected fixed, conservative values. In general, occupancy of indoor environments results in a higher dose than outdoor occupancy, so it is conservative to assume that individuals spend more time indoors.

8.5.4.1 Water Occupancy Factor, Oe (unitless)

The water occupancy factor is the annual average fraction of time that man spends immersed in water during bathing or swimming. It is used in Equation (8.36) to calculate external doses from radionuclides dissolved or suspended in the water. Canadian Shield residents likely spend more time bathing and showering than they do swimming outdoors. CSA (1987) has recommended an Oe value of 0.01. We have adopted the conservative value of 0.02, which corresponds to about 0.5 $h \cdot d^{-1}$.

8.5.4.2 Ground Occupancy Factor, Og (unitless)

This factor represents the annual average fraction of the time that man is outdoors. It is used to calculate doses from ground exposure (Equation (8.37)), and from inhalation of and immersion in outdoor air (Equations (8.25) and (8.35)). For the postclosure assessment, we have adopted an Og value of 0.2 (CSA 1987), which corresponds to 4.8 $h \cdot d^{-1}$.

8.5.4.3 Building Occupancy Factor, Ob (unitless)

The building occupancy factor is the average fraction of time that man spends indoors. It is used to calculate doses from building materials (Equation (8.38)), and from inhalation and air immersion where indoor air concentrations are involved (Equations (8.25) and (8.35)). Given an Og value of 0.2, the Ob value in BIOTRAC is fixed at 0.8 (CSA 1987), which corresponds to 19.2 $h \cdot d^{-1}$.

8.5.5 Ingestion Rates for Terrestrial Animals

Food or forage, drinking water and soil ingestion rates for terrestrial animals are needed to predict doses to man from consuming TE MILK, TE MEAT and TE BIRD. These ingestion rates can be quite variable, depending, among other factors, on body size and the productivity of the animal involved. For this reason, and because ingestion rates can have a strong effect on doses (Zach 1980b), they are treated probabilistically in CALDOS. The forage, water and soil ingestion rates are correlated to account for the observed interrelationships among them. Although the rates are expressed on a daily basis, they represent annual average values.

8.5.5.1 Feed or Forage Ingestion Rate, Qf_j (kg wet biomass$\cdot d^{-1}$)

This parameter defines the daily feed or forage ingestion rates of dairy cows, beef cattle and poultry. Qf_j values are used in Equations (8.6) and (8.12) to calculate man's intake of nuclides resulting from ingestion of TE MILK, TE MEAT and TE BIRD. They are also used in Equation (9.4) to estimate the area of the forage field needed to sustain the livestock raised by the critical group. Ingestion rates have been determined in numerous studies (Zach and Sheppard 1992). Generally speaking, dairy cows consume slightly more feed or forage than beef cattle, and considerably more than goats, sheep or pigs. Chickens consume much less feed daily. The available information suggests that Qf_j values are distributed normally.

For our assessment, we have assumed normal distributions of Qf_j values for all the food types with arithmetic means of 60.0, 50.0 and 0.4 kg wet biomass$\cdot$d^{-1} for TE MILK, TE MEAT and TE BIRD respectively. The corresponding SDs are 15.0, 12.5 and 0.1 kg wet biomass$\cdot$d^{-1}. The lower tails of the distributions are truncated because animals need a certain minimum intake to survive. In all cases, the truncation value is three SDs below the mean, or 15.0, 12.5 and 0.1 kg wet biomass$\cdot$d^{-1} for TE MILK, TE MEAT and TE BIRD respectively. Qf_j values are correlated with both drinking water and soil ingestion rates with a correlation coefficient, r, of 0.75 in both cases.

For use in Equation (9.4), Qf_j values must be expressed in units of kg wet biomass$\cdot$a^{-1}. In these units, Qf_j values are normally distributed, with arithmetic means of 2.19×10^4, 1.83×10^4 and 146 kg wet biomass$\cdot$a^{-1}, SDs of 5.48×10^3, 4.56×10^3 and 36.5 kg wet biomass$\cdot$a^{-1}, and lower truncation values of 5.48×10^3, 4.56×10^3 and 36.5 kg wet biomass$\cdot$a^{-1}, for TE MILK, TE MEAT and TE BIRD respectively. The correlation coefficients with the other ingestion rates remain unaffected at 0.75.

8.5.5.2 Drinking Water Ingestion Rate, Qdw_j (m^3 water$\cdot$d^{-1})

This parameter represents the daily drinking water ingestion rates of dairy cows, beef cattle and poultry. Qdw_j values are used in Equation (8.14) to calculate man's nuclide intake from the ingestion of TE MILK, TE MEAT and TE BIRD. They are also used in Equation (9.8) to calculate the water demand by livestock.

Few data are available on water ingestion rates of farm animals. It is not clear in the data that do exist if water contained in ingested food or water released through oxidation of the food is taken into account. To our knowledge, there are no statistical studies investigating the distribution and variation of Qdw_j values.

For the postclosure assessment, we have been guided by the observations on feed and forage ingestion rates, and assumed that Qdw_j values are normally distributed for all food types. We have adopted arithmetic means of 0.060, 0.040 and 4×10^{-4} m^3 water$\cdot$d^{-1} for TE MILK, TE MEAT and TE BIRD respectively (Zach and Sheppard 1992). The corresponding SDs are 0.015, 0.010 and 1×10^{-4} m^3 water$\cdot$d^{-1}. As in the case of Qf_j, the distributions for drinking water ingestion rates need to be truncated to avoid unreasonably low values. The truncation values for TE MILK, TE MEAT and TE BIRD are 0.015, 0.010 and 1×10^{-4} m^3 water$\cdot$d^{-1} respectively, which correspond to three SDs below the mean. As noted in Section 8.5.5.1, the Qdw_j values are correlated with feed or forage and soil ingestion rates for terrestrial animals with r values of 0.75.

For use in Equation (9.8), Qdw_j values must be expressed in units of m^3 water·a^{-1}. In these units, Qdw_j values are normally distributed; arithmetic means are 21.9, 14.6 and 0.146 m^3 water·a^{-1}, SDs are 5.5, 3.7 and 0.037 m^3 water·a^{-1}, and lower truncation values are 5.5, 3.7 and 0.037 m^3 water·a^{-1}, for TE MILK, TE MEAT and TE BIRD respectively. The correlation coefficients with the other ingestion rates remain unaffected at 0.75.

8.5.5.3 Soil Ingestion Rate, Qs_j (kg dry soil·d^{-1})

The daily soil ingestion rates of dairy cows, beef cattle and chickens are quantified by Qs_j values. Qs_j values are used in Equation (8.16) to calculate man's intake of nuclides arising from the ingestion of TE MILK, TE MEAT and TE BIRD. Soil may be ingested involuntarily while grazing, with stored feed, or during inhalation, or it may be deliberately consumed to meet nutritional deficiencies.

Some information is available on soil ingestion rates, but the distribution and variation of Qs_j values has not been studied statistically. We have taken advantage of the strong correlation between soil and feed or forage ingestion rates in setting Qs_j values for BIOTRAC. We assume soil ingestion rates are normally distributed, with arithmetic means between 6 and 7% of the corresponding dry weight values for feed or forage ingestion (Section 8.5.5.1). If we assume that water makes up three quarters of the wet plant weight (Section 8.5.1.1), this leads to means of 1.0, 0.8 and 0.006 kg dry soil·d^{-1} for TE MILK, TE MEAT and TE BIRD respectively. We assume that the ratio of SD to mean is the same for Qs_j as for Qf_j, so that the SDs for Qs_j become 0.25, 0.2 and 0.0015 kg dry soil·d^{-1}. Truncation limits were set three SDs below the mean at 0.25, 0.2 and 0.0015 kg dry soil·d^{-1}. Qs_j values are correlated with Qf_j and Qdw_j with a correlation coefficient of 0.75.

8.5.6 Energy and Food Parameters for Man

Most of the parameters discussed in this section are required to calculate man's food ingestion, drinking water ingestion and inhalation rates (Section 8.3.4). Because these rates show little variability, most of the parameters are represented by single conservative values, rather than by PDFs. In many cases these values are food-type-specific (Section 8.1).

8.5.6.1 Man's Total Energy Need, En (kJ·a^{-1})

Man's total annual energy need is used to calculate his ingestion rates of the five food types (Equation (8.42)) and of drinking water (Equation (8.44)). Man's energy need is made up of a number of components, the most important of which are the energy required for vital body functions and the energy needed for physical activity. Many surveys, experiments and theoretical studies have supplied information on human energy requirements. ICRP reference man is assumed to expend 12 600 kJ·d^{-1} (ICRP 1975), and Canadian adult males consume between 5 300 and 14 100 kJ·d^{-1} (Nutrition Canada 1977). For BIOTRAC, we have adopted a conservatively high En value of 14 600 kJ·d^{-1}, the energy required by a 70.0-kg adult male doing heavy physical work (WHO 1973). This value corresponds to about 3 500 kcal·d^{-1}, or in the units required by Equations (8.42) and (8.44), about 5.33 x 10^{6} kJ·a^{-1}.

8.5.6.2 Nutrient Contents of Foods, Cym_j, Fym_j, Pym_j (g nutrient·kg^{-1} wet biomass)

Carbohydrate, Cym_j, fat, Fym_j, and protein, Pym_j, contents of the five food types are needed to calculate man's ingestion and inhalation rates (Equations (8.42), (8.44) and (8.45)). They are also used to determine Ycf_j values (Section 8.5.6.4), which describe the contribution of each food type to man's total energy need. Because variation in nutrient contents has little effect on doses, and because much of the variability can be accounted for by using food-type-specific values, single values can be used.

Our nutrient content values for the postclosure assessment are based on the extensive tabulations by Watt and Merrill (1963), which include most of the foods used by humans in the United States. From these data, we selected all the appropriate foods and classified them according to our five food types, taking into account current production and availability on the Canadian Shield. We then calculated arithmetic means to derive Cym_j, Fym_j and Pym_j values for our food types. The results are presented in Table 8-5.

TABLE 8-5

NUTRIENT AND WATER CONTENTS OF FOOD TYPES

Food Type	Nutrient Content (g·kg^{-1} wet biomass)			Water Content (m^{3}·kg^{-1} wet biomass) Ywc_j
	Carbohydrate Cym_j	Fat Fym_j	Protein Pym_j	
TE PLANT	168.8	25.6	49.4	7.44 x 10^{-4}
TE MILK	32.0	191.1	113.9	6.44 x 10^{-4}
TE MEAT	5.4	203.4	170.0	6.11 x 10^{-4}
TE BIRD	3.2	43.9	197.8	7.45 x 10^{-4}
FW FISH	0.0	62.6	177.6	7.51 x 10^{-4}

8.5.6.3 Nutrient Fuel Values, Cec, Fec, Pec (kJ·g^{-1} nutrient)

The fuel values for carbohydrate, fat and protein are required to calculate man's food ingestion rates, U_j (Equation (8.42)), and food-type energy weighting factors, Ycf_j (Section 8.5.6.4). Fuel values show little variability (Zach and Barnard 1985), and single values are appropriate for the postclosure assessment.

As we define them, fuel values relate to the energy available to humans to satisfy their energy need. Accordingly, they correspond to the Atwater System (Watt and Merrill 1963) and are equivalent to the difference between heats of combustion of food and of the corresponding feces and urine. It is customary to use average values of 16.3, 37.7 and 16.7 $kJ \cdot g^{-1}$ for carbohydrate, fat and protein respectively (Davidson et al. 1979, Guthrie 1983). We adopted these values for BIOTRAC.

8.5.6.4 Food Type Energy Weighting Factor, Ycf_j (unitless)

The weighting factor, Ycf_j, describes the contribution of food type j to man's total energy need in the course of an average year. Normalized Ycf_j values (energy fraction for food type j, $Ycfs_j$ (unitless)) are used to calculate man's ingestion rate of the five food types (Equation (8.42)). Diets vary considerably from individual to individual. Whereas most people eat a varied diet, some, such as vegetarians, eat one type of food to the exclusion of most others. We have therefore treated Ycf_j values probabilistically to allow for variously mixed and extreme diets.

Dietary information is readily available from the literature in the form of ingestion rates of various foods (Nutrition Canada 1977; Rupp 1979, 1980a, 1980b; Rupp et al. 1980; Yang and Nelson 1986). Zach and Sheppard (1992) used this information to derive Ycf_j values for the postclosure assessment. In the absence of firm evidence regarding the distribution of food ingestion rates, they assumed that Ycf_j values are distributed lognormally for all the food types. This reflects the fact that most people consume moderate, and some disproportionately large, amounts of some of our five food types (Section 8.1). Zach and Sheppard (1992) then chose suitable GMs and GSDs for the ingestion rates of each food type. Since the raw data were expressed in terms of mass ingested per year (kg $biomass \cdot a^{-1}$), the GMs were converted to energy ingested ($kJ \cdot a^{-1}$) using the data on nutrient contents and fuel values discussed in Section 8.5.6.2 and 8.5.6.3. The energy ingestion rates were then expressed as fractions of the total energy contained in the diet, En (Section 8.5.6.1), to yield GM values for Ycf_j. These values and the GSDs for each food type are listed in Table 8-6.

TABLE 8-6

FOOD-TYPE ENERGY FRACTION VALUES, Ycf_j

Food Type	Distribution Type	Geometric Mean	Geometric Standard Deviation
TE PLANT	lognormal	0.32	1.65
TE MILK	lognormal	0.36	1.35
TE MEAT	lognormal	0.26	1.65
TE BIRD	lognormal	0.05	1.65
FW FISH	lognormal	0.01	4.48

In each BIOTRAC simulation, a Ycf_j value for each food type is randomly sampled from its specified distribution. Scaled values, $Ycfs_j$, are then calculated from Equation (8.43) for use in Equation (8.42). Our studies have shown that the $Ycfs_j$ distribution for a given food type is not significantly different than the corresponding Ycf_j distribution. Thus, scaling has little effect on dose prediction.

8.5.6.5 Water Content of Food, Ywc_j (m^3 water·kg^{-1} wet biomass)

Food-water contents are used to calculate man's drinking water ingestion rate (Equation (8.44)). Our Ywc_j values are not distributed, but we have allowed for the variability in food-water content by assigning different values to our five different food types. We derived our Ywc_j values from the same data source (Watt and Merrill 1963) that we used for our nutrient contents (Section 8.5.6.2). Accordingly, our values, which are listed in Table 8-5, are arithmetic means for foods that are currently produced and available on the Canadian Shield.

8.5.6.6 Metabolic Water Yields, Cmw, Fmw, Pmw (m^3 water·g^{-1} nutrient)

Metabolic water yields from the oxidation of ingested carbohydrate, Cmw, fat, Fmw, and protein, Pmw, are used to calculate man's drinking water ingestion rate (Equation (8.44)). Metabolic water yields can be assumed to be constants (Zach and Barnard 1985). For the postclosure assessment, we set Cmw = 6.0×10^{-7} m^3 water·g^{-1} carbohydrate, Fmw = 1.07×10^{-6} m^3 water·g^{-1} fat and Pmw = 4.20×10^{-7} m^3 water·g^{-1} protein (Davidson et al. 1979, Guthrie 1983).

8.5.6.7 Oxygen Combustion Values, Co, Fo, Po (m^3 O_2·g^{-1} nutrient)

These parameters define the volumes of oxygen (O_2) required at STP to oxidize one gram of carbohydrate, Co, fat, Fo, and protein, Po. They are used in Equation (8.45) to calculate man's inhalation rate. Oxygen combustion values can be assumed to be constants, and we adopted the commonly accepted values of 0.82, 2.03 and 0.97 L·g^{-1} for Co, Fo and Po respectively (Schmidt-Nielsen 1979). In units of m^3 O_2·g^{-1}, as required in Equation (8.45), these values become 8.2×10^{-4}, 2.03×10^{-3} and 9.7×10^{-4} m^3 O_2·g^{-1} carbohydrate, fat and protein.

8.5.6.8 Soil Ingestion Rate from Hands, Hs (kg dry soil·a^{-1})

The annual average rate at which man involuntarily ingests soil through oral contact with hands and other objects is given by Hs. It is used in Equation (8.23) to help calculate the dose from soil ingestion. The data on Hs are limited, but work by Hawley (1985) and LaGoy (1987) suggests that adult soil ingestion rates through hand-to-mouth transfer range from about 1 to 60 mg dry soil·d^{-1}, with a maximum of 100 mg dry soil·d^{-1}. We have adopted a single value of 100 mg dry soil·d^{-1}, which translates to about 0.04 kg dry soil·a^{-1}. This value is likely very conservative because it does not consider reduced soil intake in the winter when the ground is frozen and snow-covered.

8.5.6.9 Soil Contamination of TE PLANT, Ps (kg dry soil·kg^{-1} wet biomass)

Ps is the mass of soil adhering to a unit mass of man's plant foods, TE PLANT, after normal food processing and preparation. It is used in Equation (8.23) to help calculate man's dose from soil ingestion. Ps accounts for the transfer of soil to plants through local suspension mechanisms such as rainsplash and mechanical harvesting, and for soil particles adhering to root crops. The mass of soil on plants prior to food processing and preparation is typically 2.5 mg dry soil·g^{-1} wet biomass, although this can vary considerably depending on the crop type and the proximity of the crop to the ground (Pinder and McLeod 1989). If we assume that 80% of the soil is removed during food processing and preparation, the typical Ps value becomes 0.5 mg·g^{-1}, or 5×10^{-4} kg dry soil·kg^{-1} wet biomass. We have adopted this value for BIOTRAC, which, given an average ingestion rate of 375.7 kg wet biomass·a^{-1} for TE PLANT (Table 8-1), results in a soil ingestion rate of about 0.2 kg dry soil·a^{-1}.

8.5.7 Plant Interception Fractions

The plant interception fraction defines the proportion of the nuclide flux from atmospheric deposition or irrigation that is initially retained on exposed plant parts. Interception fractions are assumed to have the same value for both wet and dry atmospheric deposition (Section 7.5.4), but different values for atmospheric deposition and irrigation. They are used to predict internal and external doses to man from leaf deposition. Interception fractions show relatively little variation and are treated as fixed parameters in BIOTRAC.

8.5.7.1 Plant Interception Fraction for Food Types, r_j (unitless)

This parameter defines the fraction of aerially deposited nuclides initially retained on exposed plant parts consumed by humans (Equation (8.8)) or by animals (Equation (8.12)). The values of r_j depend on a number of factors including particle size, foliage structure, and meteorological conditions. In addition, r_j values increase with vegetation density over the course of the growing season, although it is customary to use the high values that occur at or near harvest.

A review by Miller (1980) showed that interception-fraction values for atmospheric deposition can range from 0.02 to 1.0. However, these values may be constrained by the experimental methods used to measure the dry deposition velocity, Vd, which is employed in predicting the deposition flux (Section 7.3.8). Reported Vd values are frequently based on the fraction of material intercepted by the plants alone, without regard for material that reaches the underlying soil. Use of such Vd values precludes r_j values of less than one (Miller 1980). Accordingly, for the postclosure assessment, we have used an r_j value of 1.0 for all the food types for atmospheric deposition, which is conservative.

Interception fractions for aerial irrigation are not subject to this restriction because deposition fluxes in this case are not based on deposition velocities. Few r_j values for irrigation have been reported. USNRC

(1977) recommended a value of 0.25, but this appears to be overly conservative because most irrigation water reaches the soil. We have adopted a value of 0.05 for all the food types, a value also recommended by CSA (1987).

8.5.7.2 Plant Interception Fraction for Wood, rb (unitless)

This parameter represents the fraction of aerially deposited nuclides initially retained on exposed parts of trees used for wooden building materials (Equation (8.40)). A value is required only for atmospheric deposition, since the woodlot of the critical group is not irrigated. In all other respects, rb is analogous to r_j (Section 3.5.7.1), and we adopted a conservative value of 1.0.

8.5.8 Plant Yields

Plant yield defines the mass of a crop harvested or growing per unit area. It is usually defined in terms of usable product, such as hay, lettuce, carrots or potato tubers, so that it does not necessarily relate directly to the plant parts exposed to deposition (Section 8.5.7). We assume conservatively that all the deposited nuclides retained by exposed plant parts are translocated to the usable product defined by the yield.

Plant yield and interception fraction tend to be positively correlated because increased yield implies a denser vegetation cover and less exposed bare ground (Chamberlain 1970, Miller 1980). However, there is no need to correlate yield and interception fraction in BIOTRAC because r_j and rb have been set to their maximum values (Section 8.5.7).

Plant yield is used to predict ingestion doses to man from leaf deposition and from exposure to wooden building materials. For agricultural crops, yield can vary substantially and is treated as a distributed parameter in our assessment. Yield values for wood are less variable and have a lower impact on doses; they have been assigned a fixed value.

8.5.8.1 Plant Yield for Food Types, Y_j (kg wet biomass·m^{-2} soil)

This parameter refers to the wet weight of plant crops harvested per unit area for consumption by man or animals (Equations (8.8) and (8.12)). Yield varies greatly because of plant and soil type, climate, season and agricultural practice. Measures of crop productivity, such as standing-crop biomass, are sometimes substituted for yield data, but the two quantities can be quite different.

Yield values have been reviewed in a number of studies (Koranda 1965, Whittaker 1970, Baes and Orton 1979), including one based on Ontario data (OEPCB 1987). Typically, yields vary between 0.1 and 10 kg wet biomass·m^{-2}, with a mean of about 1.0 kg·m^{-2}. In Ontario, yields tend to be lower in the north than in the south, but usually by less than a factor of two. The OEPCB (1987) values are based on commercial data, but it is likely that well-managed household gardens achieve similar or even higher yields.

Statistical information on distributions and variations of yield values is limited, but suggests that the distributions are normal and the SDs large (McGee 1988).

Zach and Sheppard (1992) reviewed the available data and specified normal distributions for all Y_j values, with means of 0.8, 0.8, 1.0 and 1.2 kg wet biomass·m^{-2} soil for TE PLANT, TE MILK, TE MEAT and TE BIRD respectively. The corresponding SDs are 0.27, 0.27, 0.33 and 0.40 kg wet biomass·m^{-2} soil. These distributions emphasize low yield values, which are conservative. To avoid unreasonably low yields - so low they might be insufficient to support the critical group - all the distributions have been truncated at their low ends. The truncation values were set at the 99th percentile, or at 0.10, 0.10, 0.15 and 0.17 kg wet biomass·m^{-2} soil for TE PLANT, TE MILK, TE MEAT and TE BIRD respectively.

Yield values are also used in the atmosphere model to determine the nuclide flux to the atmosphere as a result of agricultural fires (Section 7.3.5.1). In that context, yield refers to the biomass burned per unit area, and so requires an estimate of the plant mass left on the field after harvest. In contrast, yield as defined here refers to the mass harvested per unit area. However, about half of the total crop biomass is usually harvested so that the same values can be used for both applications.

8.5.8.2 Plant Yield for Wood, Yb (kg wet biomass·m^{-2} land)

The wet weight of trees harvested per unit area for use as organic building materials (Equation (8.40)) is given by Yb. Timber yield has been expressed in a variety of ways. For BIOTRAC, the most relevant measure of yield is gross total wet volume, which most closely reflects standing crop biomass and includes all woody parts of trees. Depending on tree species and location, the gross total volume at harvest ranges from about 100 to 400 m^3 wet biomass·ha^{-1} land. Given that low Yb values are conservative, 150 m^3 wet biomass·ha^{-1} land is a suitable value. If we assume an average density of 0.7 kg wet biomass·dm^{-3} wood (Panshin and de Zeeuw 1980), this corresponds to a yield of 10.5 kg wet biomass·m^{-2} land. In view of the limited variability in wood yield, we used this fixed value for the postclosure assessment.

A forest yield, FY, is used to calculate the nuclide flux to the atmosphere from forest or land-clearing fires in the atmosphere model, where it is assigned a lognormal distribution with a GM of 2.2 kg wet biomass·m^{-2} land (Section 7.5.2.4). The numerical values of FY and Yb differ because they represent different processes. FY refers to the biomass burned in a fire, whereas Yb is the weight of trees harvested for use as building materials. Because only a small fraction of the forest mass is normally consumed in a fire, FY is substantially less than Yb.

8.5.9 Parameters for ^{129}I Model

The parameters required for our limited specific-activity model for ^{129}I (Section 8.3.1.11) are discussed in this section. We have assigned fixed values to these parameters because they show little variability and because they relate to the dosimetry of ICRP reference man (Section 1.5.4).

8.5.9.1 Total Intake of Stable Iodine, E^{Is} ($kg\ ^{127}I \cdot a^{-1}$)

This parameter represents man's total annual intake of stable ^{127}I from ingestion and inhalation (Equation (8.27)). Iodine is an essential micronutrient for humans, who require an intake of between 50 and 1000 $\mu g \cdot d^{-1}$ to avoid metabolic disorders. Recommended dietary allowances of iodine for adults are 150 $\mu g \cdot d^{-1}$ in the United States (NAS 1980) and 160 $\mu g \cdot d^{-1}$ in Canada (Nutrition Canada 1977). In the past, iodine intake depended upon natural sources in the biosphere, and so was often relatively low and variable. More recently, with the introduction of iodized salt to the human diet and iodine supplements for livestock, intake has increased and become more uniform. Most Americans take in at least 300 $\mu g \cdot d^{-1}$ (Guthrie 1983), and the average Canadian intake may be as high as 1050 $\mu g \cdot d^{-1}$ (Fisher 1986). For our assessment, we set E^{Is} to 200 $\mu g \cdot d^{-1}$, or 7.3 x 10^{-5} kg $^{127}I \cdot a^{-1}$, a conservatively low value that corresponds to the iodine ingestion rate of reference man (ICRP 1975).

8.5.9.2 Iodine Content of the Thyroid Gland, Thi (kg iodine)

The total iodine burden of the thyroid gland is given by Thi (Equation (8.28)). The adult human body contains about 15 to 23 mg of iodine, of which 70 to 80% resides in the thyroid (Nutrition Canada 1977, Guthrie 1983). For BIOTRAC, we gave Thi a value of 12 mg iodine, or 1.2 x 10^{-5} kg iodine, which corresponds to the thyroid burden of reference man (ICRP 1975).

8.5.9.3 Mass of Thyroid Gland, Thm (kg thyroid)

The adult thyroid gland weighs about 25 g (Guthrie 1983), although this can vary considerably (Dunning and Schwartz 1981). The value for reference man is 20 g (ICRP 1975). We have conservatively adopted this lower value in Equation (8.28) so that Thm has a value of 2.0 x 10^{-2} kg thyroid.

8.5.9.4 Mass/Activity Conversion Factor for ^{129}I, gb ($kg\ iodine \cdot Bq^{-1}$)

As noted in Section 8.3, the activity per mole of radionuclide i is given by $N_A \cdot \lambda^i$ ($Bq \cdot mol^{-1}$), where N_A = 6.02 x 10^{23} atoms$\cdot mol^{-1}$ is Avogradro's number and λ^i is the radioactive decay constant (s^{-1}). The activity per unit mass is then $N_A \cdot \lambda^i / Mw^i$ ($Bq \cdot kg^{-1}$), where Mw^i is the molecular weight and equal to 0.129 $kg \cdot mol^{-1}$ for ^{129}I. The value of gb is then calculated from

$$gb = Mw^i / (N_A \cdot \lambda^i) \qquad (8.46)$$

which yields 1.53 x 10^{-10} kg iodine$\cdot Bq^{-1}$ for use in Equations (8.27), (8.28) and (8.31).

8.5.9.5 Stable Iodine Concentration in Groundwater, C^{sI}_{gw} (kg $^{127}I \cdot m^{-3}$ water)

This concentration is required in Equation (8.31) to limit the internal dose from ^{129}I. C^{sI}_{gw} is based on observed iodide (I^-) values from the WRA, and we assume a uniform distribution ranging from 5×10^{-6} to 2×10^{-5} kg $^{127}I \cdot m^{-3}$ water (Gascoyne and Kamineni 1992).

8.5.10 Parameters for ^{14}C Dose Limit for Man

The parameters required to calculate the upper limit to the ^{14}C internal dose (Section 8.3.1.12) have been assigned fixed values because they show little variability and because they relate to the dosimetry of ICRP reference man (Section 1.5.4).

8.5.10.1 Mass of Soft Tissue, Bs (kg soft tissue)

ICRP reference man has a total soft tissue mass of 63.0 kg (ICRP 1975). Of this, red and yellow bone marrow contribute 3.0 kg, with the remaining 60 kg being made up by various soft tissues throughout the body. We set Bs equal to 63.0 kg soft tissue in Equation (8.33) of BIOTRAC.

8.5.10.2 Carbon Content of Soft Tissue, Bc (kg carbon)

The amounts of carbon in soft tissue, red marrow and yellow marrow in the body of ICRP reference man are 14.0, 0.62 and 0.95 kg respectively (ICRP 1975). The sum of these values, 15.6 kg carbon, was used as the value for Bc in Equation (8.33).

8.5.10.3 Mass/Activity Conversion Factor for ^{14}C, gc (kg carbon$\cdot$Bq^{-1})

The conversion factor, gc, can be found from Equation (8.46), with gc replacing gb. With $Mw^c = 0.014$ kg$\cdot$mol^{-1} for ^{14}C, and $\lambda^c = 3.83 \times 10^{-12}$ s^{-1}, gc has a value of 6.07×10^{-15} kg carbon$\cdot$Bq^{-1} in Equations (8.32) and (8.33).

8.5.10.4 Stable Carbon Concentration in Groundwater, C^{sC}_{gw} (kg $^{12}C \cdot m^{-3}$ water)

This concentration is required in Equation (8.32) to limit the internal dose from ^{14}C. C^{sC}_{gw} is based on observed bicarbonate anion concentrations (HCO_3^-) from the WRA, and we assume a triangular PDF ranging from 2.0×10^{-2} to 6.8×10^{-2} kg $^{12}C \cdot m^{-3}$ water, with a peak at 4.0×10^{-2} kg $^{12}C \cdot m^{-3}$ water (Gascoyne 1992). This information is sufficient for specifying triangular distributions in SYVAC3.

8.5.11 Parameters for Tritium Model

The specific-activity model for tritium (Section 8.3.1.13) involves hydrogen concentrations in domestic water and in man. Neither of these parameters is particularly variable, and both are represented by single values in BIOTRAC.

8.5.11.1 Concentration of Hydrogen in Water, C_w^H (g hydrogen$\cdot$m^{-3} water)

Water is 11.1% hydrogen by mass. If we assume a density of 1.0 kg$\cdot$L^{-1}, the concentration of hydrogen in water is 111 g$\cdot$L^{-1} or 1.11×10^5 g hydrogen$\cdot$m^{-3} water. This value is required in Equation (8.34).

8.5.11.2 Concentration of Hydrogen in Man, MC^H (g hydrogen$\cdot$kg^{-1} soft tissue)

If we assume hydrogen contents of 6% for carbohydrate, 12% for fat, 7% for protein and 11% for water, ICRP reference man is about 10% hydrogen by mass. Disregarding the skeleton and teeth, which contain relatively little hydrogen, the hydrogen content of the soft tissues is about 10.5%. For the postclosure assessment, we have chosen a conservatively high value of 12% so that MC^H is 120 g hydrogen$\cdot$kg^{-1} soft tissue in Equation (8.34).

8.5.12 Conversion and Efficiency Factors

In this section, values for miscellaneous factors appearing in CALDOS are defined. None of these factors needs to be treated probabilistically in BIOTRAC because of their limited variation and influence on dose predictions.

8.5.12.1 Man's Water/Energy Conversion Ratio, ewc (m^3 water$\cdot$kJ^{-1})

This parameter defines man's water need per unit energy need, and is used to calculate man's drinking water ingestion rate (Equation (8.44)). The commonly accepted value of ewc is 2.39×10^{-4} L water$\cdot$kJ^{-1} (NAS 1980, Guthrie 1983). We have adopted a slightly higher and more conservative value of 2.5×10^{-4} L water$\cdot$kJ^{-1}, or 2.5×10^{-7} m^3 water$\cdot$kJ^{-1}.

8.5.12.2 Dry/Wet Soil Conversion Factor, dws (kg dry soil$\cdot$kg^{-1} wet soil)

This parameter is used in Equation (8.37) to convert nuclide concentrations in dry soil to concentrations in wet soil for consistency with the DCFs used to calculate man's doses from exposure to contaminated soil (Section 8.5.2.2).

The water content of soil is best defined in terms of field capacity, the maximum amount of water remaining in a soil under free drainage (Buckman and Brady 1969). We assumed a dry soil with a 5% water content, as one might encounter on an unpaved road or parking lot. This translates into a dws value of 0.95 kg dry soil$\cdot$kg^{-1} wet soil, which is conservative for all the soil types considered in our assessment (Section 6.5.1.1).

8.5.12.3 Wet/Dry Wood Conversion Factor, wdw (kg wet biomass$\cdot$kg^{-1} dry wood)

Nuclide concentrations in wood are predicted on a green or wet weight basis (Equation (8.40)); however, much of the water is removed from the wood before construction to avoid excessive shrinkage. The factor wdw converts concentrations to a dry weight basis for use in calculating doses from exposure to building materials (Section 8.3.2.4). The water content of wood when dry reflects the humidity in the surrounding atmosphere, and is typically 12 to 15% (Panshin and de Zeeuw 1980, Bramhall 1981). For the important commercial tree species on the Canadian Shield, the water loss from wet to a 12% equilibrium water content ranges from about 16.7 to 39.5% (Panshin and de Zeeuw 1980). However, this value may be an underestimate if entire trees rather than wood alone are considered (Young et al. 1964). We have assumed a conservative value of 40%, which corresponds to a wdw value of 1.7 kg wet biomass·kg^{-1} dry wood.

8.5.12.4 Inorganic Building Material/Soil Conversion Factor, sbc (unitless)

The factor sbc allows for the concentration or dilution of nuclides when soil materials are processed into inorganic building materials (Equation (8.39)). Dilution is likely in the case of sand and gravel because nuclides tend to be associated with small clay particles and organic matter (Megumi 1979, Sheppard M.I. et al. 1984a) that may be washed out in the preparation of these building materials. Nuclides may be concentrated during the processing of clay, which has a high sorptive capacity (Section 6.5.3). However, sbc is likely not very variable, and we assumed neither dilution nor concentration by using a value of 1.0 regardless of soil type.

8.5.12.5 Man's Air/Oxygen Conversion Factor, oac (unitless)

The factor oac defines the air volume per unit oxygen (O_2) volume for calculating man's inhalation rate (Equation (8.45)). Since oac shows little variation, we adopted a value of 4.78, which corresponds to the average oxygen content of air of 20.94% by volume (Neiburger et al. 1973).

8.5.12.6 Man's Oxygen Utilization Factor, ov (unitless)

The factor ov accounts for the fact that humans cannot use all the oxygen they inhale (Lloyd 1976, Guyton 1981). The factor is the reciprocal of the fraction of the oxygen content of air available to humans for respiration, and is used in Equation (8.45) to calculate man's inhalation rate. Humans are able to use from about 12 to 30% of the inhaled oxygen. In view of this limited variation, we have adopted a conservative value of 16%, which corresponds to an ov value of 6.25.

8.6 MODEL VALIDATION

To validate CALDOS one should show that the processes involved in nuclide transfer through the food chain are adequately simulated, and that the predicted concentrations in plants and animals, and the predicted doses to humans, are realistic. It is very difficult to demonstrate this experimentally or with field data. There have been many studies done on the transfer of nuclides to plants and animals, but studies involving humans are much rarer. The deliberate contamination of human subjects in tracer experiments is generally considered unacceptable, and the measurement of nuclide concentrations in the human body is difficult. Experiments rarely test the dosimetry models because human exposure to radionuclides has resulted in no detectable health effects in all but a few accidental or special situations. Moreover, results from almost all food-chain transfer studies have been incorporated into the general database that is now commonly used to derive food-chain models and their parameter values. There are, therefore, few independent data available for model validation. Finally, where new information becomes available, it is often in an inappropriate form. For example, studies conducted following the Chernobyl accident provide data for a strongly time-dependent aerial source, which does not match the chronic, steady-state conditions assumed by CALDOS.

For these reasons, the only part of CALDOS that has been validated is the EWAM model, which predicts ingestion and inhalation rates in agreement with observed values (Zach and Barnard 1987). However, confidence in CALDOS can be gained through other means, including scientific consensus, peer review, and model intercomparisons. In particular, CALDOS has undergone a rigorous model evaluation methodology developed specifically for environmental assessment models (Shaeffer 1980). These and other approaches to model validation are discussed in Section 8.7.2 and Chapter 11.

8.7 MODEL DISCUSSION

8.7.1 Assumptions

A number of assumptions were made in deriving CALDOS. In this section, we restate, explain and review the main assumptions and discuss their effects on the predictions of the model.

1. *The transfer of nuclides into and through the food chain is assumed to be linear.* This means that nuclide concentrations in a given compartment are a linear function of those in the donor compartments, which implies that vault-derived nuclides are present in the biosphere in trace quantities only, an assumption that must be valid if the concept of geological disposal is to be accepted. Radiological and chemical toxic effects would interrupt normal food-chain transfer at high nuclide concentrations. The assumption of linearity is therefore conservative because it would allow for transfer even at unrealistically high contamination levels. In CALDOS, the implications of high nuclide concentrations would be reflected both in doses to man, and in effects on plants and animals (Chapter 13).

2. *The complex processes responsible for nuclide transfer into and through the food chain can be described using simple transfer coefficients.* Most transfer processes are complex, involving physical, chemical and biological aspects. For example, root uptake depends on a variety of plant, soil and other environmental factors (Zach and Sheppard 1992). Details of the various transfer processes and their interactions are not always fully understood; similarly, the data required for detailed modelling are often not readily obtainable (Suter et al. 1985). For these reasons it is usually impractical to simulate each process individually, and the attempt to do so can exaggerate uncertainties. On the other hand, transfer coefficients are simple, highly aggregated empirical parameters that

describe the net effect of several processes. They can be readily measured in field and laboratory studies. Their use in long-term assessment applications is therefore reasonable, and most conventional food-chain and dose models rely on them (USNRC 1977, 1983b, IAEA 1982, NCRP 1984, CSA 1987).

3. *CALDOS assumes steady-state conditions.* Time scales in the food chain, which are typically less than one year, are much shorter than the time scales associated with changes in the nuclide flow rate out of the geosphere (Davison et al. 1994b, Goodwin et al. 1994). Concentrations in the source compartments for the food chain (surface water, soil and air) can therefore be considered constant. Steady-state models are entirely appropriate for such situations.

   CALDOS also assumes steady-state conditions over the course of a year or a growing season since parameter values are not allowed to vary with time. Although agricultural parameters show strong seasonal variations, they have little effect on dose predictions. For example, although the plant/soil concentration ratio, Bv^i, varies over the course of the growing season, the relevant plant concentrations can be estimated reliably from the Bv^i value at the time of harvest. As a bulk transfer coefficient, Bv^i takes account of the temporal variations in root uptake, as well as the various physical, chemical and biological processes involved. By using appropriate fixed values for the transfer coefficients in each BIOTRAC simulation, we avoid the complexity and data availability problems of time-dependent models. Moreover, most parameters exhibit values at some point in their seasonal cycle that maximize concentrations and doses. We used such values wherever possible. For example, we have used the conservatively high values that occur at harvest for our plant interception fractions (Section 8.5.7).

   A further consequence of our steady-state assumption is that when irrigation is practised (Section 6.3.7.2), deposition to leaves occurs continuously and uniformly throughout the growing season, when in fact it occurs only sporadically. However, home owners tend to water their gardens fairly frequently so that the assumption of continuous irrigation is not far from reality. In addition, the higher deposition rates associated with more intense periodic irrigation would be offset by shorter exposure times (Section 8.5.3.2), and the plant concentration after a number of periodic irrigation events would be similar to the concentration that results from continuous deposition.

   Our neglect of seasonal effects is conservative in a number of other ways. For example, man's soil ingestion rate, Hs, which was derived from summer measurements (Section 8.5.6.8), is assumed to hold in winter when frozen ground and snow cover would tend to reduce soil intake. Similarly, no credit is taken for the shielding effects of snow cover in calculating external doses from contaminated soil (Section 8.5.2.2).

4. *The recycling of nuclides is not modelled in CALDOS; however, it is accounted for implicitly.* We allow almost all nuclides entering the food chain to return to their source compartments by not depleting the source inventories when the transfer first occurs. For example, we assume that only 5% of the nuclides taken up by plants is permanently lost from the soil (Section 6.3.4); the remaining 95% is effectively returned, accounting for recycling through waste products and decay of the organisms involved. Similarly, nuclide inventories in water are not depleted when water is removed for domestic purposes. In this way we model the recycling of the nuclides implicitly when the water is discarded. The implicit treatment of recycling is appropriate because nuclide transfer processes in the food chain are rapid compared to changes in soil and surface water concentrations. However, it implies that recycling does not redistribute nuclides among the source compartments.

5. *Doses are calculated for ICRP reference man.* For nuclear fuel waste disposal, where exposure of the public may occur from infancy to old age, it might be appropriate to calculate doses using a model that explicitly incorporates age-dependent effects (Section 1.2.3). However, the DCFs for reference man account for both sexes and all age groups because they include organ weighting factors based primarily on population data from the Japanese bomb survivors. Furthermore, calculations by Zach and Mayoh (1984) suggest that infant doses are on average similar to those predicted for reference man for the nuclides of interest in nuclear fuel wastes. This is also confirmed by the postclosure assessment results (Grondin et al. 1994). The DCFs for reference man should therefore provide reasonable estimates of doses to members of the critical group. Models such as CALDOS can readily incorporate age-dependent DCFs once such values become more widely available.

6. *Transfer factors, F^i_j, for meat are based largely on beef, which tends to show slightly lower values than pork, lamb or wild game (Section 8.5.1.2).* This has little influence on doses because pigs, sheep and game animals take in less nuclides with feed, water and soil than do cattle. Moreover, F^i_j values are represented by broad PDFs and so include many animals other than cattle.

7. *When calculating the ingrowth of daughter radionuclides with half-lives between 1 d and 20 a, we assume the daughters are in secular equilibrium with their precursors (Section 8.3.1).* A time-dependent model might handle ingrowth more realistically, but is not appropriate for the postclosure assessment for the reasons given in point (3) above. The secular equilibrium approach is conservative, but not overly so because most of the daughters to which it applies are short-lived (Table 2-1), and in fact reach secular equilibrium in the organism of concern.

8. *CALDOS is a comprehensive model, but it does not treat all possible exposure pathways explicitly.* A number of pathways were omitted because they could be shown to make an insignificant contribution to man's

total dose (Appendix C; Goodwin et al., in preparation; Zach and Sheppard 1992). For example, we assume that inhalation of air by animals contributes little to the nuclide burden in the animals and the dose to man (Zach 1985b). Other pathways omitted for this reason include biotic transport of nuclides (dispersion by various organisms), external exposure from standing on contaminated ice, and (apart from tritium) absorption of nuclides through the skin from the air or from the application of contaminated cosmetics or medical products.

We have not explicitly modelled other pathways because they have been included implicitly. The contamination of plant leaves through rainsplash or mechanical harvesting is assumed to be included in animal's and man's soil ingestion pathways (Section 8.3.1.6 and 8.3.1.9). We assume that soil ingestion also accounts for nuclides ingested by animals during grooming. The direct uptake of nuclides from groundwater by phreatophytes or riparian vegetation is assumed to be accounted for by using shallow soil concentrations as the basis for predicting plant concentrations in some BIOTRAC simulations (Section 6.3.5). Other exposure pathways such as smoking cigarettes made from contaminated tobacco or applying contaminated fertilizer or herbicides to agricultural fields have been ignored because the chemical hazards of these practices far exceed the radiological dangers.

Minor exposure pathways such as these have attracted little attention, and few data are available for now to model them. It is appropriate to ignore them because the uncertainties that would arise in attempting to model them would exceed the potential effects on concentrations and dose predictions. We assume that our generally conservative approach accounts for contributions from these minor pathways.

9. *Long-term changes in the environment, and in human cultural practices, are not modelled.* CALDOS addresses humans with present-day characteristics subsisting on foods produced in conventional ways. Although food production methods will likely undergo significant advances in the future, they will probably continue to rely on the same fundamental biological principles in effect today (Goodwin et al., in preparation). Change in the values of parameters such as yield, nutrient content and transfer factors (Section 8.5) are covered, at least to some extent, by our parameter distributions. Human evolutionary history suggests that physiology or metabolism is unlikely to change significantly over the assessment period (Leaky and Lewin 1977). It is therefore reasonable to assume that CALDOS adequately describes food-chain transfer and doses to man in the future as well as under present-day conditions.

8.7.2 Evaluation

Concern for environmental and human safety has prompted numerous studies on the transfer of nuclides through food chains. These studies have led to the development of many models to assess the impacts of radionuclide releases from conventional power installations. These models have gained scientific and regulatory acceptance, and have formed the basis for the food-chain and dose model used for the assessment of our disposal concept. CALDOS is a simple, multiplicative chain model that assumes that nuclide concentrations in living organisms are directly proportional to the concentrations in the physical compartments of the biosphere. It includes all the commonly recognized exposure pathways; it also treats many additional minor pathways to ensure that all the significant contributions to man's dose are accounted for. The use of conservative assumptions and conservative parameter values further guarantees that doses are not underestimated. Although CALDOS has not been experimentally validated, it has undergone a rigorous model evaluation methodology (Zach and Sheppard 1992) developed specifically for environmental assessment models (Shaeffer 1980). CALDOS has received peer review through publication in the open literature (Zach and Sheppard 1991), as has EWAM, the model used to calculate man's ingestion and inhalation rates (Zach and Barnard 1987). CALDOS is consistent with other models used internationally to model food-chain transfer for waste management applications (Zach and Sheppard 1992).

CALDOS was specifically designed for the postclosure assessment of the concept for disposal of Canada's nuclear fuel waste. It was formulated to predict doses to members of the critical group residing at a generic discharge zone on the Canadian Shield. It was also formulated to predict nuclide concentrations in generic plants and animals. Its simplified, efficient structure, together with its distributed parameter values, make it suitable for a probabilistic assessment. It interfaces smoothly with the three other submodels of BIOTRAC and our model for predicting doses to non-human biota (Section 13.3).

Generally accepted data and recommended parameter values and distributions are available for most of the parameters appearing in CALDOS. We have screened the information and selected values relevant to the Canadian Shield biosphere and the exposure situation of the critical group. Our own work has provided values for many parameters (e.g., Johnson J.R. and Dunford 1983; Sheppard M.I. and Thibault 1983; Sheppard et al. 1984b, Holford 1988, 1989; Zach et al. 1989; Sheppard S.C. and Evenden 1990). The database is fairly extensive for most parameters; where information is lacking, we have used analogies or expert opinion to set values. We selected what we believe are conservative values for all the parameters.

We conclude that CALDOS and its associated parameter values provide an adequate description of food-chain transfer under Canadian Shield conditions, and that they provide conservative estimates of concentrations and doses when used to assess the concept for disposal of Canada's nuclear fuel waste.

9. THE INTEGRATED BIOSPHERE MODEL

The four biosphere submodels have been described individually in the four preceding chapters. In this chapter we will discuss how the submodels are linked to provide a cohesive description of nuclide transport through the biosphere as a whole. We will discuss parameters that define some aspects of the cultural behaviour of the critical group. These are parameters that are required in BIOTRAC, but do not appear explicitly in any of the four submodels. We will also discuss the water balance parameters. Precipitation, runoff and evapotranspiration are used in three different submodels (Chapters 5, 6 and 7), and must be treated in a consistent way to preserve the relationship that exists among them. We will discuss the implementation of BIOTRAC within SYVAC3, and indicate how BIOTRAC is linked to the geosphere model and to its output. We present calculations to quantify the amount of apparent nuclide mass generated in BIOTRAC and estimate the effect on the predicted concentrations and doses. Finally, we compare BIOTRAC with the model used to assess the preclosure phase of the disposal concept.

9.1 BIOSPHERE PARAMETERS

9.1.1 Household and Herd Sizes, Field Areas and Water Demand

Information on the area of terrestrial contamination and on water demand by the critical group is needed in a number of places in BIOTRAC. Field areas are required to calculate atmospheric dispersion factors and to estimate nuclide concentrations in soil following groundwater discharge or the application of fresh sediment. Water demand is needed to compute indoor air concentrations and to set boundary conditions for the geosphere model, GEONET, when a well is chosen as the water source in BIOTRAC. Field areas and water demand depend on the number of people in the household, the number of animals they need, their food requirements and their habits of water use. Values for these parameters are calculated in a consistent manner in BIOTRAC, as discussed below.

9.1.1.1 Number of Persons per Household, Nph (p or unitless)

In BIOTRAC, the number of persons per household, Nph, living in the sequence of households that make up the critical group over time is treated as a sampled parameter (Zach and Sheppard 1992). Nph is used in the atmosphere model to calculate the release of nuclides from domestic water (Section 7.3.6.2). It is also used in Equations (9.3) and (9.7) to calculate the size of the garden and the domestic water demand. The PDF for Nph was constructed from recent Canadian census data (Smith M. 1987), and is assumed to represent the household size over the entire simulation period. In fact, the average family size in Canada has been steadily dropping, so that our PDF will likely overestimate the number of people in future households unless there is a reversal in family-size trends.

The original census data were expressed as a lognormal distribution with GM = 3.5 and GSD = 1.8. A piecewise uniform PDF was extracted from this distribution, with each segment representing an integral number of people.

The probabilities for each value of Nph are listed in Table 9-1. The higher values in this distribution account for large families and for cases where a group of families pools agricultural and water resources. The minimum allowed value of Nph is 1. The median-value simulation has an Nph value of 3, although we have sometimes used a value of 4 to demonstrate resource needs.

TABLE 9-1

PIECEWISE UNIFORM DISTRIBUTION DEFINING THE NUMBER OF PERSONS PER HOUSEHOLD, Nph

Nph	Probability	Nph	Probability
1	0.068	19	5.7×10^{-4}
2	0.217	20	4.1×10^{-4}
3	0.219	21	3.1×10^{-4}
4	0.165	22	2.3×10^{-4}
5	0.113	23	1.7×10^{-4}
6	0.074	24	1.3×10^{-4}
7	0.048	25	1.0×10^{-4}
8	0.031	26	7.8×10^{-5}
9	0.021	27	6.0×10^{-5}
10	0.014	28	4.6×10^{-5}
11	0.0092	29	3.6×10^{-5}
12	0.0062	30	2.8×10^{-5}
13	0.0043	31	2.2×10^{-5}
14	0.0030	32	1.8×10^{-5}
15	0.0021	33	1.4×10^{-5}
16	0.0015	34	1.1×10^{-5}
17	0.0011	35	9.0×10^{-6}
18	0.0008	36	7.3×10^{-6}

9.1.1.2 Number of Terrestrial Animals, Na_j (unitless)

The number of terrestrial animals, Na_j, raised by the critical group is calculated from the average yield of each animal, FY_j (kg wet biomass·a^{-1}) and the quantity of each consumed per person, U_j (kg wet biomass·a^{-1}). The number of animals is given by

$$Na_j = Nph \cdot U_j / FY_j \quad . \tag{9.1}$$

Here j = TE MEAT, TE MILK and TE BIRD. Thus, a separate value is calculated for beef cattle, dairy cows and poultry. The values are increased to the next largest integer to ensure that only whole numbers of animals are considered. Na_j is used in Equations (9.4) and (9.8) to calculate the size of the forage field and the water demand by livestock.

Values for U_j are calculated using the methods discussed in Section 8.3.4.1. Values for animal food yields, FY_j, were taken from Wittenberg (1990), who provided data for Canadian farm animals, as discussed by Zach and Sheppard (1992). Because the farming industry is becoming increasingly efficient (Acker 1983), these values will likely underestimate future yields. This would result in overestimates for the number of animals required.

Animal food yield values are not distributed in BIOTRAC because the data are food-type-specific and not very variable. For the postclosure assessment, we adopted an FY_j value for milk production of 4600 L or kg wet biomass$\cdot a^{-1}$. Beef cattle have an average retail cut weight of 230 kg at an average slaughter age of about 19 months, leading to an FY_j value for TE BEEF of 145 kg wet biomass$\cdot a^{-1}$. The yield for poultry is taken by combining the yields for meat and eggs. At slaughter, broiler chickens are about six weeks old and yield about 1.1 kg of meat. A chicken lays about 265 eggs per year, which have a net weight of about 13.3 kg. If we assume that poultry ingestion is equally divided between eggs and meat, an effective yield is found by averaging the egg and meat yields on a reciprocal basis

$$1/FY_{BIRD} = (1/1.1 + 1/13.3)/2 \qquad (9.2)$$

so that FY for TE BIRD becomes 2.03 kg wet biomass$\cdot a^{-1}$.

These yield values imply that a household of four adults would need to maintain or raise one dairy cow, four beef cattle and two egg-laying hens to satisfy normal dietary requirements. They would also need about 100 broiler chickens over the course of a year, but only 12 at any one time.

9.1.1.3 Field Areas

The size of the garden is calculated as the area, A_V (m^2), needed to grow the plant food (j = TE PLANT) to meet man's ingestion rate,

$$A_V = Nph \cdot U_j / Y_j \quad . \qquad (9.3)$$

The size of the forage field is calculated similarly by considering the area, A_F (m^2), required to grow the feed or forage needed by the livestock,

$$A_F = \sum_j Na_j \cdot Qf_j / Y_j \quad . \qquad (9.4)$$

The assumption that the same fields can be used year after year for plant and forage production is implicit in both these equations. The summation in Equation (9.4) extends over j = TE MILK, TE MEAT and TE BIRD. In these equations, Y_j (kg wet biomass$\cdot m^{-2}$ soil) is the yield of plant crops, as discussed in Section 8.5.8.1. Man's ingestion rate of TE PLANT, U_j (kg wet biomass$\cdot a^{-1}$), is calculated using the methods described in Section 8.3.4.1. Values for Qf_j (kg wet biomass$\cdot a^{-1}$), the feed ingestion rates for animals, are discussed in Section 8.5.5.1. Under average conditions, a family of four would need a garden of about 0.2 ha and a forage field of about 10 ha.

The critical group may heat its home with wood or with peat (Section 7.3.5.2). Areas for the woodlot or peat bog are calculated so that they will supply sufficient fuel to heat the home. For a woodlot, the area required each year, A_w ($m^2 \cdot a^{-1}$), is given by

$$A_W = FUELUS/(EW \cdot Yb) \quad . \tag{9.5}$$

Here, FUELUS ($MJ \cdot a^{-1}$) is the energy needed to heat the home for a year, EW ($MJ \cdot kg^{-1}$ wet biomass) is the convertible energy content of wood, and Yb (kg wet biomass$\cdot m^{-2}$ land) is the wood yield. These parameters have been discussed in Sections 7.5.2.1, 7.5.2.2 and 8.5.8.2 respectively. Here we use Yb instead of FY (Section 7.5.2.4) for the yield because essentially all the wood burned in a stove or fireplace is consumed. We assume that the woodlot can be regenerated in 50 a, so that the total area required is $50 \cdot A_w$ m^2. A typical home on the Ontario portion of the Canadian Shield can thus be heated on a continuous basis from a woodlot of about 10 ha.

The area required to provide sufficient peat to heat a home is calculated in much the same way. The area needed each year, A_p ($m^2 \cdot a^{-1}$), is given by

$$A_P = FUELUS/(EP \cdot PY) \tag{9.6}$$

where EP ($MJ \cdot kg^{-1}$ dry peat) is the convertible energy content of peat, and PY (kg dry peat$\cdot m^{-2}$ land) is the peat yield. These parameters have been discussed in Section 7.5.2.3 and 7.5.2.5 respectively. Because peat is formed very slowly, new sites for peat must be continually accessed to provide a sustained source of fuel. It is unreasonable to assume that the area of contaminated peat is large enough to supply fuel over 10 000 a or more. Instead, we assume that the contaminated area is large enough to provide heat for 50 a, and that the peat is available as a fuel to the critical group at any point in time regardless of the use previous generations have made of it. A peat bog of about 2.5 ha in area can supply fuel for the average Ontario home on the Canadian Shield for 50 a.

Note that the area of terrestrial contamination, A_T (Equation (7.2)), is the sum of A_V, A_F, A_W and A_P in any given BIOTRAC simulation.

9.1.1.4 Water Demand

In BIOTRAC, water demand is made up of four components: water for domestic purposes (drinking, cooking, bathing, laundry, etc.), drinking water for domestic animals, water for irrigating the garden, and water for irrigating the forage field. The critical group does not always employ aerial irrigation. The probabilities that we have assigned to this practice are discussed in Section 6.5.5.2.

The domestic water demand, W_D (m^3 water$\cdot a^{-1}$), is found by multiplying the domestic use per person, Uwc (m^3 water$\cdot a^{-1} \cdot p^{-1}$), by the number of persons, Nph, in the household (Section 9.1.1.1):

$$W_D = Nph \cdot Uwc \quad . \tag{9.7}$$

Domestic water needs vary considerably, and drinking water (Section 8.3.4.2) is a minor component. For Ontario, Uwc is estimated to range from 100 to 165 m^3 water$\cdot p^{-1} \cdot a^{-1}$ (OME 1987). We assume that values of Uwc are lognormally distributed, and adopt a GM of 130 m^3 water$\cdot p^{-1} \cdot a^{-1}$ and a GSD of

1.26 (Zach and Sheppard 1992). The distribution is truncated at its lower end at 20 m^3 water$\cdot p^{-1}\cdot a^{-1}$. For a family of four, W_D would typically be 520 m^3 water$\cdot a^{-1}$.

Water demand for livestock, W_L (m^3 water$\cdot a^{-1}$), is calculated similarly, using the water demand per animal, Qdv_j (m^3 water$\cdot a^{-1}$), and the number of animals raised by the critical group:

$$W_L = \sum_j Na_j \cdot Qdw_j \quad . \tag{9.8}$$

The summation in Equation (9.8) extends over j = TE *MILK*, TE *MEAT* and TE BIRD. Values for Qdw_j are discussed in Section 8.5.5.2. W_L is about 85 $m^3\cdot a^{-1}$ for a typical four-member family.

The amount of water, I_W (m water$\cdot a^{-1}$), required annually to irrigate one square metre of soil is calculated from SCEMR1 results as discussed in Section 6.2.2. I_W is the same for both agricultural fields because the garden and forage field are assumed to have the same soil type and depth, and to experience the same effective precipitation. The total amount of water required to irrigate each field will differ because the fields have different areas. When irrigation is practised, the water demand for the garden, W_V (m^3 water$\cdot a^{-1}$), is given by

$$W_V = I_W \cdot A_V \tag{9.9}$$

whereas for the forage field the demand, W_F (m^3 water$\cdot a^{-1}$), is

$$W_F = I_W \cdot A_F \quad . \tag{9.10}$$

Based on the field sizes, A_V and A_F, established in Section *9.1.1.3 for a* family of four, and assuming average conditions for the Canadian Shield, W_V and W_F are approximately 1200 and 6 x 10^4 m^3 water$\cdot a^{-1}$ respectively.

9.1.2 Water Sources and Probability of Well Water Use, LW (unitless)

In BIOTRAC, water needs for domestic use, livestock and aerial irrigation may be met by either a bedrock well or a lake. Forage field irrigation places very high demands on the water source. As shown in the preceding section, about 6 x 10^4 m^3 water are required each year to satisfy the irrigation needs of a typical forage field. This is more water than the well might be able to supply. Furthermore, extensive irrigation using well water is almost unknown on the Canadian Shield for economic reasons. Accordingly, whenever the forage field is irrigated (Section 6.5.5.2), we assume that the water source is the lake described in Chapter 5.

The source for other water needs is chosen in each simulation according to current probabilities of well and lake water usage by rural residents of the Canadian Shield in Ontario. By combining population data from Statistics Canada with well data from the Ontario Ministry of the Environment, Killey (1987) estimated that from 25 to 30% of the rural residents on the Ontario portion of the Canadian Shield depend on a well for their water supplies (Zach and Sheppard 1992). However, Killey noted that 50% might be a more appropriate estimate because the official well data do not include shallow wells dug by hand. We have adopted this higher value in BIOTRAC. A higher frequency of well usage in BIOTRAC is conservative because nuclide concentrations in well water generally exceed those in lake water (Chapter 10).

Accordingly, the water source is chosen to be a well in one half of all the simulations and a lake in the other half. In BIOTRAC, this is expressed by a switch, LW (unitless), which decides whether lake or well water is to be used. Water for domestic needs, livestock and watering of the garden is all assumed to be drawn from the same source in any given simulation, unless the supply is unable to meet the demand. We saw in Section 9.1.1.4 that a typical family of four needs about 520 m^3 water$\cdot a^{-1}$ for its household needs, 85 $m^3 \cdot a^{-1}$ for livestock and 1200 $m^3 \cdot a^{-1}$ to water the garden. Maximum demands may reach 5000 m^3 water$\cdot a^{-1}$ for domestic needs, 900 $m^3 \cdot a^{-1}$ for livestock and 1.1×10^5 $m^3 \cdot a^{-1}$ for irrigation. These needs are all easily met by the lake. Given a catchment area, A_d, of 1.06×10^8 m^2 (Section 5.5.1) and a minimum runoff, R, of 0.01 m water$\cdot a^{-1}$ (Section 5.5.4), the smallest volume of water flowing through the discharge lake is 1.06×10^6 $m^3 \cdot a^{-1}$, which is more than enough to meet all possible needs. Accordingly, if a lake is chosen as the water source, nuclide concentrations used throughout BIOTRAC are set equal to the concentration in the lake (Section 5.3). In this case, well-water demand is set to zero.

The well capacity, Q_{cap}, is not unlimited (Section 4.4.4). There is always enough water to meet household and livestock needs, but there may not be sufficient water to satisfy the irrigation demand. This leads to the following algorithm for water usage where a well has been selected as the source.

1. If the well can meet all domestic, livestock and irrigation needs, well water is used throughout BIOTRAC except to irrigate the forage field. The demand on the well, W_w (m^3 water$\cdot a^{-1}$), is calculated from

$$W_w = W_D + W_L + W_V \quad . \qquad (9.11)$$

2. If irrigation demands push the total well demand over the well capacity, Q_{cap}, well water is used for domestic and livestock purposes, and the garden is irrigated with lake water. Q_{cap} is established by the geosphere model, GEONET (Davison et al. 1994b), but the test is carried out in BIOTRAC. The demand on the well is given by

$$W_w = W_D + W_L \quad . \qquad (9.12)$$

Note that W_D, W_L and W_V are defined in Section 9.1.1.4. The probability is small that the well will be unable to meet all the demands placed on it, and GEONET can handle a variety of demands (Section 4.4.4). Therefore, whenever a well is chosen as the water source in BIOTRAC, it is almost always used to provide water for domestic and livestock purposes, and for watering the garden.

9.1.3 Water Balance Parameters

Precipitation reaching the earth's surface is lost through evapotranspiration and runoff (including both surface and subsurface components), or is stored on or below the surface. Over periods of several years or more, changes in water storage at a particular location can be ignored for our purposes, and precipitation, P, is balanced by evapotranspiration, ET, and runoff, R, (Thornthwaite 1944)

$$P = ET + R \quad . \tag{9.13}$$

A value of P is required in the atmosphere model to calculate the wet deposition velocity (Section 7.3.8). Runoff is needed in the surface water model to estimate the volume of water flowing through the discharge lake (Section 5.3). The soil model is driven by the amount of water, Pe, that is able to percolate down into the soil (Section 6.5.2.2); Pe, the effective precipitation, equals precipitation, P, minus surface runoff, R_s, and minus the amount of water, E_D, that evaporates from surface pools. Since both E_D and subsurface runoff are small compared with R_s, Pe is approximately equal to ET, and we assume that this is so in BIOTRAC. The chosen values of P, R and Pe should therefore satisfy Equation (9.13) in each BIOTRAC simulation, and so must be sampled in a consistent manner. The sampling scheme and the PDFs for P, R and Pe are discussed in this section.

The PDFs were constructed from information contained in the Hydrological Atlas of Canada (Fisheries and Environment Canada 1978). This atlas presents data on P, R and ET for all of Canada in the form of maps showing isolines that connect locations experiencing the same hydrological conditions. The maps are based on data compiled during the period 1941 to 1970, and so describe long-term average conditions. Furthermore, the isolines have been smoothed to eliminate local variations, and represent a spatial average on a scale of 10 000 km^2.

The precipitation data presented in the atlas were obtained from the AES archives. AES is responsible for the collection of most meteorological data in Canada. About 125 AES observing stations were in operation on the Ontario portion of the Canadian Shield over the period 1941 to 1970, although northern areas were not well covered. At each station, the precipitation measurements were obtained using official instruments, with exposure and observing practices conforming to prescribed standards (Environment Canada 1978). In addition, the data were subject to strict quality control procedures before they were archived. The precipitation data are therefore plentiful and of high quality, and form a reliable basis for the development of a PDF representing spatial variation of annual average precipitation on the Shield.

Annual runoff values for a given catchment are derived from measurements of the annual volume of water flowing through streams that drain the basin. The streamflow data presented in the atlas are based on observations coordinated by the Inland Waters Directorate of Environment Canada. About 150 stream gauges were in operation on the Shield in Ontario over the period 1941 to 1970, although the density of the network was poor in the far north. All of the measurements were obtained using standard instruments and procedures. The streamflow observations are considered reliable, although the small flows occurring under ice cover in winter may be underestimated.

Evapotranspiration is not measured routinely, and the values given in the atlas were deduced indirectly from other information. The Canadian landmass was first broken down into grid squares, and the ground cover (open water, wetland, forest, grassland, bare rock, etc.) characteristic of each square was determined. The evaporation, E_0, occurring from open water bodies in each square was found from maps of mean annual lake evaporation (Fisheries and Environment Canada 1978) determined from evaporation pan data and supplemented by calculations using climatological data. The evapotranspiration, ET, occurring from each of the other cover types was then expressed as a fraction of E_0 using a technique described by Thornthwaite and Mather (1957). Evapotranspiration for the entire catchment was determined by weighting the various ET values by the fraction of the area represented by each cover type, and summing over all cover types.

The evapotranspiration values deduced in this manner were found to be inconsistent with the precipitation and runoff data discussed above in the sense that they did not satisfy Equation (9.13). The discrepancies were attributed partly to the ET values themselves, and also to the precipitation data. Precipitation gauges tend to catch less than the true precipitation. The undercatch relates to the gauge dimensions, wind speed, and the type of precipitation. The discrepancies were eliminated by using an optimization technique to modify the ET and P values on a regional basis to bring them into balance with R. This was achieved by increasing P by about 6% across Ontario, and reducing ET by about 12%.

We deduced histograms for P, R and ET from their respective atlas maps by determining the areas contained between successive pairs of isolines (Figures 9-1 to 9-3). These histograms give the probability of encountering a given long-term average value of the parameter at a randomly selected point on the Ontario portion of the Canadian Shield under present-day conditions. The histograms for each of the three parameters are quite irregular; in the absence of other alternatives, we assumed that they represent normal distributions. Normal curves were fitted to the histograms by setting the distribution mean equal to the observed mean and by choosing a nominal SD to cover the observed range of values. The fitted distributions are plotted on Figures 9-1 to 9-3, and their attributes are listed in Table 9-2.

Note that despite the modifications made to ET and P to achieve a water balance, the sum of the mean values of ET and R in Table 9-2 does not equal the mean value of P for this region of Canada. This underlines the difficulty in deriving reliable evapotranspiration estimates from indirect climatological data (Amiro et al. 1988).

The atlas maps of P, ET and R all show a similar pattern, with low values in northwestern Ontario and high values in the southeast. The three parameters therefore appear to be spatially correlated across Ontario. Correlation coefficients, r, were calculated between P and ET and between P and R using pairs of values extracted from the maps for about 30 locations uniformly spread across the Canadian Shield region. The coefficient for P and ET was found to be 0.7; for P and R it was 0.8.

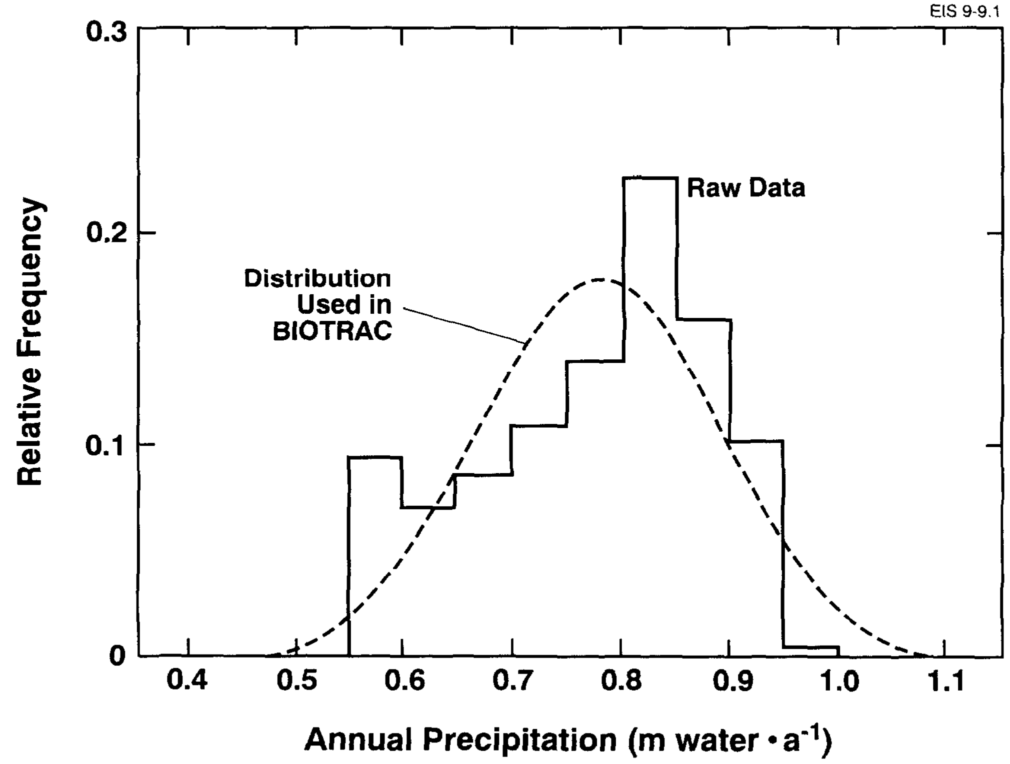


FIGURE 9-1: Distribution of Annual Precipitation, P, at Sites on the Ontario Portion of the Canadian Shield

It remains to define a sampling strategy for P, R and ET (or Pe) that will satisfy the water balance equation while preserving the distributions and correlations discussed above. Equation (9.13) is most easily satisfied if two of the parameters in it are sampled, and the remaining parameter is calculated by difference. We chose to sample P and R because these parameters are measured directly and their distributions are based on a large amount of high-quality data. The distribution for R is truncated on the lower end at 0.01 m water·a^{-1} to ensure that the flow through the discharge lake is sufficient to meet all the water demands of the critical group. P and R are assumed to be correlated, with a coefficient, r, equal to the observed value of 0.8. In each BIOTRAC simulation, Pe is then calculated from the sampled values of P and R according to

$$Pe = P - R \quad . \qquad (9.14)$$

Values of Pe calculated in this way are subject to the restriction Pe $\geq$ 0.2 m water·a^{-1} since smaller values leave too little water available to support agriculture. With this sampling strategy, the distribution for Pe is not specified a priori, but is determined by the set of values calculated using Equation (9.14). The nature of the calculated distribution was investigated by generating 1000 Pe values using the methods described above. The resulting distribution had a mean of 0.47 m water·a^{-1}, an SD of 0.064 m·a^{-1}, and a correlation coefficient with P of 0.65. These values

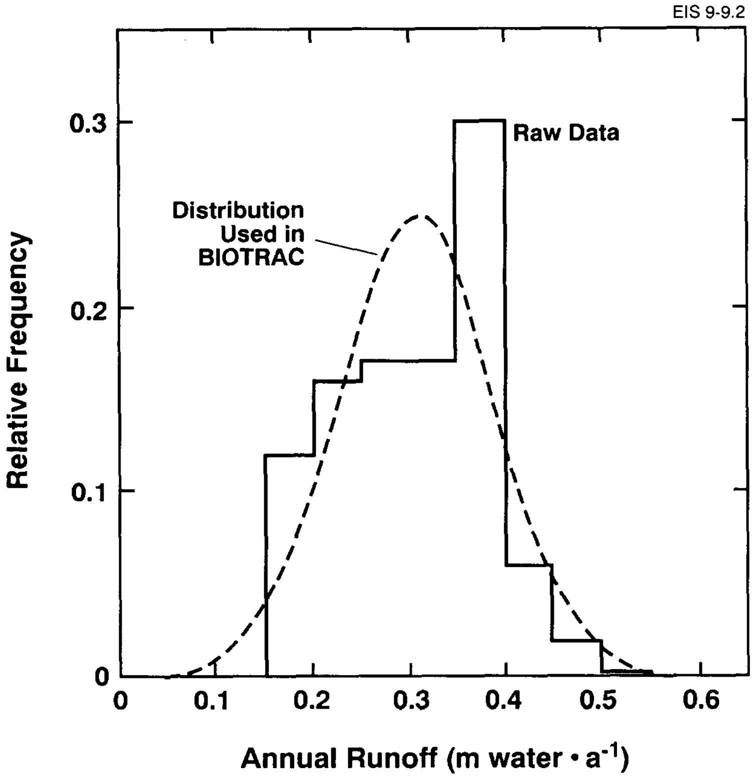


FIGURE 9-2: Distribution of Annual Runoff, R, at Sites on the Ontario Portion of the Canadian Shield

agree well with the corresponding attributes of the observed ET distribution (Table 9-2).

In summary, the distributions adopted for the water balance parameters are as follows. Precipitation, P, is distributed normally, with a mean of 0.78 m water·a^{-1} and an SD of 0.11 m·a^{-1}. Runoff, R, is distributed normally, with a mean of 0.31 m water·a^{-1}, an SD of 0.08 m·a^{-1}, and a lower truncation limit of 0.01 m·a^{-1}. P and R are correlated with a coefficient, r, of 0.8. Precipitation percolating into the soil, Pe, is calculated using Equation (9.14), subject to the restriction Pe $\geq$ 0.2 m water·a^{-1}.

Where P appears in the atmosphere model (Section 7.3.8), it is used in units of m water·d^{-1}. In these units, P is distributed normally, with a mean of 2.14×10^{-3} m water·d^{-1} and an SD of 3.01×10^{-4} m·d^{-1}.

9.2 INTEGRATION OF THE FOUR SUBMODELS

In this section we describe a walkthrough of a BIOTRAC simulation. We will discuss the order in which the calculations are made and how the output of one submodel serves as input to the next. We will also discuss BIOTRAC as one of the three models in the entire system model (Figure 1-3) under the direction of the SYVAC3 executive.

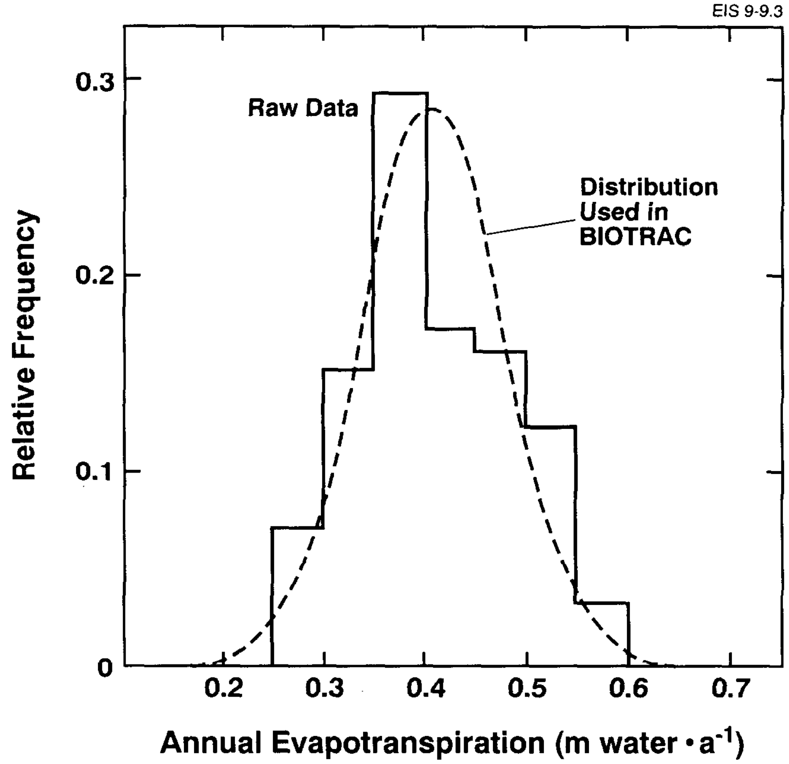


FIGURE 9-3: Distribution of Annual Evapotranspiration, ET, at Sites on the Ontario Portion of the Canadian Shield

TABLE 9-2

ATTRIBUTES OF THE NORMAL PROBABILITY DENSITY FUNCTIONS FITTED TO THE OBSERVED HISTOGRAMS OF PRECIPITATION, RUNOFF AND EVAPOTRANSPIRATION

Parameter	Mean (m water·a^{-1})	SD (m water·a^{-1})
Precipitation, P	0.78	0.11
Runoff, R	0.31	0.08
Evapotranspiration, ET	0.41	0.07

A typical SYVAC3 simulation begins by assigning values to all sampled parameters by random selection from their PDFs (Goodwin et al. 1994). Since all values are available at the start of the simulation, they can be applied consistently where they appear in more than one place in the models. The sampled values are then used to calculate values for a number of additional parameters. Man's ingestion rates for the five food types, water ingestion rate and inhalation rate are calculated from the specified diet using the methods described in Section 8.3.4. The number of domestic animals raised by the critical group and the areas of the garden, forage field, woodlot and peat bog (if an organic soil has been specified (Section 6.5.1.1) and peat is used as the heating fuel (Section 7.3.5.2)) are calculated using the equations presented in Section 9.1.1. Water demand is computed from this information (Section 9.1.1.4), taking irrigation requirements into account if the garden or forage field are to be irrigated (Section 6.5.5.2). If the well has been chosen as the water source, the well-water demand is passed to the geosphere model (Section 4.4.4).

Once all the sampled and calculated parameter values have been set, SYVAC3 directs the simulation of nuclide transport through the system. As noted in Section 2.6, time series are calculated for each time-dependent model variable in turn. The order in which the variables are treated is chosen to ensure that the information required at each point in the model is available from previous calculations. Variables associated with the vault are handled first for all the nuclides, followed by the variables pertaining to the geosphere and then the biosphere. All the nuclide and water flows out of the geosphere are therefore available to drive BIOTRAC (Chapter 4). Time series for each nuclide are calculated in turn in BIOTRAC; chains are treated member by member, starting with the precursor and working through the daughters (Goodwin et al. 1994).

Nuclide and water flows out of the geosphere are used directly to calculate time series of well-water concentrations, concentrations in compacted sediments, and pore-water concentrations beneath terrestrial discharge zones (Chapter 4). The nuclide flows are also input to the surface water submodel, and lake-water concentrations and effective sediment concentrations are calculated for all times (Chapter 5). Air concentrations resulting from the suspension of nuclides from aquatic sources are calculated next (Chapter 7). This is followed by the calculation of soil concentrations in each field resulting from groundwater contamination, irrigation (where irrigation is practised) and deposition of lake-derived material (Chapter 6). The total soil concentration for each field is then calculated, but it is replaced by the sediment concentration when sediments are used as soil (Section 6.3.7.3). Indoor and outdoor air concentrations resulting from suspension from terrestrial and biomass sources are computed, as well as the rate at which nuclides are deposited from the air to plants (Chapter 7). At this point, all the concentrations of both radioactive and chemically toxic nuclides in the physical compartments of the biosphere are known. Concentrations in plants and animals are then computed (Chapter 8). For radionuclides, the calculations are extended to dose to humans via all exposure pathways, with intermediate results available for the rate of nuclide uptake by humans. Doses for non-human biota are calculated in an analogous fashion, as discussed in Chapter 13.

The calculation of doses marks one of the end point of the models. For each simulation, values for all the sampled and calculated parameters and for all the consequences (concentrations and doses) are passed to the SYVAC3 executive for storage in an output file (Goodwin et al. 1994). SYVAC3 also writes a summary of a set of simulations, describing, for example, the number of simulations that were carried out. Further analysis of the results can be carried out by separate computer programs that access the stored results.

Step-by-step results from a typical BIOTRAC simulation are presented in Appendix D to illustrate how the model works and to put the various exposure pathways into perspective.

A schematic representation of the entire biosphere model is shown in Figure 9-4.

9.3 NUCLIDE MASS BALANCE

In formulating BIOTRAC we made certain assumptions about nuclide transport within the biosphere. These assumptions make the model tend to overpredict the nuclide mass in the various parts of the biosphere. This means that BIOTRAC tends to predict higher concentrations and doses than if the same quantities of nuclides entered the biosphere in the corresponding real system (Section 1.5.6). We now examine these assumptions related to mass balance and their implications on model predictions, focusing on ^{129}I and ^{14}C, which are by far the most important nuclides in the postclosure assessment (Section 10.3.2, Goodwin et al. 1994). The data for the analysis are based on the median-value simulation of the central group of scenarios of the postclosure assessment (Goodwin et al. 1994).

9.3.1 The Mass Balance Equation

At any time in a simulation, the mass of a nuclide in the biosphere model is given by the integrated amount that has entered from the geosphere plus ingrowth from its precursor, minus losses from outflows and radioactive decay. Mathematically, the following formal mass balance equation can be written for each nuclide modelled in BIOTRAC:

$$M_p^i(t) = M_{in}^i(t) - M_{out}^i(t) \qquad (9.15)$$

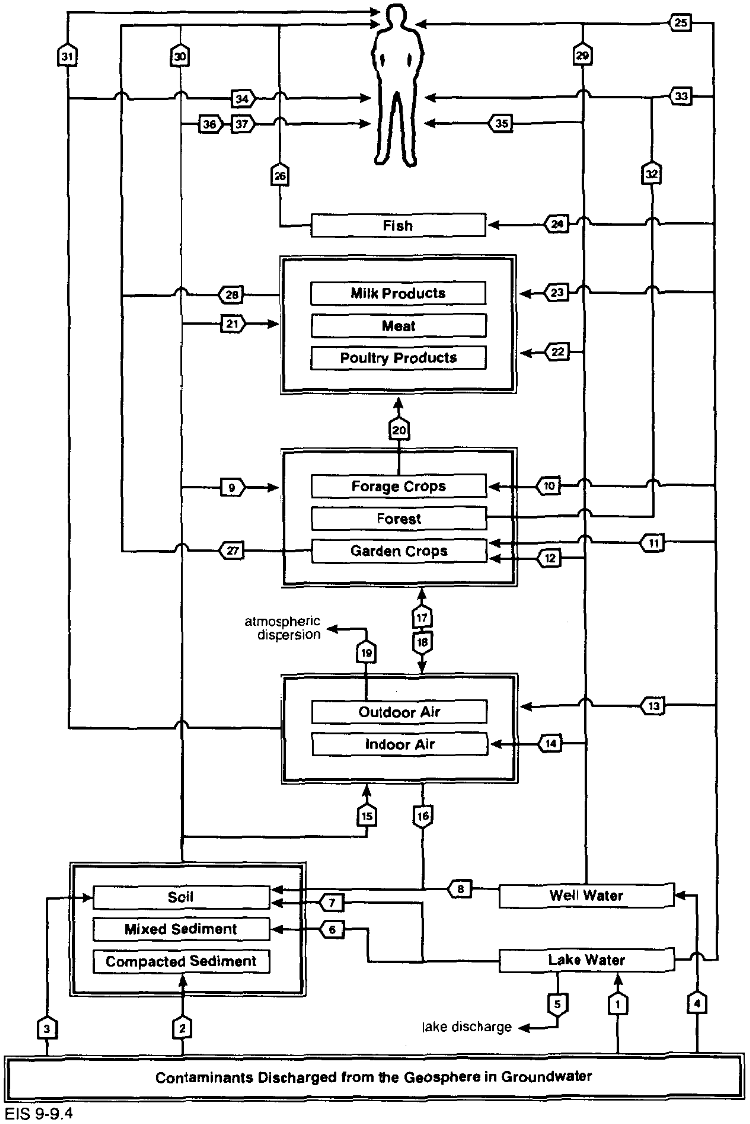
where $M_p^i(t)$ is the mass of nuclide i present in the biosphere at time t (mol),

$M_{in}^i(t)$ is the mass of nuclide i entering the biosphere up to time t, adjusted for losses from radioactive decay and ingrowth for chain nuclides (mol), and

$M_{out}^i(t)$ is the mass of nuclide i flowing out of the biosphere up to time t (mol).

$M_{in}^i(t)$ may be calculated as the convolution to time t of all the flows entering the biosphere through the geosphere discharges (Section 4.4), allowing for subsequent decay of the nuclide, and ingrowth from the precursor nuclide for chain nuclides (Section 2.6).

$M_{out}^i(t)$ may be calculated as the convolution to time t of all the flows leaving the biosphere. For BIOTRAC, the only losses are through lake outflow downstream (Section 5.3.1) and dispersion of airborne nuclides beyond the area inhabited by the critical group (Section 7.3.1).



1 Geosphere discharge to lake water
2 Geosphere discharge to compacted sediment
3 Geosphere discharge to bottom of soil profile
4 Geosphere discharge to well water
5 Loss through lake discharge
6 Transfer from lake water to mixed sediment
7 Transfer from lake water to soil (irrigation)
8 Transfer from well water to soil (irrigation)
9 Transfer from soil/sediments to plants (root uptake)
10 Transfer from lake water to leaves of forage crops (irrigation)
11 Transfer from lake water to leaves of garden crops (irrigation)
12 Transfer from well water to leaves of garden crops (irrigation)
13 Transfer from lake water to air
14 Transfer from well water to indoor air
15 Transfer from soil/sediments to air
16 Transfer from air to soil (deposition)
17 Transfer from air to plant leaves (deposition)
18 Transfer from plants to air
19 Loss through atmospheric dispersion
20 Transfer from forage crops to milk products, meat, and poultry products
21 Transfer from soil/sediments to milk products, meat, and poultry products
22 Transfer from well water to milk products, meat, and poultry products
23 Transfer from lake water to milk products, meat, and poultry products
24 Transfer from lake water to fish
25 Transfer from lake water to man (ingestion)
26 Transfer from fish to man (ingestion)
27 Transfer from garden crops to man (ingestion)
28 Transfer from milk products, meat, and poultry products to man (ingestion)
29 Transfer from well water to man (ingestion)
30 Transfer from soil/sediments to man (ingestion)
31 Transfer from air to man (inhalation)
32 External exposure of man to wooden building materials
33 External exposure of man to lake water
34 External exposure of man to air
35 External exposure of man to well water
36 External exposure of man to soil/sediments
37 External exposure of man to inorganic building materials

FIGURE 9-4: Schematic Representation of all the Nuclide Transport and Exposure Pathways for Humans Treated by BIOTRAC

If mass balance is strictly enforced, $M_p^i(t)$ will equal the sum of the nuclide amounts in the lake water, mixed sediments, soils of the four fields, well water, air, plants, terrestrial animals, fish, building materials and man. Mathematically, $M_p^i(t)$ may thus be represented as the sum of all masses of the nuclide in the different parts of the biosphere

$$\begin{aligned} M_p^i(t) = {} & M_{lake}^i(t) + M_{sed}^i(t) + M_{air}^i(t) + M_{soil}^i(t) \\ & + M_{well}^i(t) + M_{anim}^i(t) + M_{man}^i(t) + M_{pla}^i(t) \\ & + M_{fish}^i(t) + M_{bmat}^i(t) \end{aligned} \tag{9.16}$$

where $M_x^i(t)$is the mass (mol) of nuclide i at time t in part x of the biosphere. Here sed corresponds to mixed sediment, anim to terrestrial animals, pla to plants, and bmat to building materials.

Equations (9.15) and (9.16) can be combined to relate the flows into and out of the biosphere to the amount in it at any time t:

$$\begin{aligned} M_{in}^i(t) - M_{out}^i(t) = {} & M_{lake}^i(t) + M_{sed}^i(t) + M_{air}^i(t) \\ & + M_{soil}^i(t) + M_{well}^i(t) + M_{anim}^i(t) \\ & + M_{man}^i(t) + M_{pla}^i(t) + M_{fish}^i(t) \\ & + M_{bmat}^i(t) \quad . \end{aligned} \tag{9.17}$$

Some of the quantities in these equations can be calculated in a fairly straightforward manner. However, several of them require certain approximations to keep the calculations tractable.

Furthermore, assumptions about transfers between parts of the biosphere have been made throughout BIOTRAC to simplify the modelling (Section 15.2). Thus, concentrations in a source or donor compartment are generally *not* depleted when some small fraction of a nuclide is transferred to another, recipient, compartment. This overestimates the amount in the source compartment without underestimating that in the recipient compartment, and is generally used only when exact transfer data or models are lacking. These assumptions make the model tend to overpredict the total amount of each nuclide in source compartments and the doses to man and other biota. However, the overestimates in doses will be small because doses vary linearly with amount of nuclide, and only small fractions of nuclide are not subtracted and are thus counted twice.

In mathematical terms, the approximations in BIOTRAC are such that $M_p^i(t)$ in Equation (9.15) will be greater than the expression on the right-hand side of the equation. In other words, BIOTRAC will always be tracking more nuclides in the biosphere than the net difference between inflows to and outflows from the biosphere up to that time. This is entirely appropriate in an assessment model such as BIOTRAC, which is designed to run efficiently and overestimate consequences when uncertainties arise.

In the following sections, we discuss the mass balance assumptions for each compartment in turn. We show how the magnitudes of the various terms in

Equation (9.17) can be estimated, and the results are summarized in Table 9-3.

9.3.2 Mass in Lake Water, M^i_{lak}.

The mass balance for each nuclide is maintained within the surface water submodel (Chapter 5). The equation describing nuclide content of the lake water is a true mass balance equation (Equation 5.1). However, in setting up this equation it was assumed that all the nuclides released from the geosphere enter the lake directly (Section 4.4). Ignored are the delay in and the losses from the portions of the flow entering the biosphere via the well and soil that only later make their way to the lake through runoff and drainage.

The effect of this conservative assumption about well water can be estimated. As shown in Section 5.3.3, the concentration in the lake water is approximately proportional to the geosphere flow into the lake (Equation 5.19). In the median-value simulation for the central group of scenarios (Section 1.5.1, Goodwin et al. 1994), the ^{129}I flow to the biosphere at 100 000 a is about 3×10^{-6} mol·a^{-1} from the well and 9×10^{-6} mol·a^{-1} from the Boggy Creek south discharge zone (Figure 4-4). In assuming that all the well water drawn is used and released instantaneously to the lake with no loss of nuclide mass, we may overestimate the lake-water concentration by up to about one third $(3 \times 10^{-6} + 9 \times 10^{-6})/(9 \times 10^{-6})$. This is also true for times earlier than 100 000 a because the relative magnitudes of the ^{129}I flows to the biosphere remain relatively constant. This overestimation of the lake water concentration is conservative, but not overly so, given the many uncertainties in the calculations in BIOTRAC.

Also ignored are delays and losses in the flows through the parts of the discharge zones that contaminate soils from below before they, too, find their way to the lake. Since the fraction of the discharge zone underlying a terrestrial area can be no greater than 0.1 (Section 4.5.1), and since only a small portion of the discharging nuclide mass actually enters the soil profile, little mass is generated through this assumption. We also assume that nuclides and water taken from the lake for aerial irrigation or domestic use return instantaneously to the lake with no loss. Since the ratio of the annual water demand to the annual flow of water through the lake is typically 4×10^{-5}, this assumption also has an insignificant effect on the mass balance or the calculated doses. Note that the above ratio is based on a demand of 1335 m^3 water·a^{-1} (Appendix D.2), a catchment area of 1.06×10^8 m^2 (Section 5.5.1) and a runoff value of 0.31 m water·a^{-1} (Section 5.5.4).

Transfers to the atmosphere can be ignored in calculating concentrations in a given compartment if the rate constants are small compared with the rate constants describing other losses from the compartment. Table 9-4 shows the values of the rate constants in the median-value simulation for aquatic processes and for transfers to the atmosphere for which lake contents are not depleted.

Rate constants for transfers are defined as

$$\phi^i = Fl^i/Inv^i \tag{9.18}$$

TABLE 9-3

FLOWS AND AMOUNTS OF ^{129}I IN THE BIOSPHERE (mol at 100 000 a) IN THE MEDIAN-VALUE SIMULATION FOR THE CENTRAL GROUP OF SCENARIOS OF THE POSTCLOSURE ASSESSMENT

Inflows, Outflows, Mass Present and Transfers	Amounts*
Cumulative Inflow:	
- via Boggy Creek south discharge zone	2.05×10^{-1}
- via well	7.36×10^{-2}
Total: M^i_{in}	2.78×10^{-1}
Cumulative Outflow:	
- in lake outflow	2.72×10^{-1}
Total: M^i_{out}	2.72×10^{-1}
$M^i_{in} - M^i_{out}$	0.06×10^{-1}
Mass Present:	
- in lake water, M^i_{lake}	1.31×10^{-7}
- in mixed sediment, M^i_{sed}	7.00×10^{-3}
- in soil, M^i_{soil}	
- of garden	3×10^{-4}
- of forage field	9×10^{-5}
- of woodlot	2×10^{-5}
- in plant crops, M^i_{pla}	
- of garden	
- by root uptake	7×10^{-10}
- by leaf deposition	5×10^{-10}
- of forage field	
- by root uptake	9×10^{-10}
- by leaf deposition	2×10^{-8}
- of woodlot	
- by root uptake	7×10^{-9}
- by leaf deposition	2×10^{-8}
Total: $M^i_{lake} + M^i_{sed} + M^i_{soil} + M^i_{pla}$**	7.4×10^{-3}
Mass Transferred to Atmosphere over 100 000 a:	
- from lake	
- as gas	1×10^{-2}
- on particulates	3×10^{-7}
- from soil	
- as gas	7×10^{-5}
- on particulates	9×10^{-8}
- in agricultural fires	9×10^{-9}
- in biomass combustion for energy	2×10^{-9}
- in forest and land-clearing fires	5×10^{-10}

* Each entry has a numerical precision of at least 0.1%. Entries have not been modified to correct for known overpredictions of concentration.

** Not including any contributions for mass in well water, terrestrial animals, fish, man, and building materials.

TABLE 9-4

MEDIAN VALUES OF RATE CONSTANTS, ϕ, for ^{129}I and ^{14}C FOR PROCESSES IN THE SURFACE-WATER MODEL AND FOR RELEASES TO THE ATMOSPHERE FOR WHICH LAKE CONTENTS ARE NOT DEPLETED

Processes and Releases	Median Value of Rate Constant (a^{-1})		
	Formula/Source	^{129}I	^{14}C
Processes modelled in surface-water model			
Radioactive decay, λ^i	Table 1-1	4.41×10^{-8}	1.21×10^{-4}
Flushing of the lake, f	Equation (5.2)	89.0	89.0
Water/mixed-sediment transfer, α^i	Table 5-1	2.3	2.2
Gaseous evasion, η_1^i	Section 5.5.9	0	0.92
Releases from lake water to atmosphere for which lake contents are not depleted			
Aquatic gases*	$AIML/(DISP \cdot Z_1)$	3.9	**
Aquatic particulates*	$AADL/(DISP \cdot Z_1)$	8.7×10^{-5}	8.7×10^{-5}

Note: For definitions of symbols see Appendix B.

* The atmospheric dispersion factor, DISP, has units of $a \cdot m^{-1}$.
** Lake water is depleted for gaseous evasion of ^{14}C.

where ϕ^i is the rate constant for the transfer of nuclide i (a^{-1}),

Fl^i is the mass of nuclide i transferred per unit time ($mol \cdot a^{-1}$), and

Inv^i is the mass of nuclide i in the source compartment (mol).

Table 9-4 shows that releases of the volatile nuclides ^{129}I and ^{14}C from the lake to the atmosphere are small compared to losses through flushing. For ^{14}C, atmospheric releases are modelled explicitly, but this is not true for ^{129}I (Section 5.5.9). Thus, there is no depletion for ^{129}I, but the water concentration is overestimated by less than 5% as a result of this. The atmospheric releases for the other gaseous nuclides, ^{39}Ar, ^{3}H, ^{81}Kr,

^{85}Kr, ^{79}Se and ^{222}Rn, would be similar to or greater than those for ^{14}C. However, we do not model volatilization for these nuclides because they are treated in special ways (Sections 2.5 and 5.3.4).

As a result of these simplifications related to mass balance, the surface-water model will tend to overpredict nuclide concentrations in the lake water and doses from pathways involving lake water by up to about 30% (Section 5.7).

9.3.3 Mass in Mixed Sediments, M^i_{sed}

The equation used to compute nuclide concentrations in the mixed sediment on the lake bed is also a true mass balance equation (Equation 5.7). The mixed sediment is not directly accessible to the critical group, except in those simulations where the lake is dredged or drained to provide soil for food production (Section 6.5.5.4). Nuclide accumulation in these sediments can be calculated by convolution over the simulation period. Because sediment concentrations are derived from nuclide concentrations in the water column, the nuclide mass in the sediment will be overestimated to the same extent as the mass in the water column (Section 9.3.2).

9.3.4 Mass in Soil, M^i_{soil}

The soil submodel for the postclosure assessment assumes the form of regression equations that define time-dependent nuclide concentrations in soil pore-water, and buildup in surface soil in response to irrigation, air deposition and contaminated groundwater (Section 6.3). The SCEMR1 model used to derive the regression equations maintains mass balance for the processes considered. Therefore the regression model reflects an implicit mass balance in the same way. We explicitly model losses resulting from leaching, cropping (Section 6.3.4), gaseous evasion (Section 6.5.4) and radioactive decay. The nuclide mass in air, plants, animals, building materials and humans is assumed to come in part from the soil, but such losses are not included in the soil model.

The contents of the soils in the four fields considered in BIOTRAC (Section 1.5.4) are computed layer by layer (Figure 6-1) . The soil submodel directly generates concentrations for the root zone and bottom layers. The nuclide concentration in the remaining intermediate layer is estimated by averaging the surface and bottom layer concentrations. This is of little consequence to dose because the critical group never accesses the intermediate layer, but it means that the nuclide content of the soil cannot be calculated exactly.

As with the surface-water submodel, transfers to the atmosphere can be ignored in calculating concentrations in soils if the rate constants, ϕ, are small compared with the rate constants describing other losses from soils. Table 9-5 shows the rate constants in the median-value simulation of the central group of scenarios for the processes occurring in the soils and transfers to the atmosphere for which soil contents are not depleted. For both ^{14}C and ^{129}I, the ϕ values for gaseous evasion, which is modelled explicitly, are very much greater than those for any of the ignored releases. Therefore, soil concentrations are not substantially overestimated by not considering these minor depletions.

TABLE 9-5

MEDIAN VALUES OF RATE CONSTANTS, ϕ, FOR ^{129}I and ^{14}C FOR PROCESSES IN THE SOIL MODEL AND FOR RELEASES TO THE ATMOSPHERE FOR WHICH SOIL CONTENTS ARE NOT DEPLETED

Processes and Releases	Median Value of Rate Constant (a^{-1})		
	Formula/Source	^{129}I	^{14}C
Processes modelled in soil model			
Radioactive decay, λ^i	Table 1-1	4.41×10^{-8}	1.21×10^{-4}
Leaching	*	2.6×10^{-5}	1.0×10^{-5}
Cropping losses, ζ^i	Equation (6.44)	3.4×10^{-6}	4.6×10^{-4}
Gaseous evasion, η^i_s	Section 6.5.4	3.2×10^{-2}	8.8
Releases from soil to atmosphere for which soil contents are not depleted			
Terrestrial gases, η^i_s		**	**
Terrestrial particulates	$ADL/(DISP \cdot Z_r \cdot \rho_s)$	2.2×10^{-4}	2.2×10^{-4}
Agricultural fires	$\frac{Bv \cdot ff \cdot Y \cdot EMFRAC}{\rho_s \cdot Z_r}$	6.9×10^{-5}	9.2×10^{-3}
Biomass combustion for for energy	$\frac{Bv \cdot FUELUS \cdot EMFRAC}{EW \cdot \rho_S \cdot Z_r \cdot A_W}$	1.8×10^{-5}	2.4×10^{-3}
Forest and land-clearing fires	$\frac{Bv \cdot FY \cdot EMFRAC \cdot f_1}{\rho_s \cdot Z_r}$	3.7×10^{-6}	5.0×10^{-4}

Note: For definitions of symbols see Appendix B.

* Sheppard M.I. and Hawkins 1991b.
** Soil is depleted for gaseous evasion of ^{14}C, ^{129}I and also for ^{79}Se.

For all the above reasons the soil model will tend to overpredict soil concentrations to a small degree as a result of the lack of an exact mass balance.

9.3.5 Mass in Air, M^i_{air}

Because the atmosphere compartment has no explicit volume, nuclide mass in the air is calculated indirectly by considering transfers from surface water and soil. Losses need not be considered because the air concentrations are not depleted through deposition to underlying surfaces (Section 7.2). For each suspension process we can calculate a flux to the atmosphere, which, when multiplied by an appropriate time and area, provides the sum of the nuclide mass in air and the mass lost as a result of atmospheric transport. Thus transfers from water and soil to air appear in Equation (9.17) as contributors to both $M^i_{air}(t)$ and $M^i_{out}(t)$. The calculation will be approximate for those suspension mechanisms that involve mass loading parameters (Section 7.3.3).

As indicated in Chapter 7, we do not model depletion of suspended contaminant plumes as a result of deposition. The effect of this approximation can be estimated in certain cases where the fluxes from and to the surface are calculated independently, e.g., for fires. The flux of nuclide i to the atmosphere as a result of the suspension process, $Q^i(t)$, and the resulting air concentration, $C^i_a(t)$, is given by

$$C^i_a(t) = Q^i(t) \cdot DISP \tag{9.19}$$

where $C^i_a(t)$ is the concentration of nuclide i in air ($mol \cdot m^{-3}$ air),

$Q^i(t)$ is the flux of nuclide i from a source compartment to the atmosphere ($mol \cdot m^{-2} \cdot s^{-1}$ or $mol \cdot s^{-1}$), and

DISP is the appropriate atmospheric dispersion factor ($s \cdot m^{-1}$ or $s \cdot m^{-3}$).

For all the nuclides except ^{39}Ar, ^{3}H, ^{81}Kr, ^{85}Kr and ^{222}Rn (^{14}C is deposited to soil only), the flux to the surface from the contaminated air is given by (Equation (7.35) and (7.36))

$$D^i(t) = (Vd + P \cdot Wr) \cdot C^i_a(t) \tag{9.20}$$

where $D^i(t)$ is the flux of nuclide i deposited to the underlying surface ($mol \cdot m^{-2} \cdot s^{-1}$),

Vd is the dry deposition velocity ($m \cdot s^{-1}$),

P is the average precipitation rate (m water$\cdot s^{-1}$), and

Wr is the washout ratio (unitless).

Equations (9.19) and (9.20) can be combined to give the ratio of the fluxes down to underlying surfaces and up to the atmosphere for nuclide i:

$$D^i(t)/Q^i(t) = (Vd + P \cdot Wr) \cdot DISP \quad . \tag{9.21}$$

For the median-value simulation of the central group of scenarios, $Vd = 6.0 \times 10^{-3}\ m \cdot s^{-1}$, $P = 7.8 \times 10^{-1}$ m water$\cdot a^{-1}$, $Wr = 2.5 \times 10^5$ and

DISP = 22.5 $s \cdot m^{-1}$ (lake as area source) or 18.66 $s \cdot m^{-1}$ (fields as area source), and so $D^i(t)/Q^i(t) \simeq 0.2$. This implies that air concentrations will be overestimated by about 20% in some cases, such as emissions from fires, because depletion of the airborne plume is not modelled. In cases where $Q^i(t)$ is an estimate of the net flux from ground surfaces (e.g., $(Q^i)_{TG}$) (Section 7.3.4.2), air concentrations are not overestimated because both upward and downward fluxes have been considered.

9.3.6 Mass in Plants, M^i_{pla}

The nuclide mass in plant crops, $M^i_{pla}(t)$ may be calculated as the sum of crop contamination via roots and leaves for the plants in each field (Section 8.3.1).

The nuclide mass in crops via root uptake, RP, is given by

$$(M^i_{pla,e})_{RP}(t) = Bv^i \cdot C^i_{s,e} \cdot Y_e \cdot A_e \qquad (9.22)$$

where $(M^i_{pla,e})_{RP}(t)$ is the mass of nuclide i at time t in plant crops in field e from root uptake (mol),

Bv^i is the plant/soil concentration ratio for nuclide i ((Bq·kg^{-1} wet biomass)/ (Bq·kg^{-1} dry soil)),

$C^i_{s,e}$ is the annual average soil concentration of nuclide i in the root zone of field e (mol·kg^{-1} dry soil),

Y_e is the plant yield for field e, (kg wet biomass·m^{-2} soil or land), and

A_e is the area of field e (m^2).

The nuclide mass in crops from leaf deposition, LP, is given by

$$(M^i_{pla,e})_{LP}(t) = [(D^i_b)_D \cdot r_{e,D} + (D^i_b)_I \cdot r_{e,I}] \cdot [1 - \exp(-\lambda^i_E \cdot te_e)] \cdot A_e / \lambda^i_E \qquad (9.23)$$

where $(D^i_b)_D$ is the rate of atmospheric deposition of nuclide i to vegetation (mol·m^{-2}·d^{-1}),

$r_{e,D}$ is the plant interception fraction for atmospheric deposition for field e (unitless),

$(D^i_b)_I$ is the rate of deposition of nuclide i to vegetation through irrigation water (mol·m^{-2} soil·d^{-1})

$r_{e,I}$ is the plant interception fraction for irrigation for field e (unitless),

λ^i_E is the effective removal constant of nuclide i from vegetation (d^{-1}), and

te_e is the time of above-ground exposure for plant crops in field e (d).

The fields considered in Equations (9.22) and (9.23) are the garden, forage field and woodlot. The two equations are expressed in terms of these fields, e, rather than our food types, j, as is the case in the food-chain and dose submodel (Chapter 8). The parameters in the two equations are closely related to those introduced and documented in Chapter 8. Thus, Y_e corresponds to Y_j and Yb (Section 8.5.8); $r_{e,D}$ and $r_{e,I}$ to r_j and rb (Section 8.5.7); te_e to te_j and teb (Section 8.5.3.2); and, finally, A_e to A_V, A_F and A_W (Section 9.1.1.3).

9.3.7 Mass in Remaining Compartments, M^i_{well}, M^i_{anim}, M^i_{fish}, M^i_{man} and M^i_{bmat}

$M^i_{well}(t)$, $M^i_{anim}(t)$, $M^i_{fish}(t)$, $M^i_{man}(t)$ and $M^i_{bmat}(t)$ will always be negligible with respect to the other parts of the biosphere because of the relatively small amounts of nuclides involved. Accordingly, these masses can be ignored in the mass balance.

When nuclides enter these compartments, an equal amount is not removed from the source compartment (water, soil or air (Section 8.2)). So, again, the concentrations in these latter compartments will tend to be slightly overestimated.

9.3.8 Discussion

The assumptions related to mass balance made in the different submodels of BIOTRAC are justified on various grounds. Some processes, like air suspension and redeposition of particles, are fast and largely cyclical, and result in low net mass transfer (Section 7.7). Some flows, like surface runoff from soil to the lake, represent significant net transfers to the recipient compartments assumed in the model, but with some delay (Sections 5.7 and 6.7). Assuming that such transfers take place instantaneously is therefore conservative. Yet other transfers, such as the use of soil for building materials or the use of sediment as soil, represent shifts of material normally inaccessible to man to a location where they can give a dose to individuals of the critical group. Hence the mass balance assumptions will lead to overpredictions of doses.

Detailed data to calculate the rates of such transfers are often unavailable. It is also not always obvious whether the mass of nuclides in question would lead to a greater dose if present in the source or the recipient compartments. Ignoring losses from the source compartments greatly simplifies modelling; the approximation will lead to an overestimation of concentrations and doses because the assumed nuclide mass in the source compartment is always overestimated while the proper mass is transferred to the receptor compartment (Section 15.2).

Figure 9-5 shows the accumulated mass of ^{129}I flowing into the biosphere from the geosphere (Section 4.4) and out of it through lake outflow (Section 5.3) up to 100 000 a for the median-value simulation of the central group of scenarios (Goodwin et al. 1994). We use ^{129}I as an example because it gives by far the largest dose to man at all times over the postclosure assessment period (Section 10.3.2.1, Goodwin et al. 1994). Table 9-3 summarizes the relative magnitudes of the masses in the different parts of the biosphere as a function of flows to and from the biosphere.

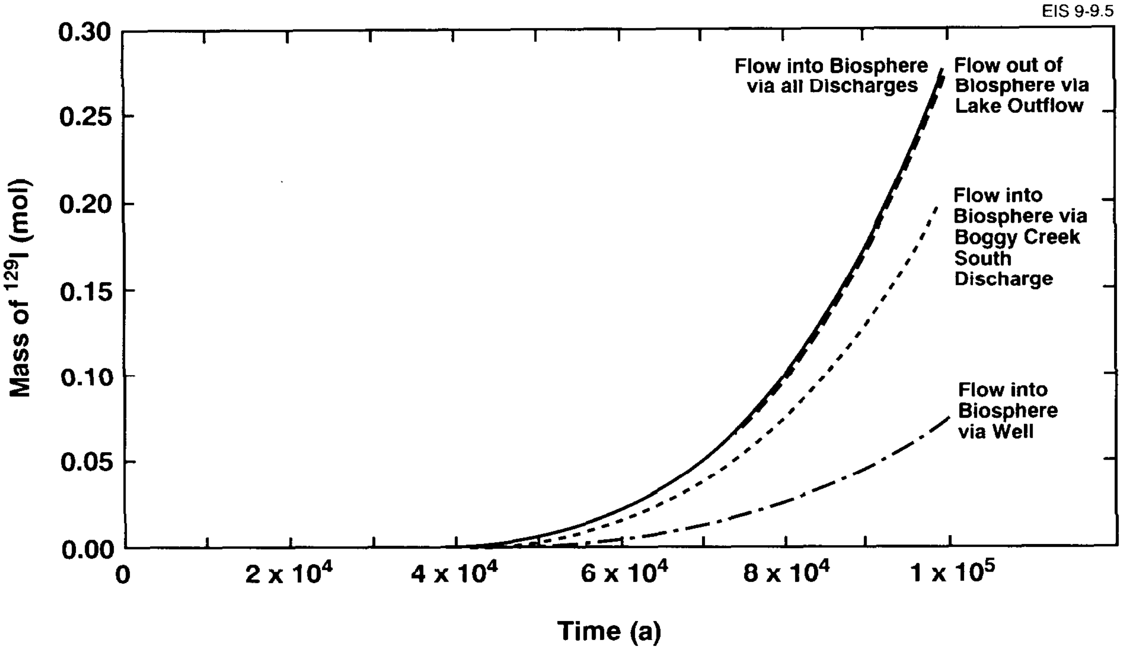


FIGURE 9-5: Flows of ^{129}I Into and Out of the Biosphere up to 100 000 a. The results are for the median-value simulation of the central group of scenarios (Goodwin et al. 1993). The flows have been corrected for radioactive decay. Almost all the flow into the biosphere is diluted in the lake and leaves downstream. The bulk of the remainder, about 2.5%, remains in mixed lake sediments. Much lower amounts are present in lake water, soils, air, plant crops, terrestrial animals, well water, fish, man and building materials.

Almost all of the ^{129}I that enters the area inhabited by the critical group leaves in lake outflow and so $M^i_{in} - M^i_{out}$ assumes a small value of 0.06×10^{-1} mol·a^{-1} (Equation (9.17)). This value represents the mass of nuclide present in the biosphere. The value is slightly lower than the 7.4×10^{-3} mol·a^{-1} calculated for the mass present in the biosphere from a variety of separate contributions (Table 9-3). This is largely because we do not subtract the gaseous transfer from the lake to the atmosphere from the lake-water contents. This leads to an overprediction of the lake-water concentration and downstream losses, thereby underestimating the difference between M^i_{in} and M^i_{out}.

We calculated each of the integrals for M^i_{in} and M^i_{out} numerically accurate to about 0.1%. From the values in Table 9-3, the error in $M^i_{in} - M^i_{out}$ would be about 6.0×10^{-4}, an order of magnitude less than the value of $M^i_{in} - M^i_{out}$ itself. Thus, the combined limits of accuracy of the individual terms makes up a substantial portion of $M^i_{in} - M^i_{out}$. As a result, the difference may not always provide a reliable estimate of the mass present in the biosphere.

The limitation on numerical accuracy is an additional reason for not attempting to maintain an exact mass balance by depleting source compartments because of transfers to recipient compartments. The ignored losses are often smaller than the achievable numerical accuracy in calculating the nuclide mass in the source compartment. Nevertheless, the calculated flows are accurate enough to estimate the small amounts entering the recipient compartment.

After 100 000 a, about 2.5% of the ^{129}I that has entered the biosphere over the period remains in the mixed lake sediments. Much smaller amounts remain in the other parts of the biosphere occupied by the critical group and other biota.

9.3.9 Effect on Dose

Most of the dose to individuals of the critical group comes from ^{129}I through drinking well water, eating plants grown on garden soil irrigated with well water, and eating foods from animals whose drinking water also came from the well (Section 10.3, Goodwin et al. 1993). Dose estimates related to ingestion of well water by man and animals are not affected by the assumptions leading to the overestimation of mass in BIOTRAC because well-water concentrations are calculated explicitly in GEONET (Section 4.4.4).

Similarly, the excess nuclide mass in BIOTRAC will not lead to overestimation of the nuclide concentrations in garden soil irrigated with well water because we include all the important transfers explicitly. The dose resulting from ingestion of garden crops contaminated through root uptake should therefore be accurate. On the other hand, the dose from deposition on vegetation of ^{129}I suspended from the lake may be overestimated by about one third, corresponding to the overestimated ^{129}I concentration in the lake water (Section 9.3.2). This is of minor importance if the garden is irrigated with well water because the bulk of the ^{129}I deposited on vegetation emanates from the soil, which has been contaminated by irrigation water. Additional details on the ^{129}I pathways and concentrations involved in this simulation are given in Goodwin et al. (1993).

We conclude that the generation of mass in BIOTRAC may lead to overestimation of predicted doses to man by at most 30% when lake water is the only water source and fields are on soil derived from sediments. In most instances, the impact of generated mass will be much lower. Overestimation of consequences, as indicated here, is entirely appropriate in a model such as BIOTRAC designed for the postclosure assessment of nuclear fuel waste disposal.

9.4 PARAMETER AND VARIABLE LINKAGES

BIOTRAC is a complex model because it links up with the geosphere model and is composed of various models and submodels. These include the geosphere/biosphere interface model (sediment, surface water, soil and well) (Chapter 4), and the surface water, soil, atmosphere, and food-chain and dose submodels (Chapters 5 to 8). An integration model is needed for linking the four submodels to form BIOTRAC (Chapter 9). Many parameters and variables are calculated in one model or submodel, only to be used in another. These linkages are listed in Table 9-6. The first three columns of Table 9-6 list the parameters and variables involved in linkages according to the model or submodel; the remaining columns describe the use of the parameters and variables according to model or submodel, and equation. Note that full parameter definitions are given in Appendix B.

We have not included our model for predicting radiological doses to non-human biota (Section 13.3) in Table 9-6 because the linkages are essentially the same as those for the food-chain and dose submodel.

9.5 COMPARISON WITH THE PRECLOSURE BIOSPHERE MODEL

BIOTRAC was developed to assess the performance of the disposal facility in its postclosure phase (Goodwin et al. 1994). An analogous assessment model is required for the preclosure phase, which comprises the construction, operation, decommissioning and closure of the facility (Grondin et al. 1994), and a separate model has been designed to address this need. The Preclosure Radiological Environmental Assessment Code (PREAC) was developed by Ontario Hydro to evaluate the radiological impact on members of the public and non-human biota from chronic nuclide emissions from a used-fuel disposal facility during the preclosure phase (Russell 1993). Although the modelling requirements in the two phases are quite different, they both deal with nuclide transport and exposure in the biosphere immediately surrounding the facility. The two models are not closely linked, i.e., BIOTRAC starts where PREAC ends, without receiving input from PREAC. This coincides with the interface between the pre- and postclosure phases. Given an actual site, the two models could be more closely linked.

In this section, we discuss how the design of the models allows them to treat two different types of releases while providing a consistent assessment of consequences. Grondin et al. (1994) discuss this topic further from the preclosure perspective.

9.5.1 Overview of the Preclosure Model

In the preclosure phase, the disposal facility can be compared to a modern industrial operation. Nuclide emissions to the environment as a result of routine operation would be controlled to very low levels, as required by the AECB and other regulators. Releases would normally occur directly to the atmosphere through a stack, and to a water body through a discharge pipe (Grondin et al. 1994, Simmons and Baumgartner 1994). Public access within 1.5 km of the facility would be restricted. The facility is expected to be in active operation for about 40 a.

The facility can therefore be assessed using traditional methods and models that have been developed for assessing nuclear power installations. The critical individual is assumed to be self-sufficient and residing on a farm at the site boundary. The assessment is based on a single deterministic evaluation of consequences using best-estimate or conservative parameter values. The pre- and postclosure assessments share the common difficulty of performing an evaluation of a facility for which no site has been selected. The preclosure approach to this restriction is to define reference environments for each of three regions of the Canadian Shield in Ontario (Gee 1981). Environmental data (Reid H.E. and Grondin 1993,

TABLE 9-6

BIOSPHERE MODEL LINKAGES FOR PARAMETERS AND VARIABLES CALCULATED IN ONE OF THE MODELS (GEOSPHERE, INTERFACE, SURFACE WATER, SOIL, ATMOSPHERE, FOOD CHAIN AND DOSE, AND INTEGRATION) AND USED IN ANOTHER MODEL

Definition of Parameter			Use of Parameter			
Source*	Quantity	Symbol	Model	Equation	Calculated Quantity	Symbol
Geosphere Model						
-	retardation factor	RE	Interface	4.2	flow nuc. out of geosphere	χ^d
-	well capacity	Q_{cap}	Integration	9.11,9.12	well demand	W_W
-	critical well demand	Q_{crt}	Interface well	4.18	con. nuc. i in well water	C_{ww}^i
-	flow nuc. i into well	χ_w^i	Interface well	4.18	con. nuc. i in well water	C_{ww}^i
-	surf. water flow into well	V_{ld}	Interface well	4.18	con. nuc. i in well water	C_{ww}^i
-	flow nuc. i out of comp. sed.	χ_{ds}^i	Interface sed.	4.3	con. nuc. i in comp. sed. pore w.	C_{pw}^i
-	water flow out of comp. sed.	Fw_{ds}	Interface sed.	4.3	con. nuc. i in comp. sed. pore w.	C_{pw}^i
-	comp. sed. porosity	P_{ds}	Interface sed.	4.7	con. nuc. i in comp. sed.	C_{ds}^i
-	comp. sed. part. coeff.	Kd_{ds}^i	Interface sed.	4.7	con. nuc. i in comp. sed	C_{ds}^i
-	flow nuc. i from comp. sed.	χ^i	Surface water	5.6	con. nuc. i in surf. water	C_l^i
-	flow nuc. i out of overb.	χ_{ob}^i	Interface soil	4.14	con. nuc. i in bott. soil pore w.	C_{pw}^i
-	water flow out of overb.	Fw_{ob}	Interface soil	4.14	con. nuc. i in bott. soil pore w.	C_{pw}^i
-	flow nuc. i out of overb.	χ_{ob}^i	Soil	6.45	con. nuc. i in shall. soil pore w.	C_{pw}^i
-	water flow out of overb.	Fw_{ob}	Soil	6.45	con. nuc. i in shall. soil pore w.	C_{pw}^i
-	total area of discharge zone	A_{DZ}	Interface soil	4.1	area of terr. discharge	A_{TD}
-	flow nuc. i from comp. sed.	χ^i	Atmosphere	7.11	con. of Ar and Kr in air	$(C_a^i)_{AG}$
Interface Model						
Eq 4.1	area of terr. discharge	A_{TD}	Soil	4.14	eff. con. nuc. i in bott. soil pore water (Section 6.3.7.1)	C_{pw}^i

continued...

TABLE 9-6 (continued)

Definition of Parameter			Use of Parameter			
Source*	Quantity	Symbol	Model	Equation	Calculated Quantity	Symbol
Interface Model (sediment)						
Eq 4.7	con. nuc. i in comp. sed.	C^i_{ds}	Surface water	5.15	eff. con. nuc. i in sed.	C^i_{se}
Interface Model (soil)						
Eq 4.14	con. nuc. i in bott. soil pore water	C^i_{pw}	Soil	4.14	eff. con. nuc. i in bott. soil pore water (Section 6.3.7.1)	C^i_{pw}
Interface Model (well)						
Eq 4.18	con. nuc. i in well water	C^i_{ww}	Soil	6.39	con. nuc. i in soil root zone for irrig.	$(C^i_s)_I$
Eq 4.18	con. nuc. i in well water	C^i_{ww}	Atmosphere	7.32	flux nuc. i to ind. air from dom. water	$(Q^i)_{IGW}$
Eq 4.18	con. nuc. i in well water	C^i_{ww}	Food-chain/dose	8.9	dep. nuc. i to veg. for irrig.	$(D^i_b)_I$
Eq 4.18	con. nuc. i in well water	C^i_{ww}	Food-chain/dose	8.14	man's int. nuc. i from anim. d.w.	$(E^i_j)_{WA}$
Eq 4.18	con. nuc. i in well water	C^i_{ww}	Food-chain/dose	8.20	man's int. nuc. i from d.w.	$(E^i)_W$
Eq 4.18	con. nuc. i in well water	C^i_{ww}	Food-chain/dose	8.36	man's ext. dose from w. imm.	$(D^i)_{WI}$
Surface Water Submodel						
Eq 5.6	con. nuc. i in surf. water	C^i_l	Soil	6.39	con. nuc. i in soil root zone for irrig.	$(C^i_s)_I$
Eq 5.6	con. nuc. i in surf. water	C^i_l	Atmosphere	7.13	con. nuc. i in air from aq. part susp.	$(C^i_a)_{AP}$
Eq 5.6	con. of Rn in surf. water	C^{Rn}_l	Atmosphere	7.17	flux of Rn gas to air from surf. water	$(Q^{Rn})_{AG}$
Eq 5.6	con. of I in surf. water	C^I_l	Atmosphere	7.20	con. of I in air from aq. emiss.	$(C^I_a)_{AG}$
Eq 5.6	con. of C in surf. water	C^C_l	Atmosphere	7.22	con. of C in air from aq. emiss.	$(C^C_a)_{AG}$
Eq 5.6	con. nuc. i in surf. water	C^i_l	Atmosphere	7.32	flux nuc. i to ind. air from dom. water	$(Q^i)_{IGW}$

continued...

TABLE 9-6 (continued)

Definition of Parameter			Use of Parameter			
Source*	Quantity	Symbol	Model	Equation	Calculated Quantity	Symbol
Eq 5.6	con. nuc. i in surf. water	c_1^i	Food-chain/dose	8.9 and 8.12	dep. nuc. i to veg. for irrig.	$(D_b^i)_I$
Eq 5.6	con. nuc. i in surf. water	c_1^i	Food-chain/dose	8.14	man's int. nuc. i from anim. d.w.	$(E_j^i)_{WA}$
Eq 5.6	con. nuc. i in surf. water	c_1^i	Food-chain/dose	8.18	man's int. nuc. i from fish	$(E_j^i)_{WF}$
Eq 5.6	con. nuc. i in surf. water	c_1^i	Food-chain/dose	8.20	man's int. nuc. i from d.w.	$(E_j^i)_W$
Eq 5.6	con. nuc. i in surf. water	c_1^i	Food-chain/dose	8.36	man's ext. dose from w. imm.	$(D^i)_{WI}$
Eq 5.15	eff. con. nuc i in sed.	c_{se}^i	Atmosphere	7.12	con. nuc. i in air from terr. part. susp.	$(C_a^i)_{TP}$
Eq 5.15	eff. con. of Ra in sed.	c_{se}^{Ra}	Atmosphere	7.15	flux of Rn gas to air from soil	$(Q^{Rn})_{TG}$
Eq 5.15	eff. con. of I in sed.	c_{se}^I	Atmosphere	7.18	flux of I gas to air from soil	$(Q^I)_{TG}$
Eq 5.15	eff. con. of C in sed.	c_{se}^C	Atmosphere	7.21	con. of C in air from terr. emiss.	$(C_a^C)_{TG}$
Eq 5.15	eff. con. of Se in sed.	c_{se}^{Se}	Atmosphere	7.23	con. of Se in air from terr. emiss.	$(C_a^{Se})_{TP}$
Eq 5.15	eff. con. nuc i in sed.	c_{se}^i	Atmosphere	7.25	con. i in biomass	C_b^i
Eq 5.15	eff. con. nuc i in sed.	c_{se}^i	Atmosphere	7.28	con. nuc. i in air from heating fires	$(C_a^i)_{EF}$
Eq 5.15	eff. con. nuc i in sed.	c_{se}^i	Atmosphere	7.29	flux nuc. i to air from outd. fires	$(Q^i)_{LF}$
Eq 5.15	eff. con. of Ra in sed.	c_{se}^{Ra}	Atmosphere	7.31	con. of Rn in ind. air from soil	$(C_a^{Rn})_{IGS}$
Eq 5.15	eff. con. nuc i in sed.	c_{se}^i	Food-chain/dose	8.2	man's int. nuc. i from root upt.	$(E_j^i)_{RP}$
Eq 5.15	eff. con. nuc i in sed.	c_{se}^i	Food-chain/dose	8.6	man's int. nuc. i from anim. root upt.	$(E_j^i)_{RPA}$
Eq 5.15	eff. con. nuc i in sed.	c_{se}^i	Food-chain/dose	8.16	man's int. nuc. i from anim. soil ing.	$(E_j^i)_{SA}$
Eq 5.15	eff. con. nuc i in sed.	c_{se}^i	Food-chain/dose	8.22	man's int. nuc. i soil ing.	$(E^i)_S$
Eq 5.15	eff. con. nuc i in sed.	c_{se}^i	Food-chain/dose	8.37	man's ext. dose from ground	$(D^i)_G$
Eq 5.15	eff. con. nuc i in sed.	c_{se}^i	Food-chain/dose	8.39	con. nuc. i in inorg. build. mat.	C_{ibm}^i
Eq 5.15	eff. con. nuc i in sed.	c_{se}^i	Food-chain/dose	8.40	con. nuc. i in wooden build. mat.	C_{wbm}^i

continued...

TABLE 9-6 (continued)

Definition of Parameter			Use of Parameter			
Source*	Quantity	Symbol	Model	Equation	Calculated Quantity	Symbol
Soil Submodel						
Tb 6-6	con. nuc. i in soil root zone	C_s^i	Atmosphere	7.12	con. nuc. i in air terr. part. susp.	$(C_a^i)_{TP}$
Tb 6-6	con. of Ra in soil root zone	C_s^{Ra}	Atmosphere	7.15	flux of Rn gas to air from soil	$(Q^{Rn})_{TG}$
Tb 6-6	con. of I in soil root zone	C_s^I	Atmosphere	7.18	flux of I gas to air from soil	$(Q^I)_{TG}$
Tb 6-6	con. of C in soil root zone	C_s^C	Atmosphere	7.21	con. of C in air from terr. emiss.	$(C_a^C)_{TG}$
Tb 6-6	con. of Se in soil root zone	C_s^{Se}	Atmosphere	7.23	con. of Se in air from terr. emiss.	$(C_a^{Se})_{TP}$
Tb 6-6	con. nuc. i in soil root zone	C_s^i	Atmosphere	7.25	con. i in biomass	C_b^i
Tb 6-6	con. nuc. i in soil root zone	C_s^i	Atmosphere	7.28	con. nuc. i in air from heating fires	$(C_a^i)_{EF}$
Tb 6-6	con. nuc. i in soil root zone	C_s^i	Atmosphere	7.29	flux nuc. i to air from outd. fires	$(Q^i)_{LF}$
Tb 6-6	con. of Ra in soil root zone	C_s^{Ra}	Atmosphere	7.31	con. of Rn in ind. air from soil	$(C_a^{Rn})_{IGS}$
Tb 6-6	con. nuc. i in soil root zone	C_s^i	Food-chain/dose	8.2	man's int. nuc. i from root upt.	$(E_j^i)_{RP}$
Tb 6-6	con. nuc. i in soil root zone	C_s^i	Food-chain/dose	8.6	man's int. nuc. i from anim. root upt.	$(E_j^i)_{RPA}$
Tb 6-6	con. nuc. i in soil root zone	C_s^i	Food-chain/dose	8.16	man's int. nuc. i from anim. soil ing.	$(E_j^i)_{SA}$
Tb 6-6	con. nuc. i in soil root zone	C_s^i	Food-chain/dose	8.22	man's int. nuc. i soil ing.	$(E^i)_S$
Tb 6-6	con. nuc. i in soil root zone	C_s^i	Food-chain/dose	8.37	man's ext. dose from soil	$(D^i)_G$
Tb 6-6	con. nuc. i in soil root zone	C_s^i	Food-chain/dose	8.39	con. nuc. i in inorg. build. mat.	C_{ibm}^i
Tb 6-6	con. nuc. i in soil root zone	C_s^i	Food-chain/dose	8.40	con. nuc. i in wooden build. mat.	C_{wbm}^i
Eq 6.1	irr. water required	I_W	Integration	9.9	irrigation demand for garden	W_V
Eq 6.1	irr. water required	I_W	Integration	9.10	irrigation demand for forage field	W_F
Eq 6.1	irr. water required	I_W	Food-chain/dose	8.9	dep. nuc. i to veg. for irrig.	$(D_b^i)_I$

continued...

TABLE 9-6 (concluded)

Definition of Parameter			Use of Parameter			
Source*	Quantity	Symbol	Model	Equation	Calculated Quantity	Symbol
Atmosphere Submodel						
Eq 7.13	con. nuc. i in air from aq. part. susp.	$(C_a^i)_{AP}$	Soil	6.16	dep. nuc. i air to soil	D_s^i
Eq 7.20	con. of I in air from aq. emiss.	$(C_a^I)_{AG}$	Soil	6.16	dep. I from air to soil	D_s^I
Eq 7.22	con. of C in air from aq. emiss.	$(C_a^C)_{AG}$	Soil	6.16	dep. C from air to soil	D_s^C
Tb 7-2	con. nuc. i in indoor air	$(C_a^i)_I$	Food-chain/dose	8.25	man's int. nuc. i from inhal.	$(E^i)_A$
Tb 7-1	con. nuc. i in outdoor air	$(C_a^i)_O$	Food-chain/dose	8.25	man's int. nuc. i from inhal.	$(E^i)_A$
Tb 7-2	con. nuc. i in indoor air	$(C_a^i)_I$	Food-chain/dose	8.35	man's ext. dose from air imm.	$(D^i)_{AI}$
Tb 7-1	con. nuc. i in outdoor air	$(C_a^i)_O$	Food-chain/dose	8.35	man's ext. dose from air imm.	$(D^i)_{AI}$
Eq 7.40-7.43	dep. nuc. i air to veg.	D_b^i	Food-chain/dose	8.8	man's int. nuc. i from leaf dep.	$(E_j^i)_{LP}$
Eq 7.40-7.43	dep. nuc. i air to veg.	D_b^i	Food-chain/dose	8.12	man's int. nuc. i from anim. leaf. dep.	$(E_j^i)_{LPA}$
Eq 7.40-7.43	dep. nuc. i air to veg.	D_b^i	Food-chain/dose	8.40	con. nuc. i in wooden build. mat.	C_{wbm}^i
Integration Model						
Eq 9.11,9.12	well demand	W_W	Geosphere	-	flow nuc. i into well	X_W^i
Eq 9.11,9.12	well demand	W_W	Geosphere	-	flow nuc. i from comp. sed.	X_i
Eq 9.3-9.6	areas of fields	$A_{V,F,W,P}$	Soil	-	conc. nuc. i in bott. soil pore water (Section 6.3.7.1)	C_{pw}^i
Eq 9.3-9.6	areas of fields	$A_{V,F,W,P}$	Soil	-	conc. nuc. i in soil root zone (Section 6.3.7.3)	C_s^i
Eq 9.3-9.6	areas of fields	$A_{V,F,W,P}$	Atmosphere	7.4	atm. disp. fact. for terr. area	$(DISP)_T$

* Tb = Table and Eq = Equation.

Grondin et al. 1994) are synthesized to produce parameter values that describe a representative, hypothetical biosphere for each of these regions.

The preclosure model PREAC is described in detail by Russell (1993). Briefly, it is a compartment model that simulates transport using equilibrium transfer coefficients to represent physical, chemical and biological processes. It is based largely on Canadian Standards Association guidelines for calculating derived release limits for radioactive material in airborne and liquid effluents for normal operation of nuclear facilities (CSA 1987).

In the preclosure phase, nuclides may be emitted directly to the atmosphere and to a surface water body. Air concentrations downwind from the source are calculated using the long-term average form of the Gaussian plume dispersion model for a continuous release (Pasquill and Smith 1983), and representative data for wind speed, wind direction and atmospheric stability. Nuclides can be deposited from the air to soil, plants and the surface water body; depositional fluxes are calculated using wet and dry deposition velocities.

Nuclide concentrations in water are computed for discharge to both a river and a lake, and the higher values are used in subsequent dose calculations. In both cases, nuclides enter the water directly from the disposal facility; in the case of lake discharge, deposition from the atmosphere also acts as a source term. Water concentrations are calculated using a mass balance approach that includes sedimentation. Mixing is assumed to occur instantaneously throughout the water body, and emission rates and flow rates are average values that are assumed to remain constant over the year and also over the entire preclosure assessment period. Nuclide concentrations in the surface water body are assumed to be the maximum concentrations that occur during the operating period of the facility. Concentrations for beach sediments are calculated from water concentrations using a partition coefficient.

The soil in the preclosure model becomes contaminated by atmospheric deposition of nuclides and by the application of contaminated irrigation water. Time-dependent soil nuclide concentrations are calculated from the nuclide flux to the surface and an effective removal constant, which determines the fraction of the incoming flux that remains in the topmost soil layer, taking leaching, sorption and radioactive decay into account. The maximum calculated soil nuclide concentration that occurs during the operating period of the facility is used for dose prediction.

PREAC calculates the dose to man via ten primary exposure pathways. Four of these are external pathways: immersion in air, immersion in water, ground exposure or groundshine, and exposure to contaminated beach sediments. The remaining six are internal pathways: inhalation, and the ingestion of vegetables and fruit, fish, water, soil and animal products (milk, beef, pork, eggs and poultry). Vegetables and fruit are assumed to be contaminated by airborne deposition and irrigation water. In both cases, the pathway includes direct transfer to plants via deposition and indirect transfer via root uptake from the soil. Animal products are assumed to become contaminated through inhalation and the ingestion of feed or forage, water and soil. Pasture land is assumed not to be irrigated. In each pathway, the dose to humans is linearly related to concentrations in the various environmental compartments through steady-state transfer coefficients and other parameters.

Individual doses are calculated at the site boundary for both adult reference man (ICRP 1975) and an infant. Collective doses are calculated for the population residing within 100 km of the facility using hypothetical but conservative demographic and food production data. Doses to non-human biota, represented by a series of target organisms, are also calculated.

9.5.2 Comparison of BIOTRAC and PREAC

It is clear from the above discussion that the BIOTRAC and PREAC models differ, but this is not surprising. The differences do not constitute inconsistencies, given the different goals of the models and the different phases they address. This is illustrated below for some of the more obvious differences.

1. Basically, PREAC produces a single dose prediction for humans based on best-estimate parameter values, whereas BIOTRAC is part of a systems variability analysis code that produces a distribution of consequences. A probabilistic model is required for the postclosure assessment to account for the variability and uncertainty involved in dealing with a deep underground source over very long periods of time (Section 1.5.2). The preclosure model is not faced with this challenge, and a deterministic approach is justifiable. Moreover, PREAC uses data for three reference environments to provide an estimate of the variability in its predictions.

2. Nuclide air concentrations are calculated very differently in the two models. However, each model uses methods that are appropriate for its source configuration. The Gaussian plume model used by PREAC is the standard and accepted way to model the dispersion of routine releases from an elevated point source (CSA 1987). On the other hand, BIOTRAC uses methods appropriate to an area source at ground level (Section 7.3). Deposition is handled similarly in both models.

3. Surface water concentrations are calculated in much the same way in the two models. BIOTRAC does not explicitly consider a river, although for some combinations of parameter values its lake could resemble a pool in a creek or a large, slowly moving river rather than a lake (Section 5.5). Less contaminated sediment would be retained in a river so that a lake is the conservative choice for postclosure (Section 5.1).

4. The lake area adopted in PREAC (>1.0 km^2) is much larger than that used in BIOTRAC (Section 5.5.3). The operational facilities during preclosure must be sited on a lake that is large enough to meet the demands for process water without disrupting the existing aquatic life. Deep groundwater flowing through the vault may discharge to this lake, but it is just as likely that local

hydraulic head distributions and the subsurface fracture structure could force discharge to another lake of a different size. This can only be determined in a site-specific setting. Since smaller lakes have higher nuclide concentrations in water, the lake areas adopted in BIOTRAC are conservative.

5. Sediment concentrations are calculated differently in the two models. In BIOTRAC, compacted sediments become contaminated through contact with discharging groundwater (Section 4.4.1) and mixed sediments above by sedimentation from the water column (Section 5.3.2). Concentrations are calculated using partition coefficients and rate constants respectively in the two layers. PREAC does not need to consider the compacted sediment layer because nuclides in the preclosure phase are discharged directly to the water column and time spans are relatively short. PREAC calculates mixed-sediment concentrations using partition coefficients. This is not in conflict with the BIOTRAC approach because partition coefficients are related to rate constants in a simple way (Bird et al. 1992). Sediments are also used for different purposes in the two models. In BIOTRAC, they can become available as agricultural soil (Section 6.3.7.3), and so can participate in a number of exposure pathways, including ingestion and exposure. In PREAC, sediments are assumed to be unavailable, and can contribute to dose only through external exposure on a contaminated beach. The probability that sediments will be accessed in other ways is low and does not need to be considered for an operational facility with a lifetime of about 40 a.

6. The formulations of the BIOTRAC and PREAC soil models are quite different, but both models are similar in concept. Soil concentrations in BIOTRAC are calculated using regression equations that are based on SCEMR1 output and depend on water balance and soil parameters (Section 6.2). Soil concentrations in PREAC are calculated using a simple empirical equation that is based on experimental data and depends on essentially the same parameters as in BIOTRAC. The somewhat more complex approach taken in BIOTRAC was developed primarily to treat the case in which the soil becomes contaminated from below. This case need not be considered in PREAC.

7. A number of transport and exposure pathways considered in BIOTRAC do not appear in PREAC. Most of these (e.g., gaseous emission from the lake, release of nuclides to indoor air through use of domestic water, and burning of biomass contaminated through aerial irrigation) involve transport from the lake to the atmosphere (Section 7.3). These pathways are important in BIOTRAC because suspension from the lake and other surfaces is important in determining air contamination. On the other hand, nuclides are released directly to the atmosphere in the preclosure phase and suspension processes contribute little to air concentrations.

8. PREAC assumes that all the water demands are supplied by the surface water body. In the absence of an underground contaminant

source, well water would be uncontaminated. Exposure to contaminated building materials is not considered in PREAC because it is thought unlikely that trees or gravel near an operating facility would be contaminated and used for construction. The ingrowth of daughters is not considered in the food-chain compartments of PREAC. The inventory of nuclides that might be released in the preclosure phase contains no long-lived daughters, and buildup is not significant. PREAC includes animal inhalation as an exposure pathway, whereas BIOTRAC does not. Few data are available to evaluate this pathway, but the dose contribution to man due to animal inhalation is expected to be very much less than the contribution from water or food ingestion and man's inhalation (Zach 1985b, Zach and Sheppard 1992).

9. PREAC treats ^{14}C using a specific-activity model, whereas BIOTRAC employs a transport model. As noted in Section 2.5.2, the specific activity approach is appropriate for a direct release to the atmosphere, but is unsuitable for treating an underground source, as is the case in postclosure.

10. PREAC calculates collective doses whereas BIOTRAC does not. It is not possible to predict population patterns on the Canadian Shield far in the future, and so collective dose estimates for the postclosure phase would have little meaning (AECB 1987).

11. PREAC calculates doses to a series of target organisms for all the radionuclides considered in preclosure, whereas BIOTRAC calculated such doses for the most important postclosure radionuclides only. This is appropriate because they are the only potentially important dose contributors (Section 10, Goodwin et al. 1993).

12. The values of parameters common to the two models are not always equal. Where parameter values in BIOTRAC are fixed and describe particular physical or biological processes (e.g., radioactive decay constants (Table 1-1) or DCFs (Tables 8-3 and 8-4)), the two models generally adopt identical values. Where parameter values in BIOTRAC are distributed, the central values are usually equal to the fixed values specified in PREAC. Where differences occur, the postclosure values are generally more conservative to reflect the greater uncertainty arising from the longer time frame of the assessment. In all cases, the PDFs used in BIOTRAC encompass the fixed values adopted in PREAC, so that the generic biosphere defined in BIOTRAC simulations will include the three reference environments of the preclosure assessment.

To summarize, the pre- and postclosure models show some differences in terms of philosophy, model formulation and parameter values. But these differences do not constitute inconsistencies, given the different situations that the models address. The models are well suited to their respective purposes and provide complementary approaches to different phases of the assessment of the overall project.

10. SENSITIVITY ANALYSIS OF BIOTRAC

10.1 INTRODUCTION

The objective of a sensitivity analysis is to determine how the output of a model responds to changes in the values of the input parameters. The analysis identifies and attempts to rank the parameters to which the output is most sensitive. In a model such as BIOTRAC, sensitivity analysis can also be used to identify the processes, pathways and nuclides that are important in determining environmental concentrations and doses to humans. Sensitivity analysis is particularly useful in non-linear models, where the relationship between the input parameters and model predictions is not always obvious.

Sensitivity analysis has four main benefits:

1. It can aid in setting priorities for the future development of the model and the determination of parameter values. Work can focus on the sensitive processes and parameters, allowing an improved model to be produced with a minimum of resources and effort. The model may be simplified by removing pathways or processes shown to be unimportant. In the case of a probabilistic model, the analysis can help to indicate which parameters need to be distributed to account for uncertainty, and which can be assigned fixed values.

2. The knowledge of how a model responds to its parameter values can help to optimize the design of a disposal facility. Similarly, the results of a sensitivity analysis can be used to identify the characteristics that might make a disposal site technically more acceptable.

3. Sensitivity analysis can be used as a quality assurance tool. Confidence in the model is increased if it responds to changes in the parameter values as expected on an intuitive basis. Complex models can show emergent properties and behave unexpectedly. Such behaviour may lead to reevaluation and modification of the model and its parameters.

4. Sensitivity analysis can increase understanding of a model by revealing the relationship between its parameters and its predictions, and by providing the opportunity to examine its behaviour under a variety of conditions.

Several standard methods exist for performing a sensitivity analysis of a given model (Rose and Swartzman 1981, Straskraba and Gnauck 1985). We have adapted and extended these methods (Frech and Andres 1987, Walker 1987) to meet the needs of our concept assessment models, which pose special challenges because of their large numbers of parameters, pathways and nuclides, and because of their probabilistic nature.

The first step in analyzing the sensitivity of BIOTRAC was to perform separate analyses on each of its four submodels. This was a comparatively straightforward procedure because each submodel contains a relatively small number of parameters and has a well-defined mathematical structure. The complexity of the method used was chosen to match the complexity of the submodel. These analyses provided an understanding of the submodels. In particular, the sensitivity analysis of the soil model, SCEMR1, was used to develop the regression soil model used in BIOTRAC (Section 6.3.2). The results from the submodels were also useful for establishing analysis procedures for BIOTRAC itself, and for interpreting the results. Similarly, the results from BIOTRAC provided input for the sensitivity analysis of the combined vault, geosphere and biosphere models (Figure 1-3, Goodwin et al. 1994).

In this chapter, we discuss the sensitivity analysis of each submodel briefly, and then present the methods and results for BIOTRAC as a whole. We will identify the nuclides and pathways that significantly influence predictions, and quantify the response of the model to changes in all the sensitive parameters.

10.2 SENSITIVITY ANALYSIS OF THE SUBMODELS

Sensitivity analysis was used as a development tool to design and evaluate each submodel. Accordingly, the analyses were performed before the models had been combined in BIOTRAC and before BIOTRAC had been coupled to the geosphere and vault models. Actual source terms were therefore not available, and the analyses were done for a unit input. As a result no conclusions could be drawn regarding the actual importance of the various nuclides and exposure pathways. Instead, the emphasis was placed on identifying and ranking the parameters or pathways to which each submodel is most sensitive. In each case, the response was measured by varying one parameter at a time, with the others held constant at their mean or nominal values. Sensitivity was assessed relative to the predictions with all the parameters at their nominal values. The submodels are simple enough that results obtained by varying one parameter at a time are adequate, although there could be hidden parameter interdependences.

Different end points were used in the sensitivity analyses of the four submodels. The surface-water submodel used nuclide concentrations in the water column and mixed sediment, the soil submodel focussed on nuclide concentration in soil, the atmosphere submodel employed nuclide concentration in air, and the food-chain and dose submodel used radiological dose to humans. All these end points have a direct or indirect bearing on the protection of humans and other biota from radiological and chemical toxicity.

10.2.1 Surface Water Submodel

The sensitivity of the surface water submodel was investigated for each parameter in turn by assessing the response of the model to minor and major changes in parameter values. For the former, parameters were changed to 5% above and below the nominal or mean values (Section 5.5); for the latter, parameters were changed to the upper and lower 95% confidence limits. The model was driven by a constant nuclide input of 1 $\text{mol}\cdot\text{a}^{-1}$, and sensitivity was evaluated when the water and mixed-sediment concentrations had reached steady state. The constant input corresponds to the mass flow from compacted sediments into the lake water, χ^i (Section 4.4.2).

Nuclide concentrations in lake water, C_l^i, depend upon seven parameters: runoff, R, catchment area, A_d, lake area, A_l, mean lake depth, Z_l, water to sediment transfer rate, α^i, gaseous evasion rate, η_l^i (for volatile nuclides), and radioactive decay rate, λ^i (Equation 5-6). Steady-state mixed-sediment concentrations, C_{sed}^i, depend on these parameters and on the sedimentation rate, S_{sed}. We examined the response of the model to each of these parameters in turn. Since sediment transfer rates are element-specific (Section 5.5.8), we considered both a nonreactive element, calcium, and a reactive element, lead. Reactivity here refers to the tendency to attach to organic particles and thereby become removed from the water column. In addition, the analysis was performed for carbon to determine the model's sensitivity to variations in the gaseous evasion rate.

Bird et al. (1992) discuss the results of the sensitivity analysis in detail. Briefly, for long-lived, nonreactive, non-volatile nuclides, water concentrations are most sensitive to variations in A_d and R, which control the rate of hydrological flushing. This behaviour can be understood using Equation (5.19), which, for small α^i, η_l^i and λ^i, reduces to

$$C_l^i(t) = \chi^i(t)/(A_d \cdot R) \quad . \qquad (10.1)$$

For short-lived nuclides, or nuclides with high sedimentation (lead) or evasion (carbon) rates, the model also becomes sensitive to α^i, η_l^i and λ^i. In this case, lake area and depth become important because some of the loss mechanisms determined by these parameters depend on the volume of the lake. Overall, the results indicate that the model is most sensitive to the parameters involved in the process that dominates the loss of nuclides from the water column.

Nuclide concentrations in mixed sediment are always sensitive to the sedimentation rate, as is evident from Equation (5.22). For nonreactive non-volatile nuclides, the model also responds strongly to variations in A_d, R, Z_l and α^i. The lake area also becomes important for reactive nuclides. For short-lived nuclides, or nuclides with a substantial evasion rate, the sediment concentrations are sensitive to all the parameters, with the evasion rate having the least influence. The sensitivity to A_d, R and Z_l decreases as losses due to radioactive decay or evasion become large.

The total influence of a given parameter on model predictions depends not only on the response of the model to small changes in parameter values, as discussed above, but also on the range of values that the parameter can assume. Accordingly, we repeated the sensitivity analysis of the surface water model using the methods discussed above, but setting the values of each parameter in turn to its ±95% confidence limits based on its specified distribution. In Table 10-1, we list nominal ranges for each parameter based on the distributions discussed in Section 5.5. The minimum value, P_L, is the value two SDs below the mean, and the maximum value, P_U, is two SDs above it. The range of the values is expressed as the ratio P_U/P_L. Although the catchment area, A_d, is not a distributed parameter in BIOTRAC (Section 4.3), the range of values on the Canadian Shield is large. The values given in Table 10-1 are the minimum and maximum areas reported by Minns (1984).

TABLE 10-1

NOMINAL MAXIMUM AND MINIMUM PARAMETER VALUES FOR THE SENSITIVITY ANALYSIS OF THE SURFACE WATER MODEL

Parameter	Units	Minimum Value, P_L	Maximum Value, P_U	P_U/P_L
Runoff, R	$m \cdot a^{-1}$	0.15	0.47	3.1
Catchment Area, A_d	m^2	3.0×10^4	2.52×10^9	8.4×10^4
Lake Area, A_l	m^2	0.13	3.83×10^2	2.95×10^3
Lake Depth, Z_l	m	1.15	18.4	16
Sedimentation Rate, S_{sed}	$kg \cdot m^{-2} \cdot a^{-1}$	0.026	0.984	38
Transfer Rate, α^i	a^{-1}			
Calcium		2.91×10^{-5}	0.858	2.95×10^4
Lead		0.139	26.0	187
Carbon		0.1	48.6	486
Evasion Rate, η_l^i	a^{-1}			
Carbon		0.0312	27.1	869

After conducting simulations with each parameter value separately at its ±95% confidence limits, and all the other parameters set at their nominal values, it was found that A_d has the greatest potential to affect predicted nuclide concentrations in water. With the catchment area held constant, as is the case for the postclosure assessment (Section 4.3), reactive nuclides (high α^i values) are most sensitive to the lake area, A_l, and the sediment transfer rate, α^i. Nonreactive nuclides (low α^i values) are sensitive to variation in the runoff, R, and α^i. Uniquely, carbon is also sensitive to the gaseous evasion rate, η_l^c. Nuclide concentrations in mixed sediment are most sensitive to the catchment area, A_d, and the sedimentation rate, S_{sed}. However, with the catchment area held constant, other parameters are more important. Reactive nuclides are sensitive to A_l and S_{sed}, and nonreactive elements are sensitive to α^i.

At a specific site, the catchment area, lake area, lake depth, sedimentation rate and runoff could all be measured fairly precisely for present-day conditions. This suggests that improved databases for the transfer and evasion rates provide the best opportunities for reducing uncertainties in the predicted water and mixed-sediment concentrations. The response of the model revealed by the sensitivity analysis is entirely consistent with the observed behaviour of nuclides in aquatic systems. This inspires confidence that the model adequately represents the important processes, and is working as expected. Because the model has some sensitivity to all its input parameters, it cannot be simplified by reducing the number of parameters.

10.2.2 Soil Submodel

Sensitivity analysis played a major role in deriving the soil submodel from the detailed mechanistic SCEMR1 model (Section 6.3.2). The model, which takes the form of regression equations, includes only those parameters to which SCEMR1 predictions are most sensitive (Sheppard M.I. 1992). The methods used to analyze SCEMR1 and the results of the analysis (Sheppard M.I. and Bera 1984) are discussed briefly here.

SCEMR1 was analyzed using methods similar to those applied to the surface water submodel. The sensitivity of the model was determined for each input parameter in turn by comparing the predictions of a model simulation based on a standard set of nominal parameter values with the predictions of a simulation in which a single parameter value was altered. Most parameters were varied by a factor of 10 in both directions; where this range was unrealistically large, the parameter was varied over the range of values that occur on the Canadian Shield. Nuclides were assumed to enter the soil profile from below with contaminated groundwater (Section 4.4.3). For this analysis, SCEMR1 was run for 30 a in its pulse mode; the lowest soil layer was subject to a unit pore-water concentration at $t = 0$, after which time no further contaminant was added. Such an input produces a root-zone soil concentration that increases with time to a peak concentration, and then decreases. The response of the model was evaluated on the basis of the peak nuclide concentrations in the top soil layer.

We investigated the sensitivity of 25 SCEMR1 parameters describing various properties of the soil, the vegetation canopy and the climate (Section 6.5). The model responded strongly to variations in 13 of these parameters: soil type, soil depth, soil solid/liquid partition coefficient, annual effective precipitation, vapour pressure, saturated hydraulic conductivity, distribution of plant roots with depth, fraction of soil area occupied by roots, leaf area index (summer and winter values), leaf critical water potential (summer value), maximum leaf surface resistance (winter value) and resistance of litter to vapour loss. Because these parameters are all known to have a significant effect on soil concentrations, SCEMR1 appears to be working as expected. These results agree with those obtained from a sensitivity analysis of TEHM (from which SCEMR1 was developed), where over 250 parameters were varied (Begovich and Luxmoore 1979).

Only the first four of the sensitive parameters listed above (soil type (Section 6.5.1.1), soil depth, Z_s (Section 6.5.1.2), solid/liquid partition coefficient, Kd^i (Section 6.5.3), and annual effective precipitation, Pe (Section 6.5.2.2) were treated as distributed parameters in our submodel. The nominal values for four of the other parameters (vapour pressure, saturated hydraulic conductivity, leaf critical water potential and summer leaf area index) produced soil concentrations larger than or comparable to those predicted for either larger or smaller values of these parameters. Because these nominal values result in conservative predictions, these parameters were assigned fixed values in SCEMR1 (Section 6.5.6), and do not appear explicitly in the regression equations. Too little is known about four other parameters (fraction of soil area occupied by roots, winter leaf area index, maximum leaf surface resistance and resistance of litter to vapour loss) to define meaningful distributions for Canadian Shield conditions. Accordingly, these parameters were given reasonable fixed values in SCEMR1 and were not included explicitly in the regressions. The final parameter, depth distribution of plant roots, was also assigned a fixed value because all the other plant properties are defined generically and because our nominal value for this parameter is commonly accepted.

At a given site, the soil type, soil depth and effective precipitation can be characterized fairly easily. The greatest potential for reducing uncertainties in the predictions of the regression model therefore lies with the solid/liquid partition coefficients, Kd^i.

10.2.3 Atmosphere Submodel

All the nuclide concentrations in air are calculated from simple linear equations (Section 7.3). A formal analysis is therefore not required to identify the parameters to which the model is most sensitive. Instead, this was achieved by ranking the pathways contributing to the air concentrations, C_a^i. The model will necessarily be sensitive to the parameters associated with the important pathways. Mean values of the relevant parameters were used in these calculations, and the model was driven by unit nuclide concentrations in the soil, C_s^i, and water, C_w^i, compartments. For this reason, concentrations could be compared only for pathways with a common source, either terrestrial or aquatic. Because different nuclides reach the atmosphere via different pathways, the analysis was done for different groups of nuclides. The results are reported in detail by Amiro (1992b) and summarized briefly below.

For non-volatile nuclides, air concentrations from terrestrial sources are made up of nuclides suspended from the soil and released through fires (Section 7.3). For plant/soil concentration ratio, Bv^i (Section 8.5.1.1), values greater than about one, air concentrations are dominated by the fire pathways. As Bv^i decreases, soil suspension becomes relatively more important. The concentrations from terrestrial sources are therefore sensitive to all the parameters appearing in the fire and particulate suspension models. Non-volatile nuclides from aquatic sources can reach the atmosphere only through particulate suspension, so that concentrations via this pathway are sensitive to the aquatic atmospheric dust load, AADL (Section 7.5.1.2).

Indoor radon concentrations (Section 7.3.4.1) are dominated by release from domestic water. Outdoor radon concentrations are determined largely by gaseous and particulate releases from soil and evasion from the lake, although land-clearing fires can also contribute if the soil type is organic (Section 6.5.1.1).

Gaseous evasion from the soil and the lake are the main contributors to ^{14}C and ^{129}I concentrations in outdoor air (Section 7.3.4). Indoor concentrations of these nuclides are also influenced by releases from domestic water, e.g., during the use of humidifiers.

A further output of the atmosphere model is the rate at which nuclides are deposited to soil, D_s^i, and vegetation surfaces, D_b^i (Section 7.3.8). For mean values of the relevant parameters, wet and dry deposition make about equal contributions to the total flux. Thus the model is sensitive to variations in the dry deposition velocity, Vd (Section 7.5.4.1), the washout ratio, Wr (Section 7.5.4.2), and the total annual precipitation, P (Section 9.1.3).

The parameters appearing in the atmosphere model vary over very different ranges (Section 7.5), and so have greater or lesser potential for affecting air concentrations. The normally distributed parameters (radon aquatic transfer coefficient, ATC^{Rn} (Section 7.5.4.1), wind speed, UCAV (Section 7.5.3.1), and wind speed weighting factor, UWGHT (Section 7.5.3.2)), have narrow PDFs and a restricted range. The dispersion factors for ground-level area sources also have a limited range because they are very weak functions of the area of the contaminated source (Section 7.3.1). The lognormally distributed parameters show a much greater variability. In Table 10-2, we list nominal ranges for each lognormally distributed parameter using the definitions of minimum, P_L, and maximum, P_U, values established in Section 10.2.1.

The parameters quantifying the transfer rates of gaseous nuclides (evasion rates, η^i_l (Section 5.5.9) and η^i_s (Section 6.5.4), aquatic iodine mass loading parameter, AIML (Section 7.5.1.5), indoor radon transfer coefficient, INDRN (Section 7.5.1.6), and radon emission rate, q^{Rn} (Section 7.5.1.3)) exhibit by far the largest P_U/P_L ratios. The deposition velocity also varies considerably.

The pathways contributing to the air concentration vary, depending on the nuclide of interest and the contaminant source. However, all the pathways included in the model contribute to some extent to some combination of nuclide and source. No pathways (or parameters) can therefore be dropped from the model. The special suspension mechanisms (fires, gaseous evasion and indoor releases) make important contributions to the air concentrations and are thus important in the model.

10.2.4 Food-Chain and Dose Submodel

CALDOS has not undergone a formal sensitivity analysis because the model is mainly expressed in terms of linear, multiplicative chain equations (Section 8.3) and its response to variations in parameter values can be inferred without the need for a formal analysis. However, the version of the model used in the first interim assessment, FOOD III (Mehta 1985), was analyzed extensively (Zach 1980b).

Basically, each of the many exposure pathways in CALDOS is driven by the nuclide concentration in either soil, C^i_s, water (lake or well), C^i_w, or air (indoor or outdoor), C^i_a. The following discussion assumes a unit nuclide concentration in each source compartment and man's dose as the end point. The contributions to dose of the various pathways associated with a given compartment depend upon the nuclide of interest. A given pathway is important for some nuclides and not for others, but all pathways contribute significantly for some nuclides. Similarly, all nuclides with a half-life of a few days or more contribute to the dose via one pathway or another. The total dose tends to be dominated by the internal exposure pathways (Section 8.3.1) rather than the external pathways (Section 8.3.2). For

TABLE 10-2

NOMINAL MAXIMUM AND MINIMUM VALUES FOR LOGNORMALLY DISTRIBUTED PARAMETERS FOR THE SENSITIVITY ANALYSIS OF THE ATMOSPHERE MODEL

Parameter	Units*	Minimum Value, P_L	Maximum Value, P_U	P_U/P_L
Atmospheric Dust Load, ADL	kg soil·m^{-3} air	2.97×10^{-8}	1.17×10^{-7}	3.95
Aquatic Atmospheric Dust Load, AADL	m^3 water·m^{-3} air	1.46×10^{-10}	5.77×10^{-10}	3.95
Aquatic Iodine Mass Loading Parameter, AIML	m^3 water·m^{-3} air	3.28×10^{-7}	5.16×10^{-4}	1.57×10^3
Radon Emission Rate, q^{Rn}	(mol Rn·m^{-2}·s^{-1})/ (mol Ra·kg^{-1} soil)	5.79×10^{-10}	1.26×10^{-8}	21.8
Indoor Radon Transfer Coefficient, INDRN	(mol Rn·m^{-3} air)/ (mol Ra·kg^{-1} soil)	2.33×10^{-6}	7.95×10^{-4}	3.41×10^2
Evasion Rates from Soil, η_s^i (^{129}I and ^{79}Se)	s^{-1}	10^{-11}	10^{-7}	10^4
Evasion Rate from Soil, η_s^c (^{14}C)	s^{-1}	2.78×10^{-9}	2.78×10^{-5}	10^4
Evasion Rate from Lake (^{14}C), η_l^c	s^{-1}	9.84×10^{-10}	8.55×10^{-7}	8.69×10^2
Forest Yield, FY	kg·m^{-2} land	0.86	5.63	6.55
Building Width, BW	m	8.4	14.0	1.66
Deposition Velocity, Vd	m·s^{-1}	1.5×10^{-3}	2.4×10^{-2}	16
Washout Ratio, Wr	unitless	1.0×10^5	6.24×10^5	6.24

* For q^{Rn} and INDRN, Rn refers to ^{222}Rn and Ra to ^{226}Ra.

this reason, the model is very sensitive to variations in the ingestion DCF, DFe^i (Section 8.5.2.1), which appears in all internal pathways except inhalation.

Geometric standard deviations and P_U/P_L ratios (Section 10.2.1) for the lognormally distributed parameters appearing in CALDOS are listed in Table 10-3. Values of P_L and P_U, which are element-specific, are not shown individually in order to keep Table 10-3 to a manageable size. The aquatic concentration ratio, B_j^i (Section 8.5.1.3), and the plant/soil concentration ratio, Bv^i (Section 8.5.1.1), have by far the widest ranges, and therefore the greatest potential for affecting predicted doses. The terrestrial animal transfer coefficients, F_j^i (Section 8.5.1.2), also show a considerable range. For each of these parameters, the variability largely reflects the uncertainty in the database used to derive the distributions and in the transfer processes that the parameters describe. The ranges of the remaining parameters (plant environmental halftime, tp (Section 8.5.3.3), and food type energy weighting factors, Ycf_j (Section 8.5.6.4)), are relatively small, and are caused primarily by natural variability in the environment and human diet.

TABLE 10-3

GEOMETRIC STANDARD DEVIATIONS FOR THE LOGNORMALLY DISTRIBUTED PARAMETERS APPEARING IN CALDOS

Parameter	GSD	P_U/P_L
Plant/soil concentration ratio, Bv^i ((Bq·kg^{-1} wet biomass/(Bq·kg^{-1} dry soil)) (all nuclides)	10	10^4
Aquatic concentration ratio, B_j^i (L water·kg^{-1} wet biomass) (all nuclides)	12	2.1×10^4
Terrestrial animal transfer coefficients, F_j^i (d·L^{-1} or d·kg^{-1} wet biomass) (all nuclides and animal types)	3.2	1.05×10^2
Environmental halftime, tp (d)	2.0	16
Food type energy weighting factors, Ycf_j		
TE PLANT	1.65	7.4
TE MILK	1.35	3.3
TE MEAT	1.65	7.4
TE BIRD	1.65	7.4
FW FISH	4.48	4.0×10^2

10.3 SENSITIVITY ANALYSIS OF BIOTRAC

As shown in Figure 9-4, the BIOTRAC model, with its numerous pathways, is complex. It is not immediately obvious which nuclides, pathways and parameters are most important in determining consequences. This was established through a rigorous sensitivity analysis of the complete integrated model. As a measure of the response of BIOTRAC, the most obvious end point is the total dose to man. However, doses from individual nuclides or from specific pathways for a given nuclide are also useful end points. We first investigated the nuclides that contribute most to the total dose, followed by the pathways that dominate the dose for each of these nuclides, and then the parameters that influence this dose along each of these pathways. These doses are discussed in Section 8.3.3. We also identified the parameters that have the most influence on the total dose, irrespective of nuclide and pathway. For convenience, we designate the various pathways by the parts of the model through which the nuclides flow. For example, air/plant/man indicates the pathway in which nuclides are deposited from the air to vegetation, which is then ingested by humans as TE PLANT. Although we used radiological dose to humans as the end point, results of our analysis are also relevant for radiological doses to other biota and for chemical toxic effects.

A standalone, quality-assured version of BIOTRAC was used for the sensitivity analysis. This version employs a single input time series, representing a simplified but realistic geosphere output. The sensitivity analysis of the entire system model combining the vault, geosphere and biosphere models, and using the actual geosphere output time series from the postclosure assessment, is reported in Goodwin et al. (1994).

10.3.1 Methods

Because BIOTRAC contains a large number of distributed parameters, the values selected in one model simulation can describe a much different system than the values chosen in another simulation. The response of the model to changes in the value of a particular parameter may then differ from simulation to simulation. A stochastic sensitivity analysis is required to assess the overall response of the model. A large number of simulations were performed with our standalone version of BIOTRAC (five cases with 1000 simulations each, amounting to a combined case of 5000 simulations), in which all the distributed parameters were allowed to vary simultaneously. The overall response of the model to a given parameter was then determined by comparing the output for those simulations in which a parameter took on relatively high and low values. We have not tested the response of the model to variations in parameters such as the DCFs, which are not distributed (Section 8.5.2). Our ranking of parameters therefore defines the relative importance of the distributed parameters only. These parameters have distributed values because of their high degree of variability, uncertainty and importance (Section 2.7). Furthermore, the sensitivity analysis of the combined vault, geosphere and biosphere models involves perturbation of all the parameter values, whether they are distributed or not (Goodwin et al. 1994).

Preliminary sensitivity analyses of BIOTRAC were done with actual time series inputs predicted by the geosphere model, run with nominal or median parameter values. This tested our procedures and yielded reproducible results (Reid J.A.K. and Corbett 1992); however, it made our analyses directly dependent on output from the geosphere model. So, to make the BIOTRAC sensitivity analysis independent, the final analysis was performed using simplified, but realistic input that reflects the predicted release of nuclides from the geosphere model.

Several inputs were tested and a time series was chosen that has a quick, straight-line buildup to a constant geosphere discharge at 10 a, as shown for ^{14}C, ^{129}I and ^{99}Tc in Figure 10-1. The discharge then continues at

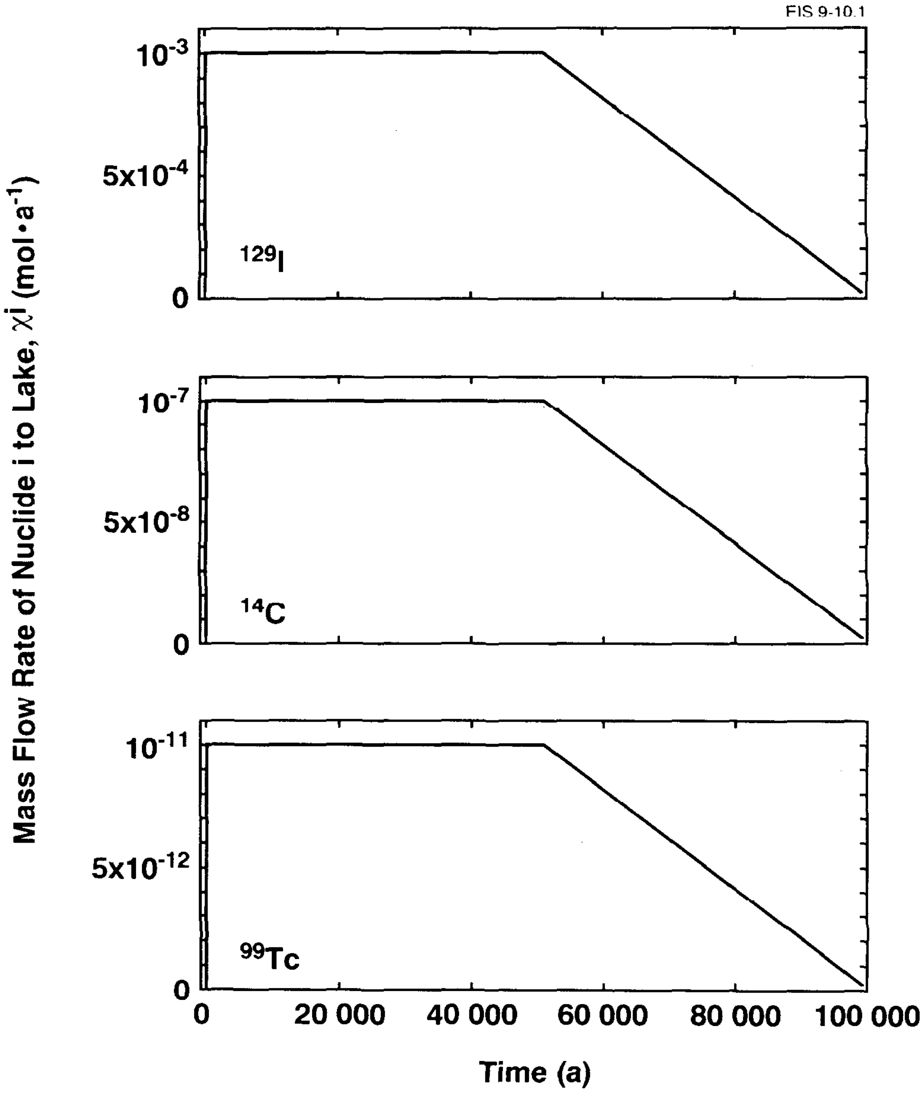


FIGURE 10-1: Simplified Input Time Series to BIOTRAC (Boggy Creek South Discharge Zone and Well) for ^{129}I, ^{14}C and ^{99}Tc Used to Drive the Surface Water Model for the Stochastic Sensitivity Analysis

levels of the same order of magnitude, the same as peak geosphere discharges from typical complete system simulations, followed by a straight-line decline beginning at 50 000 a. This produced realistic sensitivity results, comparable to those from the combined models, without direct connection to the vault and geosphere models, and gave us confidence that we could proceed with a meaningful sensitivity analysis using BIOTRAC alone.

The input time series shown in Figure 10-1 correspond to the nuclide mass flow, χ^i (mol·a^{-1}), and they were used to drive the surface-water model (Section 5.3). These series include the flows used to calculate well-water concentrations (Section 4.4.4), and have the same basic shape as the χ^i series, but a lower maximum value. For each nuclide, the maximum value for the well-water series was estimated from the output of the combined vault and geosphere models. In order to calculate well-water concentrations in BIOTRAC, the volume of water, V_{1d} (m^3 water·a^{-1}), drawn from the lake into the well is required from GEONET. Without direct GEONET input, we used a constant V_{1d} value of 40 m^3 water·a^{-1}.

10.3.1.1 Ranking Nuclides, Pathways and Parameters

A variety of methods was used to rank the importance of the nuclides, pathways and parameters. Because doses are additive over nuclides and pathways (Figure 10-2), simple scanning of mean doses can be used to identify the important nuclides and pathways. Multiple linear regression (MREG) of the log-transformed doses was used to rank the parameters, to allow us to investigate the multiplicative relationships between parameters and dose, and among parameters. We found this method to be reliable, reproducible and comparable with other methods used. The importance of parameters was examined for total dose, nuclide doses and pathway doses for a given nuclide. We identified about 15 parameters that are most important in determining the total dose. In the case of individual pathway doses, the number of important parameters varied with the pathway and nuclide in question.

We used the forward selection procedure of logarithmic MREG (Sokal and Rohlf 1981). The basis of this method is to start with a dose of interest, identify the parameter that explains the largest proportion or fraction of the variability associated with that dose, and then add other parameters, one at a time, in order of importance until the addition of further parameters no longer produces an appreciable increase in the variability accounted for. An appreciable increase is defined through the C(p) statistic proposed by Mallows (Mallows 1973, SAS 1985), which declines with the addition of each new parameter to the MREG model. When the number of parameters equals C(p), essentially all the variability has been explained and no additional parameters will cause significant increases.

10.3.1.2 Measures of Parameter Importance

To measure parameter importance we used a method called SENSYV (Hoffman et al. 1984b). For each parameter, two sets of simulations were extracted from the total number of simulations in a case, one containing the 10% of simulations where a parameter took on its lowest values and the other the 10% of simulations where it had its highest values. The geometric mean of the total dose from the lower 10% of simulations, GM_L (Sv·a^{-1}), is then

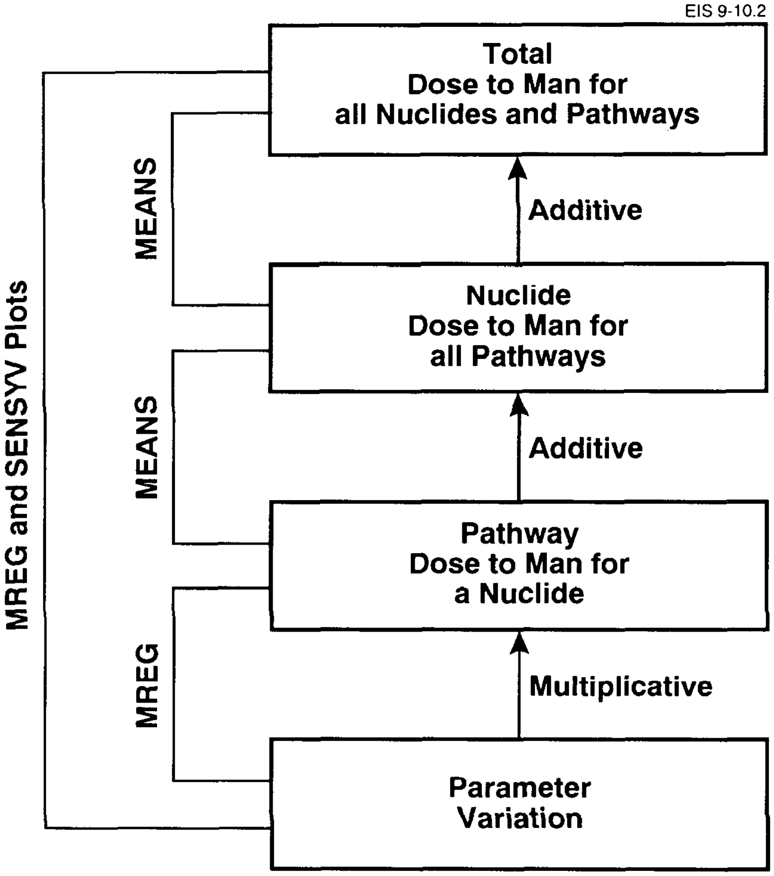


FIGURE 10-2: Conceptual Diagram of the Formation of the Total Dose to Man as the Sum of the Nuclide and Pathway Doses, and Related Methods Used for Sensitivity Analysis. The pathway doses are shown to be multiplicative combinations of the parameter values. The methods used to establish the importance of the various pathways are: MEANS - comparison of mean doses, MREG - logarithmic multiple regression, and SENSYV plots - plots of SENSYV factor, Sf (Section 10.3.1.2).

calculated, as is the GM from the upper 10%, GM_U. The SENSYV factor, Sf (unitless), is then

$$Sf = GM_L/GM_U \tag{10.2}$$

as shown schematically in Figure 10-3. Sf is the factor by which the mean total dose increases or decreases between sets of simulations with low and high values for a parameter. For example, an Sf value of 2.0 indicates a twofold increase in dose, on average, from the lower 10% to the upper 10% of the parameter values. This method was developed to analyze results from the second interim assessment (Hoffman et al. 1984b) and has been tested and shown to be complementary to other ranking techniques.

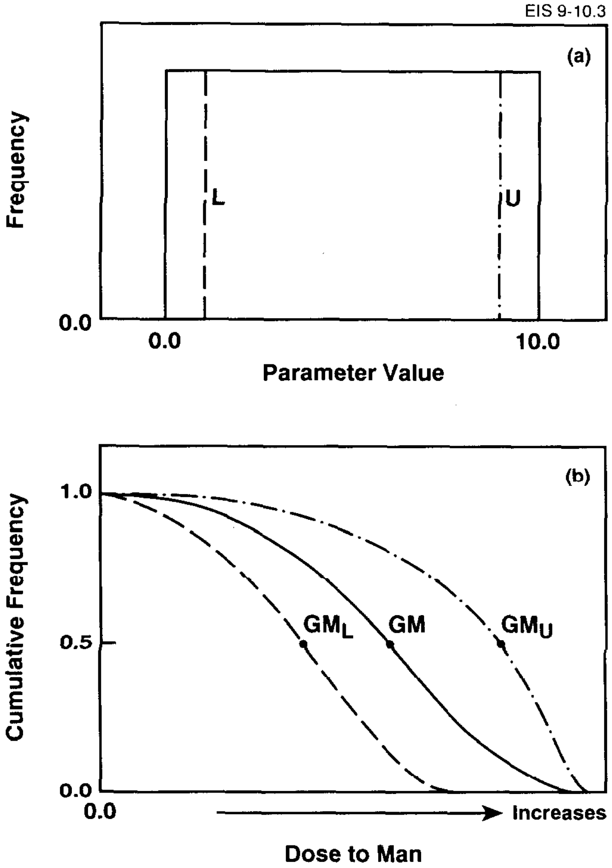


FIGURE 10-3: Plots of Frequency Versus Parameter Value for (a) a Simple Uniformly Distributed Parameter and (b) the Resulting Cumulative Frequency Distribution for the Dose to Man. The geometric mean dose, GM, is calculated for the full range of the parameter and then for simulations where the parameter's values are restricted to the lower, GM_L, and upper, GM_U, segments of the range. GM_L/GM_U represents the SENSYV factor, Sf.

The SENSYV technique was extended by calculating Sf using GM_L and GM_U values calculated from different percentages, fs (unitless), of the total number of simulations in which a parameter took on its lowest and highest values.

The Sf values were then plotted against the total number of simulations used in the case (SENSYV plot). To put the magnitude of Sf into perspective, SENSYV plots were established for the perfect predictor of total dose, the dose itself, and for a random variable known to be totally unrelated to dose, the simulation number. In this way the importance of a parameter of interest can be displayed over its range of values and contrasted with both a perfect and an unimportant predictor.

10.3.1.3 Reliability of the Methods

Our sensitivity analysis methods have all been tested extensively using BIOTRAC, and have been shown to produce consistent results when compared with other methods, including fractional factorial and rank correlation analysis (Reid J.A.K. and Corbett 1992). Our methods have been shown to consistently predict the top three to five parameters in the same order of importance. They can also identify a further ten or twelve parameters as being important, but cannot rank them consistently. This level of confidence can be reached with a single 1000-run case. Experience has shown that approximately 3000 to 5000 simulations of BIOTRAC are needed to rank the top dozen or so parameters in an order that can be reproduced (Reid J.A.K. and Corbett 1992).

10.3.2 Results: Sources of Total Dose To Man

We investigated the sources of the total dose to man by looking first at the dose attributable to each nuclide. We then examined the pathways that are most important for each nuclide, and identified the parameters that govern each pathway. Finally, we investigated the parameters that determine the total dose to man.

10.3.2.1 Nuclides

Table 10-4 shows the mean percent of the total dose to man by nuclide at 10 000 a from a typical 1000-run case in which all the distributed BIOTRAC parameters were allowed to vary simultaneously. These results show that ^{129}I dominates the dose. This is true at 10 000 a and also at shorter times. Besides ^{129}I, only ^{14}C makes a significant contribution. All the other radionuclides make negligible contributions, substantially below that of ^{99}Tc. This ranking, which reflects the results from the postclosure assessment (Goodwin et al. 1994), is partly caused by the difference in the size of the input from the geosphere for each nuclide (Figure 10-1). In terms of subsequent sensitivity analyses results it is important to note that ^{14}C and ^{129}I are volatile nuclides, whereas ^{99}Tc is not. So the three radionuclides exercise all the pathways of BIOTRAC.

TABLE 10-4

NUCLIDE IMPORTANCE MEASURED BY CONTRIBUTION TO THE TOTAL DOSE TO MAN

Nuclide	Percent of Total Dose
^{129}I	95.8
^{14}C	4.19
^{99}Tc	3.10×10^{-4}

10.3.2.2 Pathways and Parameters

The importance of the pathways along which nuclides can move to man (Figure 9-4) depends on the nuclide investigated. Thus, pathway results are discussed on a nuclide-specific basis using nuclide doses at 10 000 a. The importance of various parameters affecting pathway doses was assessed using logarithmic MREG (Section 10.3.1.1). We used results from the same 1000-run case discussed in Section 10.3.2.1 for these analyses.

Iodine-129

The mean contribution of the various exposure pathways to the ^{129}I dose is indicated in the first bar chart in Figure 10-4. It shows that the soil/plant/man pathway (Section 8.3.1.1) is most important, followed by the air/plant/man (Section 8.3.1.3), water/man (ingestion) (Section 8.3.1.8), and soil/plant/milk/man pathways (Section 8.3.1.2). By considering only the important pathways (Figure 10-5), the pattern of ^{129}I flows to man can be shown much more simply than in Figure 9-4. Note that the air/plant/man pathway includes volatilization from the lake and irrigation.

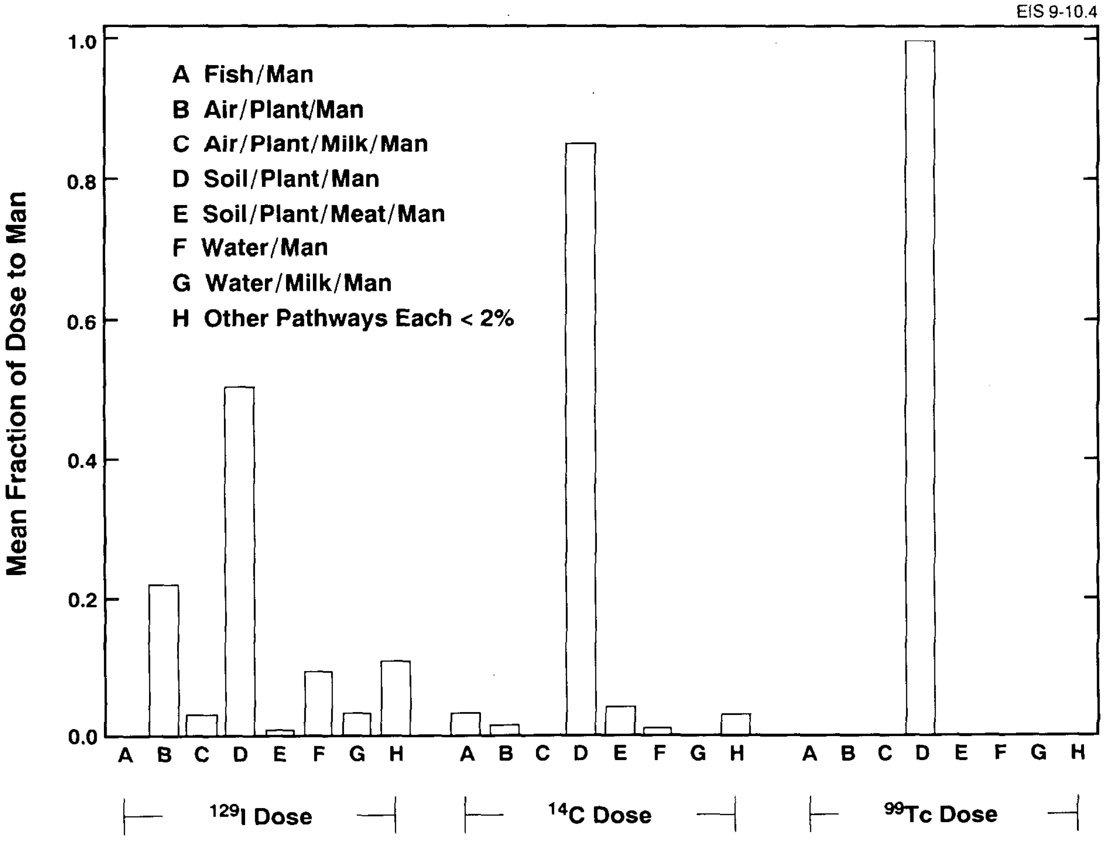


FIGURE 10-4: Bar Charts Showing Mean Fractions of the Dose at 10 000 a to Man From Various Pathways for ^{129}I, ^{14}C and ^{99}Tc for the Input Time Series Shown in Figure 10-1

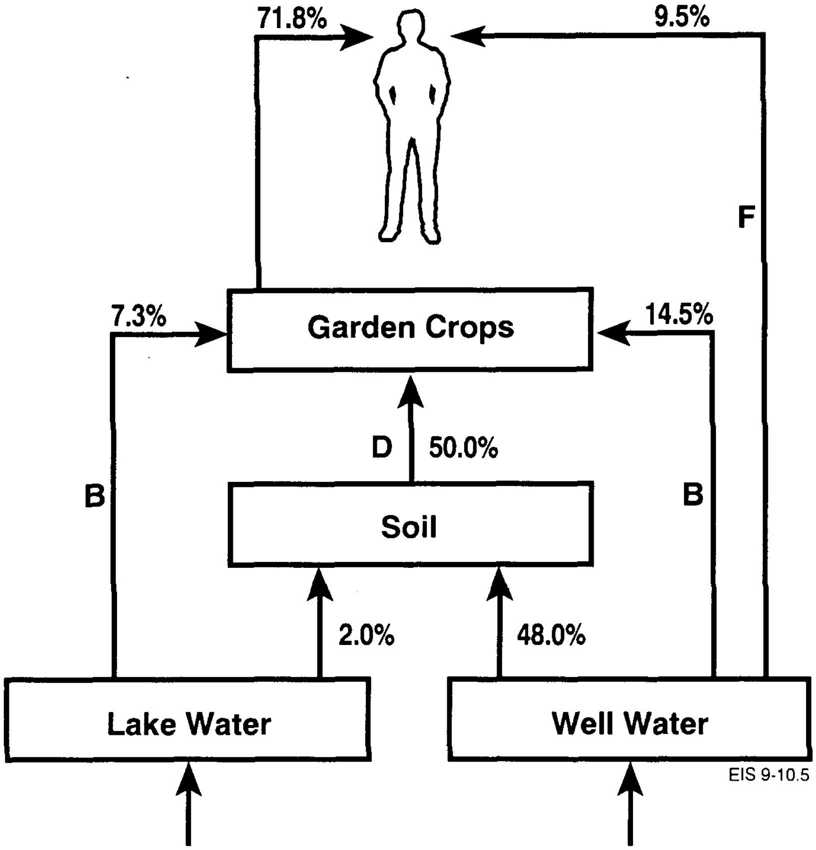


FIGURE 10-5: Iodine-129 Flow Diagram Indicating the Most Important Pathways to Man and the Percent Contribution to the ^{129}I Dose. B - air/plant/man (7.3% volatilization and 14.5% irrigation), D - soil/plant/man and F - water/man (ingestion) pathways.

For the three most important ^{129}I pathways (Figure 10-4), the lake/well-water switch, LW (Section 9.1.2), is by far the most significant parameter (Figure 10-6), particularly in the case of man's drinking water. The reason for this is that well water tends to be much more contaminated than lake water. For the air/plant/man pathway, the garden irrigation switch, PI (Section 6.5.5.2), is also relatively important because watering the garden can cause increased contamination of crops, depending on the level of water contamination. For the soil/plant/man pathway, the plant/soil concentration ratio, Bv_i (Section 8.5.1.1), and the gaseous evasion rate from soil, η_s^i (Section 6.5.4), emerge as important parameters. The importance of Bv_i is related to its very high variability and this is also so for η_s^i. Both these parameters have very high GSDs with a value of 10. Note that the aquatic mass loading parameter, AIML (Section 7.5.1.5), is the second most important parameter in the air/plant/man pathway.

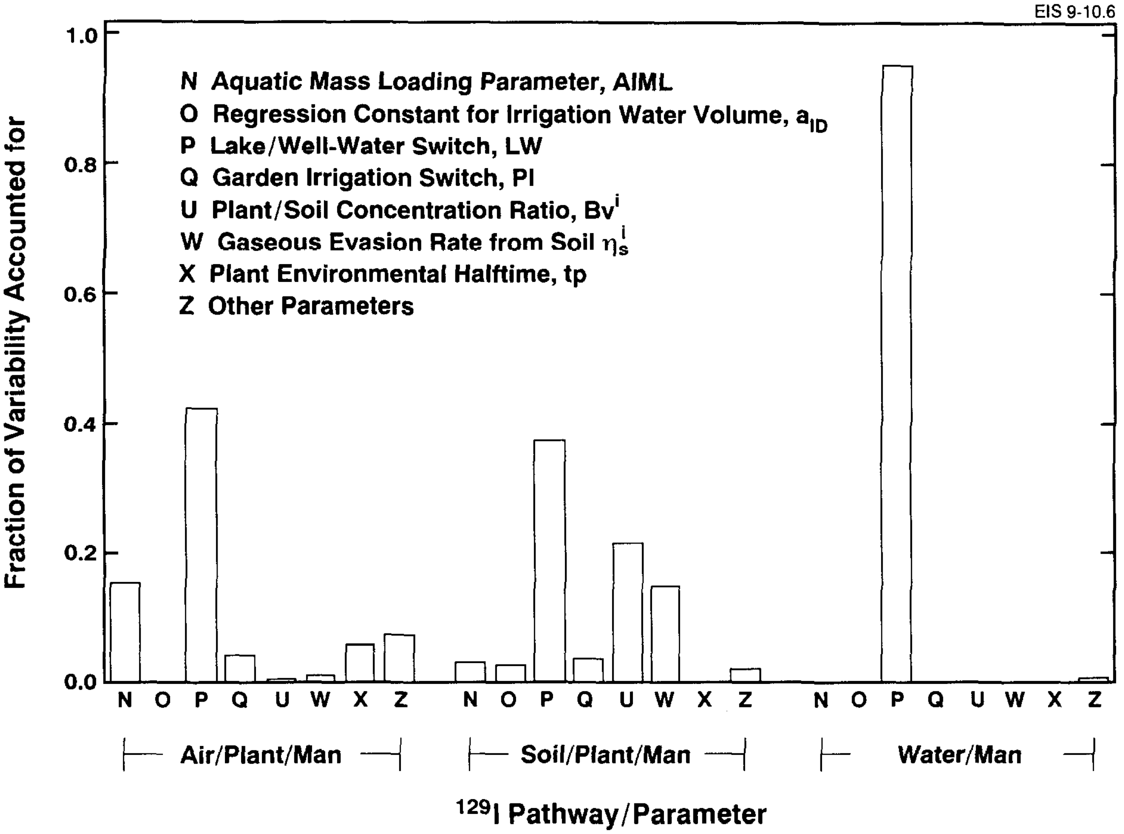


FIGURE 10-6: Bar Charts Showing the Fractions of the Variability in the ^{129}I Dose to Man From the Parameters Governing the Most Important Pathways

It must be recognized that the importance of a parameter is not always simple to describe because there are many parameter interactions. For example, LW would not be important in the soil/plant/man pathway if the garden is not irrigated.

Carbon-14

The three most important pathways for ^{14}C are soil/plant/man (Section 8.3.1.1), soil/plant/meat/man (Section 8.3.1.2), and fish/man (Section 8.3.1.7), as shown in Figure 10-4. Of these, the first pathway is totally dominant. Other pathways of very minor importance are air/plant/man, soil/plant/bird/man, water/man (ingestion), water/meat/man and soil/plant/milk/man. The main nuclide flow to man for ^{14}C is shown in Figure 10-7.

The soil/plant/man pathway is dominant for ^{14}C (Figure 10-4), and the most important parameter is the lake/well-water switch, LW (Section 9.1.2) as shown in Figure 10-8. The reasons for this are that well water tends to be much more contaminated than lake water, and watering of the garden can substantially enhance contamination of crops. This in turn is reflected in

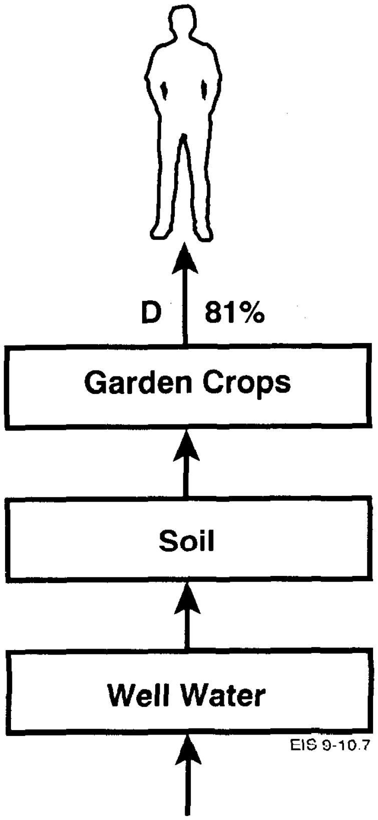


FIGURE 10-7: Carbon-14 Flow Diagram Indicating the Most Important Pathway to Man and the Percent Contribution to the ^{14}C Dose. D - soil/plant/man pathway.

high dose variability. The gaseous evasion rate from soil, η_s^i (Section 6.5.4), and the plant/soil concentration ratio, Bv^i (Section 8.5.1.1), are also important for reasons similar to those given for ^{129}I.

Technetium-99

The bar chart in Figure 10-4 indicates that the soil/plant/man pathway (Section 8.3.1.1) is the only important pathway for ^{99}Tc. Thus, the pathways diagram for this nuclide (Figure 10-9) shows a single route of transfer from well water to garden soil to garden crops and ultimately to man. This simple pattern of flow to man is the same as that for ^{14}C (Figure 10-7).

As in the case of ^{129}I and ^{14}C, the lake/well-water switch, LW (Section 9.1.2), is the most important parameter determining ^{99}Tc doses (Figure 10-10). The reasons for this are the same as those indicated above for ^{14}C. Given these similarities, it is not surprising that the plant/soil concentration ratio, Bv^i, with its large GSD of 10 is also important.

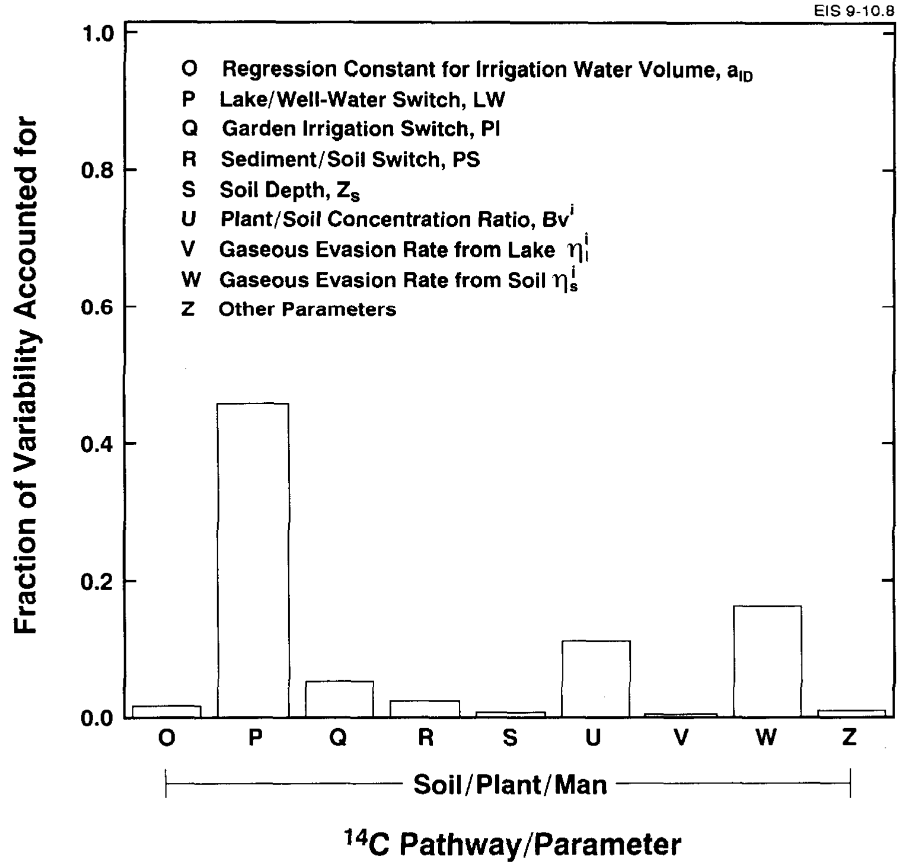


FIGURE 10-8: Bar Chart Showing the Fractions of the Variability in the ^{14}C Dose to Man From the Parameters Governing the Most Important Pathway

Gaseous evasion from soil is unimportant for ^{99}Tc because this nuclide is not volatile, as are ^{129}I and ^{14}C. So the third most important parameter is the irrigation switch for the garden, PI (Section 6.5.5.2), which determines whether or not the garden is watered.

10.3.3 Results: Individual Parameters Predicting Total Dose

The overall importance of individual parameters, irrespective of nuclide and pathway, was assessed by first ranking the parameters according to contribution to the total dose to man. This was followed by measuring the importance of the most significant parameters, using the SENSYV technique (Section 10.3.1.2).

10.3.3.1 Ranking the Parameters

Logarithmic MREG was applied to each of the five 1000-simulation cases individually, and also to the combined 5000-simulation case. In each instance, the same three parameters emerged as most important. They were, in decreasing order of importance (Table 10-5),

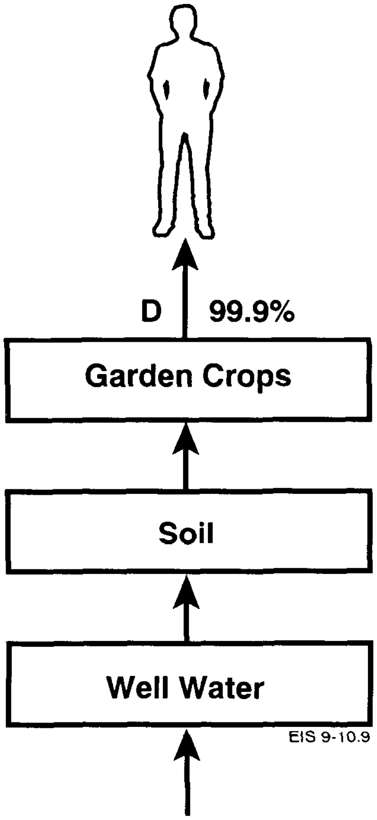


FIGURE 10-9: Technetium-99 Flow Diagram Indicating the Most Important Pathway to Man and the Percent Contribution to the Total ^{99}Tc Dose. D - soil/plant/man pathway.

- lake/well-water switch, LW,
- aquatic iodine mass loading parameter, AIML, and
- iodine gaseous evasion rate from soil, η_s^i.

These parameters were indicated in the same order of importance in all five 1000-run cases. As well, this order was maintained when the five cases were analyzed together. In all instances, the three parameters accounted for about 70% of the variation in total dose (Table 10-5). The SENSYV factor, Sf, based on the lower and upper 10% of the simulations, also reflected the same order of parameter importance (Table 10-5).

The next nine parameters in Table 10-5 explain about 7% of the variation in the total dose. However, the ranks are no longer consistent among the 1000-run cases, as expressed in relation to the 5000-run case. This is also true for the remaining parameters that individually explain a negligible amount of variation, but combined account for about 23% of it. The results in Table 10-5 show that even small proportions in explained variation can be significant, as judged by Mallows' test. This is particularly so in the 5000-run case with its larger sample size.

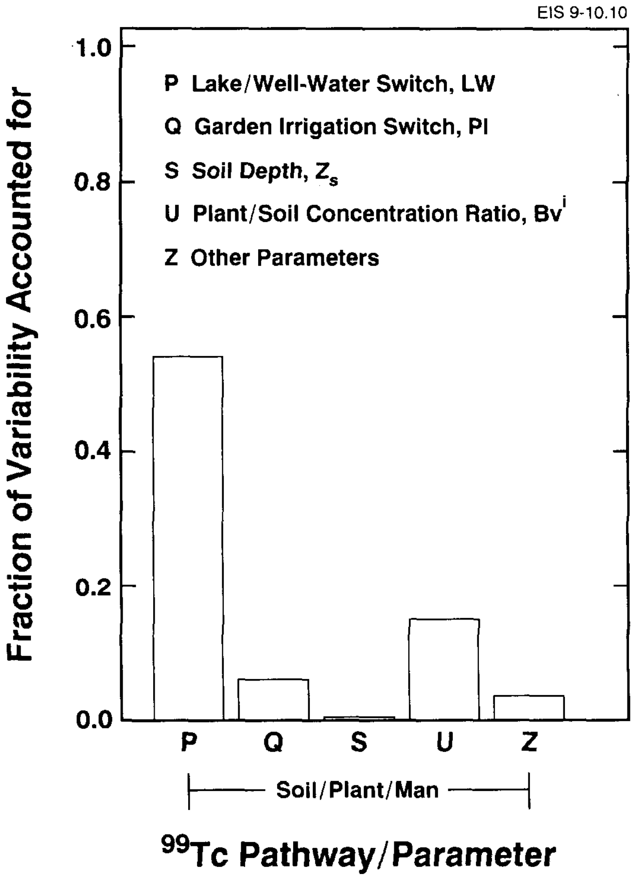


FIGURE 10-10: Bar Chart Showing the Fractions of the Variability in the ^{99}Tc Dose to Man From the Important Parameters Governing the Most Important Pathway

10.3.3.2 SENSYV Plots

Using the combined 5000-run case, we plotted the shift in total dose at 10 000 a indicated by the SENSYV factor, Sf, for each of the three most important parameters identified by logarithmic MREG analysis (Figure 10-11). Sf values for the aquatic iodine mass loading parameter, AIML, and for the lake/well-water switch, LW, are greater than one, indicating increased dose with increasing parameter value. Sf values for the iodine gaseous evasion rate from soil, η_s^i, are less than one, indicating decreased dose with increasing parameter value. The significance of Sf for each of these parameters can be evaluated by comparing their curves to those for the dose itself (perfect predictor), simulation number (random predictor) and the reference line (zero sensitivity).

These comparisons show that none of the three most important parameters is a very close predictor of total dose, but they do much better than the random predictor. Furthermore, LW is a better predictor than AIML, which is better than η_s^i, as indicated by the positions of the curves relative

TABLE 10-5

SUMMARY OF LOG MULTIPLE REGRESSION RESULTS SHOWING THE MOST SIGNIFICANT PARAMETERS FOR THE COMBINED 5000-SIMULATION CASE AND FOR THE FIVE SEPARATE 1000-SIMULATION CASES (A TO E)

Description	Defined in Section	Param. Symbol	Ranks (R) and Proportion of Variation Explained (Prop)													*Sf* Factor
			R	5000 Prop	A R	1000 Prop	B R	1000 Prop	C R	1000 Prop	D R	1000 Prop	E R	1000 Prop	Mean Rank	
Lake/Well-Water Switch	9.1.2	LW	1	0.5639	1	0.5527	1	0.5340	1	0.5527	1	0.5562	1	0.5890	1	40
I Mass Loading Lake to Air	7.5.1.5	AIML	2	0.0858	2	0.1043	2	0.0927	2	0.1043	2	0.0742	2	0.0819	2	20
I Evasion Rate Soil to Air	6.5.4	η_S^i	3	0.0434	3	0.0358	3	0.0473	3	0.0358	3	0.0480	3	0.0478	3	0.1
I Plant/Soil CR	8.5.1.1	Bv^i	4	0.0257	4	0.0282	4	0.0334	4	0.0282	5	0.0260	4	0.0245	4	
Sediment/Soil Switch	6.5.5.4	PS	5	0.0116	12	0.0021	5	0.0164	6	0.0075	4	0.0280	5	0.0093	6.4	
Garden Irrigation Switch	6.5.5.2	PI	6	0.0082	6	0.0083	6	0.0065	5	0.0117	7	0.0063	6	0.0077	6	
I Solid/Liquid Partition Coef	6.5.3	Kd^i	7	0.0057	5	0.0113	7	0.0047	7	0.0069	6	0.0084	*38	0.0003	7	
Stable I Conc in Groundwater	8.5.9.5	C_{gw}^{sI}	8	0.0041	7	0.0072	8	0.0032	11	0.0047	11	0.0039	10	0.0018	9.4	
Dry Deposition Velocity	7.5.4.1	Vd	9	0.0034	8	0.0031	14	0.0015	8	0.0056	15	0.0024	8	0.0045	13.2	
I Transfer Coef TE MILK	8.5.1.2	F_j^i	10	0.0035	9	0.0031	16	0.0015	9	0.0055	13	0.0026	7	0.0055	13.5	
Plant Env Halftime	8.5.3.3	tp	11	0.0023	13	0.0023	15	0.0015	12	0.0030	8	0.0047	11	0.0011	11.8	
Washout Ratio	7.5.4.2	Wr	12	0.0019	*22	0.0010	12	0.0019	10	0.0049	14	0.0025	13	0.0009	12.25	
Lake Area	5.5.3	A_l	13	0.0017	10	0.0033					17	0.0010		0.0037	12	
Well Depth	4.4.4	D_w	14	0.0017			13	0.0017	13	0.0021	9	0.0044	15	0.0007	12.5	
Sediment Transfer Rate	5.5.8	α^i	15	0.0014	15	0.0019	18	0.0013	16	0.0013	20	0.0015	*26	0.0006	16.8	
Runoff	5.5.4	R	16	0.0013	17	0.0019	*22	0.0008	23	0.0007	12	0.0029	*18	0.0006	18.75	
Plant Yield Food Crops	8.5.8.1	Y_j	17	0.0012	*22	0.0010	10	0.0024			10	0.0042			12.33	
Soil Depth	6.5.1.2	Z_s	18	0.0009	11	0.0026			18	0.0011	18	0.0015			15.67	
Food Energy Factor TE PLANT	8.5.6.4	Ycf_j	19	0.0008			11	0.0024	22	0.0008	*24	0.0008	*21	0.0006	19.5	
C Aquatic CR	8.5.1.3	B_j^i	20	0.0008	*18	0.0019					*16	0.0007	*19	0.0007	21	
I Transfer Coef TE BIRD	8.5.1.2	F_j^i	21	0.0008					14	0.0017	19	0.0015	*17	0.0007	16.67	
Food Energy Factor FW FISH	8.5.6.4	Ycf_j	22	0.0006			*19	0.0009	20	0.0011			*16	0.0007	18.3	
Peat/Wood Switch	7.3.5.2	PT	23	0.0004												
Feed Ing Rate TE MEAT	8.5.5.1	Qf_j	24	0.0004												
Food Energy Factor TE MEAT	8.5.6.4	Ycf_i	25	0.0003												
Mean Lake Depth	5.5.2	Z_l	26	0.0003												
Terrestrial Discharge Fraction	4.5.1	δ	27	0.0003												
Tc Plant/Soil CR	8.5.1.1	Bv^i	28	0.0003												
C Plant/Soil CR	8.5.1.1	Bv^i	29	0.0002												
C Solid/Liquid Partition Coef	6.5.3	Kd^i	*30	0.0007												
C Transfer Coef TE BIRD	8.5.1.2	F_j^i	*31	0.0002					17	0.0012	16	0.0017	12	0.0009	15	

* Indicates a failure to pass Mallows' test for significance.

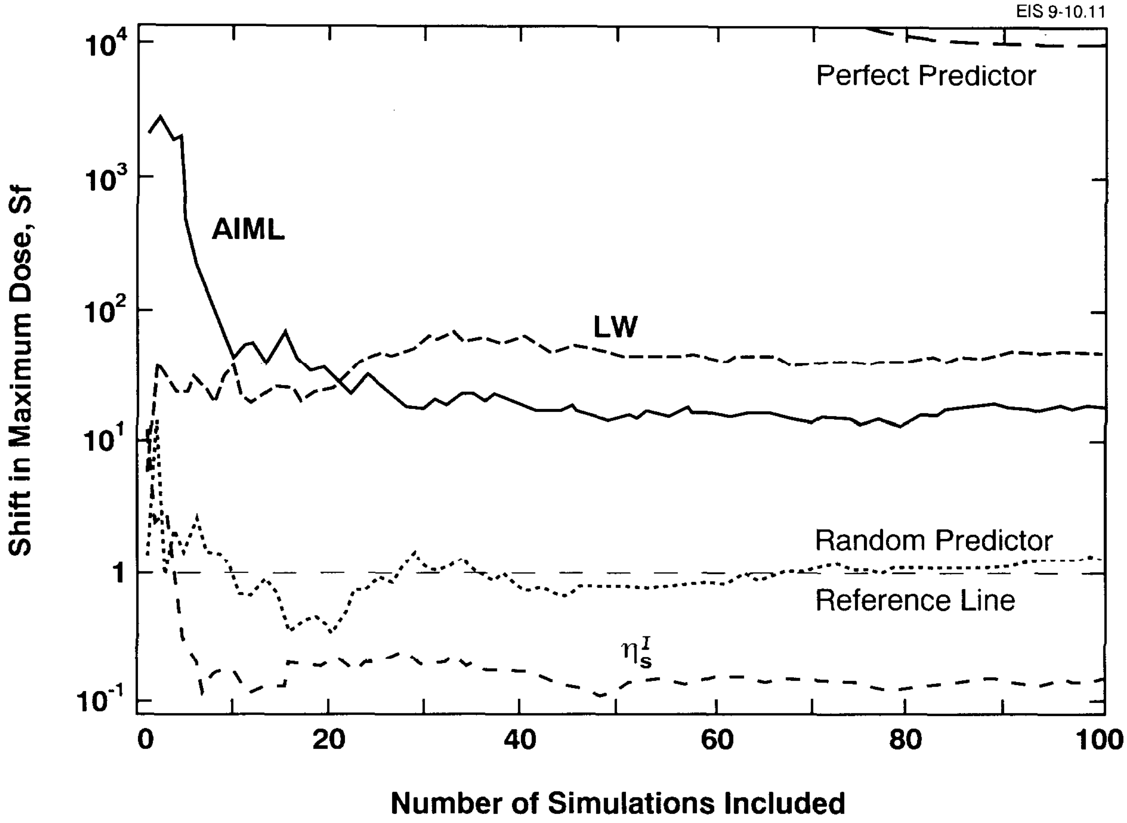


FIGURE 10-11: SENSYV Plot (Sf, shift in mean maximum total dose, versus number of simulations included in calculating Sf) for Lake/Well-Water Switch, LW, Aquatic Iodine Mass Loading Parameter, AIML, and Iodine Gaseous Evasion Rate from Soil, η_s^I. Curves for these most important parameters can be contrasted with the perfect predictor (dose itself), a random predictor (simulation number) and the reference line, indicating no sensitivity. Sensitivity of a parameter increases away from the reference line, and the parameter/dose relationship is positive above and negative below this line.

to the reference line of 1. This rank order reflects that from the logarithmic MREG (Section 10.3.3.1). Figure 10-11 also shows that Sf values are quite variable when calculated from a small number of simulations; about 25 simulations must be tabulated before Sf stabilizes at the average values indicated in Table 10-5. Note that curves for unimportant parameters would deviate little from the reference line and converge towards it with an increasing number of simulations.

The lake/well-water switch, LW, the plant soil concentration ratio, Bv^i, the aquatic iodine mass loading parameter, AIML, and the iodine gaseous evasion rate from soil, η_s^i, are the most important parameters determining the ^{129}I dose to man (Figure 10-6). Since ^{129}I is by far the most important contributor to the total dose to man (Table 10-4), it is not surprising that this list includes the most significant parameters overall, which are LW, AIML and η_s^i (Table 10-5 and Figure 10-11). Note that AIML exerts its influence on dose through air but not water concentrations, i.e., the air/plant/man pathway (Figure 10-6), because water is not depleted when ^{129}I is lost to the atmosphere (Section 5.3.4). As can be seen in Table 10-5, Bv^i for ^{129}I also ranks as an important parameter overall.

Another significant result of the BIOTRAC sensitivity analysis is that half of the six most important parameters in determining the total dose to man are switch parameters (Table 10-5 and Section 2.7.3). They include the lake/well-water switch, LW, the sediment/soil switch, PS, and the irrigation switch for the garden, PI. Even though these switch parameters are very important, values for them can often not be readily assigned (Sections 6.5.5.2, 6.5.5.4 and 9.1.2). The sensitivity analyses for the four submodels (Section 10.2) did not address any of these switch parameters because they are related to the integration of the submodels in BIOTRAC.

The sensitivity analysis results for BIOTRAC as a whole are complementary to those from the separate submodels (Section 10.2). For example, both the aquatic iodine mass loading parameter and the iodine gaseous evasion rate from soil were identified in the atmosphere submodel (Section 10.2.3) as important in determining air concentration. Similarly, the plant/soil concentration ratio and the soil solid/liquid partition coefficient appear as important parameters in the food-chain and dose submodel (Section 10.2.4) and the soil submodel (Section 10.2.2) respectively. On the other hand, not all the parameters identified as important in the submodels are significant in BIOTRAC because they may not be important for ^{129}I. Furthermore, all the parameters were included in some of the submodel analyses, whether they are distributed or not. Thus, catchment area, A_d, was identified as important in determining lake-water concentration in the surface water submodel (Section 10.2.1). This parameter does not appear in the BIOTRAC analysis because it is not distributed (Section 5.5.1).

The sensitivity analysis results for BIOTRAC are also complementary to those from the combined vault, geosphere and biosphere models (Goodwin et al. 1994), particularly the importance of ^{129}I and the pathways linked to well water. However, because our analysis is limited to the biosphere, we could not detect one potentially important effect on dose involving the relationship between the number of persons per household, Nph (Section 9.1.1.1), and the well-water demand (Section 9.1.1.4). These two parameters are positively correlated, but increased water demand also tends to reduce the total dose because more potentially less contaminated surface water is drawn into the well at high demand (Equation (4.18)). This effect is observable only if the geosphere model is included in the analysis. This has been documented for ^{129}I in an earlier study by Reid J.A.K. et al. (1989). Furthermore, with the effects of the vault and geosphere model dampened, Reid's study also identified the lake/well-water switch, the aquatic iodine mass loading parameter and the iodine gaseous evasion rate from soil as the most important biosphere parameters.

10.3.4 Summary and Conclusions

Our sensitivity analysis of BIOTRAC ties in well with the analyses of both the individual submodels and the combined vault, geosphere and biosphere models, and it provides an improved understanding of the way BIOTRAC functions. The results agree with our intuitive expectation of how the model should behave, and this increases our confidence in its predictions.

Our analysis suggests that the most important nuclides at times up to at least 10 000 a are, in decreasing order, ^{129}I, ^{14}C and ^{99}Tc. This is partly related to the relative magnitude of the release of these nuclides from the geosphere. However, the importance of ^{129}I and ^{14}C can also be attributed to the mobility of their gaseous forms. The major ^{129}I pathways are soil/plant/man, air/plant/man, water/man (ingestion), and soil/plant/milk/man. The major ^{14}C pathways are soil/plant/man, fish/man, soil/plant/meat/man, and air/plant/man. The major ^{99}Tc pathway is soil/plant/man. Aerial irrigation using well water constitutes the most important exposure pathway for soil, with ingestion pathways producing the largest doses. Technetium-99 has the simplest flow pattern in the biosphere and it resembles that for ^{14}C; ^{129}I has a more complex pattern.

The most important BIOTRAC parameters are the lake/well-water switch, the aquatic iodine mass loading parameter, and the iodine gaseous evasion rate from soil. In general, switch parameters are among the most important parameters.

Many of the sensitivity analyses results were anticipated from our studies of the interim assessment results and preliminary simulations of the combined vault, geosphere and biosphere models (Goodwin et al. 1994). Thus, while developing BIOTRAC, we paid particular attention to ^{14}C, ^{129}I and ^{99}Tc, and the pathways and parameters that control their consequences.

11. BIOTRAC VALIDATION

11.1 EXPERIMENTAL VALIDATION

Validation is the process of demonstrating that a model adequately represents the system that it is meant to describe. In the case of geological disposal of nuclear fuel wastes, validation means showing that our models represent the processes responsible for the transport of nuclides from the vault to the biosphere, and provide a realistic estimate of environmental and human consequences.

Validation is normally achieved by showing that model predictions agree with field or experimental observations when all the uncertainties are taken into account. Such a procedure is not possible for some of the models developed for the postclosure assessment. The observational data do not exist to validate all aspects of BIOTRAC for an underground contaminant source, even over short periods of time. The data are obviously not available for comparison thousands of years into the future.

The work carried out to validate the individual submodels of BIOTRAC is described in Chapters 5 to 8, and in more detail in the four submodel reports (Amiro 1992b, Bird et al. 1992, Sheppard M.I. 1992, Zach and Sheppard 1992). Both the soil and surface water models have been validated extensively with data from studies that lasted over several years and dealt with a variety of nuclides under a variety of conditions in natural environments on the Canadian Shield (Sections 5.6 and 6.6). These studies have shown that the predictions of the soil and surface water models are consistent with actual observations, or overestimate them. Similarly, the dispersion relationships used in the atmosphere model were derived from a model that agrees well with experimental data (Section 7.6). The duration of the aquatic and atmospheric studies was long compared with the time required for air and water concentrations to reach steady state. Parts of the atmosphere and surface water models can therefore be assumed to perform well over long periods of time. On the other hand, soil concentrations were still changing at the end of some of the experiments, and validation can be claimed only for times that are short compared with the time needed to reach steady state.

The food-chain and dose model (Section 8.6), and the suspension and deposition processes in the atmosphere model have not been validated. As noted in Section 7.6, suspension and deposition are difficult to study in the field when they occur over large areas. In addition, it was not possible to study the large number of suspension processes considered in the atmosphere model. Experimental validation of the food-chain and dose model is nearly impossible because of extremely low nuclide concentrations in the environment. It is also difficult to obtain test data on nuclide concentrations in human tissues and organs.

The BIOTRAC model as a whole has not been validated experimentally. The difficulties in validating the atmosphere and food-chain submodels apply to BIOTRAC as well. In addition, it would be hard to establish a meaningful groundwater source term for the biosphere for a full-scale field study. Very large amounts of tracer would have to be used to raise environmental concentrations above detectable levels. It would be difficult to design an experiment that would cover all of the exposure pathways treated in BIOTRAC. The end point of the study would have to be environmental concentrations or doses to non-human biota because it would be unacceptable to contaminate humans. Because of time restraints, results could be gathered only for the early stages of nuclide buildup in the sediments and soils, which play a large role in the main exposure pathways. Achievement of steady-state conditions might take hundreds or thousands of years. For these reasons, suitable natural analogs might be more useful for model validation than experimental tracers (Section 11.4).

We have not validated BIOTRAC in the formal sense. Instead, we used a variety of other approaches to establish its credibility. We evaluated the model and data in the light of the body of knowledge available on nuclide behaviour in the biosphere, and ensured that it conforms to accepted scientific practices for establishing assessment models. We exposed the model and the experimental work on which it is based to peer review, and modified it according to comments received. We evaluated the model against the behaviour of systems that provide analogs for nuclide transport in the biosphere. We compared our results with those of other models designed to treat similar situations. Where information on a particular process or parameter is lacking, we adopted conservative assumptions and approximations. We followed an informal quality assurance program based on the use of qualified personnel and well-established scientific methodologies. We ensured that the model is working as expected by verifying its predictions through hand calculations, comparing them with those from independent codes and evaluating them with results from analytical solutions. Finally, we show that the systems variability analysis approach (Section 1.5.2) enhances confidence in the model by ensuring that all reasonable conditions have been taken into account, and by placing appropriate limits on the range over which consequences are likely to occur. These approaches to model validation are discussed in turn below.

11.2 MODEL EVALUATION

In this section, we discuss some qualitative aspects of model validation. We show that BIOTRAC reflects the accumulated body of information on nuclide transport through the biosphere, and that it follows current scientific practice for models designed to assess geological disposal systems.

It was noted in Section 2.2 that the various disciplines on which biosphere modelling is based all have long histories of study. Whicker (1983) estimated that the literature contains about 10 000 publications on nuclide transport in terrestrial pathways alone. The physical, chemical and biological processes that occur within each environmental compartment and the contaminant transport through them are reasonably well understood. Concentrations in the various compartments can be calculated with relatively little uncertainty when the source term and the transport pathways are well defined and when the time scale is short. Over the last ten years, this understanding has been applied in many countries to develop models for assessing geological disposal systems for nuclear wastes. This effort has been aided by experience with codes designed to assess conventional nuclear power installations, by intensive research programs set up to address the unique challenges posed by geological disposal, and by international co-operation.

We have taken advantage of all these resources in creating BIOTRAC. The development of each submodel began with a thorough literature review to identify the latest and best information on the relevant processes, modelling approaches and parameter values (Amiro 1992b, Bird et al. 1992, Sheppard M.I. 1992, Zach and Sheppard 1992). These were combined with findings from our own research program (Zach et al. 1987) to provide an appropriate model for assessing the performance of an underground disposal facility. We have accessed international experience in biosphere modelling through periodic meetings with groups from Finland, France, Germany, Japan, Spain, Sweden, Switzerland, the United Kingdom, the United States, and through the model intercomparison program BIOMOVS (BIOsphere MOdel Validation Study) (Section 11.5).

Essentially all the countries with a nuclear power program have developed or acquired a biosphere model to assess the performance of geological disposal systems (Bergström et al. 1982, Korhonen and Savolainen 1982, Lawson and Smith 1984, NAGRA 1985, Malbrain and Lester 1987). These models are all very similar, although not all of them are probabilistic (Barry et al.

1993). Surface waters and soils are generally assumed to be well-mixed compartments, and time-dependent nuclide concentrations are calculated using mass balance equations. Transfers into and out of the compartments are described by first-order rate constants. The atmosphere and food-chain models are generally expressed as steady-state multiplicative chains. Concentrations in air, plants, animals and humans are assumed to adjust instantaneously to changes in concentration in the donor soil and surface water compartments. For the most part, the models all treat the same compartments, the same processes and the same exposure pathways. All have very simple mathematical formulations. This reflects the general belief (Desmet 1988) that, in assessing geological disposal systems, the simplest model that is adequate for the purpose should be used. For the most part, complex models do not provide greater predictive accuracy because their extensive data requirements cannot be met and because of uncertainties associated with the long time frame of the assessment.

The general similarity among models designed to assess geological disposal systems reflects a common understanding of the important processes and of the most appropriate ways to simulate them. Because BIOTRAC shares many of the characteristics of these models, it can claim the credibility that attaches to international consensus. A consensus has been reached that safety assessment methods are available to evaluate the potential long-term radiological impacts on humans and the environment from a carefully designed radioactive waste disposal facility (OECD 1991).

Peer review has helped to ensure that BIOTRAC meets accepted scientific standards. The model and the research program on which it is based have been continually exposed to critical review. They have been discussed at many meetings, both internally with AECL personnel, and externally with other national and international experts. Where possible, we have published our research results in the open literature. We solicited formal, independent reviews of the soil and atmosphere models from experts in the appropriate fields (Elrick 1988, IIASA 1988). Finally, since its inception, our program has been reviewed by the Bioscience Subcommittee of TAC, an independent group of distinguished scientists nominated by Canadian professional societies. Members of TAC have had complete access to all aspects of the biosphere program, and have provided direction and critical review on an ongoing basis (e.g., TAC 1992). Peer review in all forms was aided by the publication of two interim assessments (Wuschke et al. 1981; Wuschke et al. 1985a, 1985b), which described the evolving model and its predictions at intermediate points in the program. The interim assessments provided the opportunity for a thorough evaluation of the model at different stages in its development (Section 10.3.4).

AECL's public consultation program (Greber et al. 1994) has influenced work in several areas. We have also evaluated our modelling approach and model in the light of the scoping hearings for establishing the EIS guidelines (Federal Environmental Assessment Review Panel 1992), and in the light of the guidelines themselves. This has led to many improvements, particularly in the area of assessing environmental protection, focusing on non-human biota.

Peer review has resulted not only in the endorsement of BIOTRAC, but in its improvement. Whenever comments were received, they were carefully considered and incorporated into the model where appropriate. The effects of this process of review and revision are most clearly seen by examining the way the model evolved through the two interim assessments. Transport processes were treated very simply in the first interim assessment (Wuschke et al. 1981), and only the most basic exposure pathways were considered. Review and further research led to a much improved model for the second interim assessment (Mehta 1985; Wuschke et al. 1985a, 1985b). Transport processes were treated more realistically and in greater detail, e.g., a sediment compartment was added to the surface water model, and suspension processes were included to allow the calculation of air concentrations (Sections 5.2). Moreover, a number of additional exposure pathways were treated, including a well as a direct link to the geosphere (Section 4.4.4). The present version of BIOTRAC for the postclosure assessment reflects the improvements made following four additional years of review and research. The compartment model of the soil was replaced by regression equations based on the detailed, mechanistic SCEMR1 model (Section 6.2). Several additional suspension processes were modelled to allow a more complete treatment of the atmospheric pathways (Section 7.2). Irrigation was added as a possible mechanism of soil and plant contamination (Section 6.3.7.2), and soil ingestion by man and animals was included (Section 8.2). Thus, the ongoing review process made it possible to identify deficiencies in the model at an early stage, and to correct them in subsequent versions. There could undoubtedly be further improvements, which would likely reduce conservatism and consequence estimates.

11.3 DATA EVALUATION

A model and its database are closely related. For results to be meaningful, parameter values input to the model must be consistent with model objectives and with the processes being simulated. They should be derived from field and experimental observations made under appropriate conditions. The data should also be of high quality, with minimal random or systematic measurement errors (Section 1.5.7). Moreover, the database should be complete enough to provide good estimates of central values, variability in space and time, and uncertainty.

Values and PDFs for each BIOTRAC parameter were set only after a careful examination and synthesis of the available observations. Generally accepted data and recommended values and distributions exist for many parameters (e.g., Ng et al. 1977, 1982a, 1982b; Hoffman and Baes 1979; Rupp et al. 1980; Minns 1984). These data were screened to identify values that were of high quality, and were relevant to the Canadian Shield biosphere and to the assessment needs. These were augmented by data generated through our own and international waste management research programs. The parameter values and PDFs derived from this information have undergone the same peer review process as the model itself. This has ensured that the data used in BIOTRAC are up-to-date and relevant, and reflect current usage in waste management assessments.

The data required to define values and PDFs for some BIOTRAC parameters are incomplete or missing. Several different approaches were adopted to deal with these situations (Section 2.7). Values could sometimes be set using theoretical arguments. Lognormal distributions were assigned to some parameters since many environmental data are positively skewed and distributed approximately lognormally (Eberhardt 1976, Shaeffer and Hoffman 1979, Ott 1990). It was possible to limit the range of some parameter values through physical arguments. Where data were missing for one nuclide, they could often be deduced from the information for another with similar chemical properties (Baes et al. 1984). Expert opinion, based on our own expertise and that of others, also played a role in establishing some parameter values. Conservative values and PDFs were selected to reflect the large uncertainty, rather than known variation. Reasonable parameter values and distributions could be determined by these means even in the absence of complete data.

11.4 NATURAL ANALOGS

A natural analog is an occurrence, uncontrolled by humans after it has occurred, in which materials, processes or conditions are either identical or similar to those occurring in a man-made situation (Cramer 1993). Analogs may be available for an entire vault, geosphere and biosphere system or for part of it. Analogs are not limited to natural phenomena, but can also result from human activities. Analog information is obtained by reconstructing the history of the occurrence, and so may not be wholly quantitative because of uncertainties. The study of analogs, however, can provide confidence that the behaviour of a system or process is well understood and appropriately modelled.

Cramer (1993) discussed the use of natural analogs in support of the assessment for disposal of Canada's nuclear fuel waste. From a biosphere perspective, analogs consist primarily of nuclides in the environment whose chemical, transport and toxic behaviour parallels that of nuclides in nuclear fuel waste. The most useful analogs comprise long-established systems that provide information on nuclide behaviour over periods of time for which experimental validation of the predictive models is not possible.

Many of the nuclides in nuclear fuel waste (Table 1-1) and other nuclides of similar chemistry occur naturally in the environment, or have been introduced through activities such as nuclear weapons testing and uranium mining (Amiro 1992a). These nuclides are subject to the same transport processes and food-chain transfer as nuclides from a disposal facility might experience. The biosphere in its present state is therefore an analog of the biosphere contaminated by nuclides from the vault. Many of the specific transport processes that vault-derived nuclides would undergo are presently active with analog nuclides. In many cases, studies of these analogs have provided parameter values for use in BIOTRAC. The data for our model of radon emissions from soil are based on work done on uranium mine tailings (Rogers et al. 1980). Our values for the plant/soil concentration ratio for uranium were derived in part from studies in areas of high natural soil uranium concentrations (Sheppard M.I. and Thibault 1983). Transfer rates of particles and gaseous iodine from surface waters to the atmosphere were based on measurements using analog nuclides in the marine environment (Amiro 1992b). A study of moose and other native mammals helped to establish transfer coefficient values for natural food chains (Zach et al. 1989). Information on analog nuclides was used extensively in the human metabolic transfer models on which our internal DCFs for humans are based (Johnson J.R. and Dunford 1983).

We also used biosphere analogs to support the validity of our approaches to modelling nuclide transfer over long periods of time. Profiles of ^{210}Pb that accumulated in sediments over considerable periods of time have been used to develop our model of nuclide transfer from the water column to sediments in the lake (Bird et al. 1992). The development of soil profiles, with unique and diagnostic horizons or layers, provides evidence of the processes responsible for nuclide movement in soils over very long periods of time (Sheppard M.I. 1992) The record of human habitation in Europe and Asia, which extends over several thousand years, provides an analog for the continuous cultivation of soil by the critical group at the discharge zone. Finally, the different biospheres found across Canada today provide analogs for the changes, including continental glaciation, that can be expected to occur over time at a given site on the Canadian Shield.

11.5 MODEL INTERCOMPARISONS

Model intercomparisons are valuable when a number of models have been independently developed to treat the same situation. Similarity in model structure and agreement among model predictions suggest that a common understanding exists concerning the processes of importance in the system and the way in which they should be modelled. This increases the confidence that can be placed in the models, and suggests that the calculations have been performed accurately.

The main vehicle of model intercomparison for BIOTRAC has been the BIOMOVS program (Haegg and Johanson 1988). BIOMOVS is an ongoing international co-operative study to test models designed to calculate the environmental transfer and bioaccumulation of radionuclides and other trace substances. It was initiated by the Swedish National Institute of Radiation Protection in 1985, and includes members from Canada, Japan, the United States and twelve European countries. In the first phase of BIOMOVS (the second phase started in 1991), several test scenarios were defined involving nuclide transport through some part of the biosphere. Each participant modelled the scenarios independently and submitted results to the project secretariat for compilation and analysis. Differences between the predictions of the various models were identified and evaluated in terms of the differences in input data, model structure and assumptions. Participants were also asked to estimate the uncertainty associated with the predictions of their models.

The scenario definitions were deliberately left vague. They were largely descriptive, and contained little quantitative information. Almost all the parameter values were left undefined, and therefore had to be selected by each participant. This was done to encourage discussion of the processes, compartments and pathways that needed to be modelled, and of the parameter values relevant in each case. This vagueness often resulted in different interpretations of the scenarios by different participants and the choice of different parameter values, which led in turn to predictions that frequently showed considerable variation. Therefore, BIOMOVS cannot claim to be a model intercomparison study in the usual sense, and few conclusions were drawn from the quantitative results. Rather, BIOMOVS emphasized the processes of model and data evaluation, and conclusions were based on discussions of model structure, assumptions, processes and parameter values.

Calculations based on our models were submitted for four BIOMOVS scenarios.

1. Irrigation with contaminated groundwater (Scenario B2, Grogan 1989). In this scenario, an agricultural field was assumed to be irrigated with water contaminated with ^{237}Np and ^{99}Tc. Nuclide concentrations in soil, plant crops, meat, milk and air were calculated at various times up to 10 000 a.

2. Discharge to a terrestrial zone—generic case (Scenario B6a, Jones 1990). In this scenario, groundwater contaminated with ^{129}I and ^{237}Np was assumed to discharge below the soil root zone in a farming area. Details of the surface and subsurface water balances, and the characteristics of the soil profile, were not specified. Nuclide concentrations in soil, plant crops and air were calculated at various times until steady-state values were reached.

3. Discharge to a terrestrial zone—site-specific cases (Scenario B6b, Jones 1990). This scenario was a site-specific version of B6a. The models were applied to two specific sites for which water balances and soil characteristics were specified to try to reduce the large variability evident in the B6a results.

4. Discharge to a river (Scenario B7, Zeevaert 1990). In this scenario, groundwater contaminated with ^{137}Cs, ^{237}Np, ^{239}Pu and ^{90}Sr was assumed to discharge through sediments into a river. Nuclide concentrations in river water, sediments and fish were calculated at various distances downstream of the source region at various times until steady-state conditions were attained.

Each of these scenarios deals with pathways that are central in the movement of nuclides from an underground source through the biosphere. Each considers a long time frame and a situation for which data are unavailable for true model validation. Each treats nuclides of importance in the waste management context. Taken together, the four scenarios have tested the key elements of all the submodels of BIOTRAC. They have dealt with the geosphere/biosphere interface in both aquatic and terrestrial settings, with soil contamination from both subsurface and above ground sources, and with transport through surface waters, the atmosphere and the food chain.

To assess the four scenarios, we used BIOTRAC and its submodels as documented in Chapters 4 to 9. The exception is Scenario B7. Since the surface water compartment of BIOTRAC represents a lake, it had to be changed slightly to allow it to handle a river. However, we were able to maintain much of the original model structure in the modified version, including the mass balance formulation and the use of rate constants to describe sedimentation. Results from the B7 scenario can therefore be used to evaluate the performance of the lake compartment of BIOTRAC.

The parameter values that we used in our calculations came from a number of sources. Values that were specified in the scenario description were used as given. Where parameters were undefined, we adopted values that we believed were appropriate for the scenario. These were usually the values that have been discussed in Chapters 4 to 9, but occasionally other values were thought to be more appropriate. Parameters whose values were uncertain were distributed, with PDFs based on the distributions established for the postclosure assessment. The predictions submitted for the intercomparison used the median values of the distributions. The models were also run probabilistically to provide estimates of the uncertainties in the predictions.

Some of the median results from various participants (Table 11-1) are shown in Figures 11-1 to 11-4 for the four scenarios in which we participated. All the environmental compartments and all the nuclides considered in the scenarios are represented in these examples. They are typical of the results obtained for other combinations of compartments and nuclides. Not unexpectedly, given the nature of the scenario definitions, the results are characterized by high variability. For a given scenario, compartment and nuclide, the steady-state predictions of the various models typically span between two and five orders of magnitude. This is due in part to the different parameter values adopted by the various modellers. For example, the participants in Scenario B6a (Figure 11-2) made different assumptions concerning the water balance parameters. Almost all the variability in soil concentrations for this scenario can be explained by the different values adopted for the amount by which precipitation exceeded evapotranspiration (Jones 1990). Another source of variability relates to scenario interpretation. For example, results for both fixed and fluctuating water tables were submitted for Scenario B6b because the scenario description allowed for both interpretations. This increased the associated variability. Finally, it was not always clear from the scenario description what processes were operating. For example, in Scenario B7, each of the four participants assumed a different process (advection, diffusion, sedimentation and burial) was responsible for the transfer of nuclides from compacted sediments to the underlying layers. This difference is again reflected in the variability of the results.

Under these circumstances, it is difficult to draw quantitative conclusions regarding BIOTRAC performance, but some qualitative statements can be made. BIOTRAC predictions do not stand out in any way from the overall body of results. The concentrations that we predict in the various compartments show the same trend with time as those calculated by the other participants. On a relative basis, our results are never extremely high nor extremely low, but usually lie within the range of values predicted by the other models. However, our steady-state concentrations tend to be on the high side when compared with those of others. This reflects the conservative bias (Section 1.5.6) built into BIOTRAC, and is a desirable feature in a model designed for long-term assessments.

Examples of the uncertainties associated with the model predictions are shown in Figures 11-5 and 11-6. Figure 11-5 shows the uncertainties in the estimates of steady-state ^{99}Tc concentrations in soil for Scenario B2, and Figure 11-6 gives the uncertainties associated with steady-state ^{129}I concentrations in plant crops for Scenario B6a. A notable feature of these

TABLE 11-1

LIST OF BIOMOVS PARTICIPANTS FOR THE FOUR SCENARIOS IN WHICH AECL PARTICIPATED

BIOPATH (EIR/NAGRA)	Swiss Federal Institute for Reactor Research, Nationale Genossenschaft für die Lagerung Radioaktiver Abfälle, Switzerland
BIOPATH (Studsvik)	Studsvik Energiteknik, Nyköping, Sweden
BIOS	National Radiation Protection Board (NRPB), United Kingdom
BIOSPHERE	Studiecentrum voor Kernenergie (SCK/CEN), Belgium, Health Physics Department
BIOTRAC	Atomic Energy of Canada Limited (AECL), Canada
CEGB Nuclear	Central Electricity Generating Board, Berkeley Laboratories, United Kingdom
DECOS	Associated Nuclear Services (ANS), United Kingdom
DETRA	Technical Research Centre of Finland
ECOS	Associate Nuclear Services, United Kingdom
ECOSYS	Institut für Strahlenschutz, Gesellschaft für Strahlen und Umweltforschung (GSF), Germany
IAEA	International Atomic Energy Agency (IAEA Safety Series 57), Austria
NCRP	National Council on Radiation Protection and Measurements, United States
NRIRR	National Research Institute for Radiobiology and Radiohygiene, Hungary
PSI/NAGRA	Paul Scherrer Institut, Nationale Genossenschaft für die Lagerung Radioaktiver Abfälle, Switzerland
SCK/CEN	Studiecentrum voor Kernenergie (SCK/CEN), Belgium
SIRATEC	National Research Institute for Radiobiology and Radiohygiene (NRIRR), Hungary
TERRA	Oak Ridge National Laboratory (ORNL), United States
TODOS	Oak Ridge National Laboratory (ORNL), United States
TRUMP	KEMAKTA Consulting Company, Sweden
USNRC 1.109	United States Nuclear Regulatory Commission (NRC Regulatory Guide 109), United States

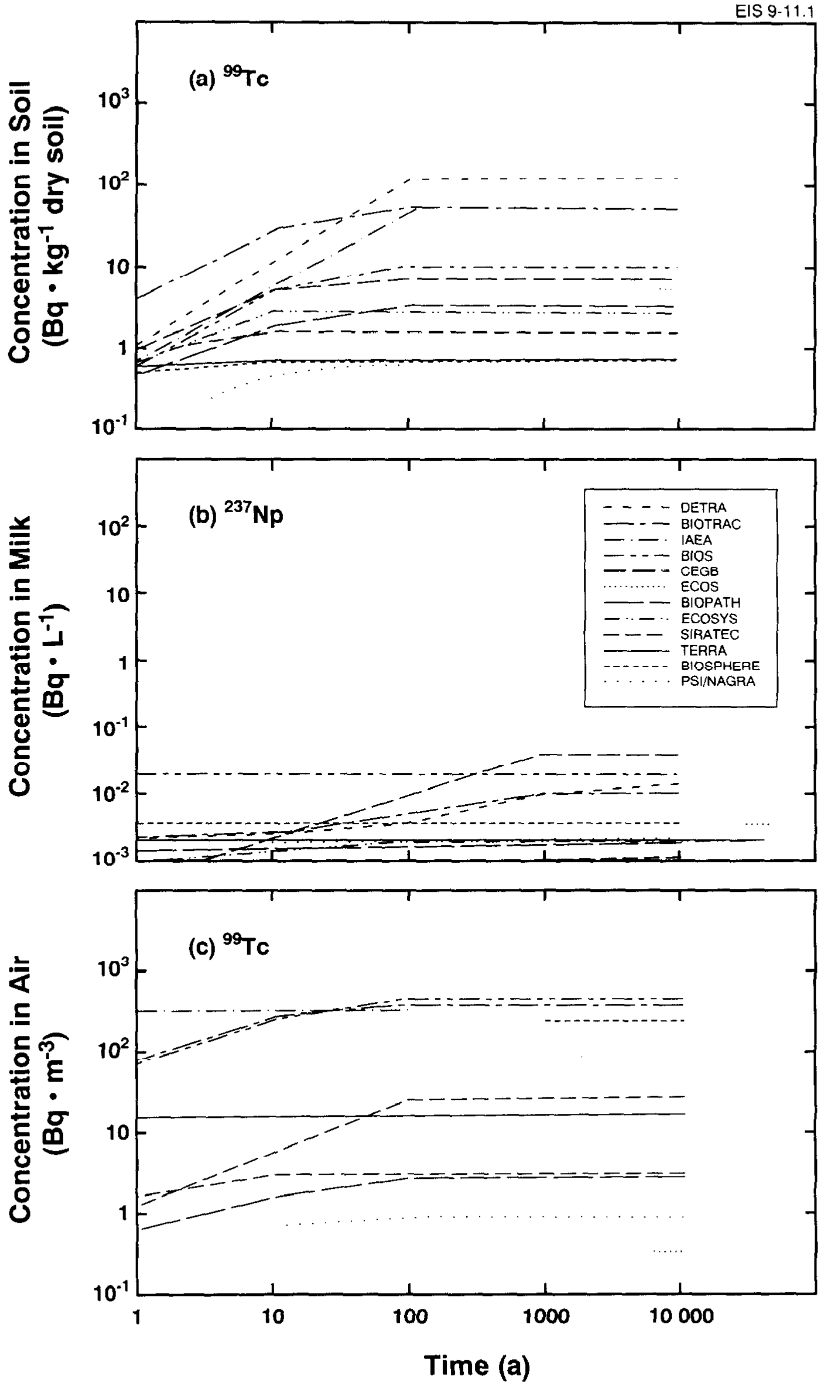


FIGURE 11-1: Results of Scenario B2 (Irrigation with Contaminated Water). (a) ^{99}Tc concentration in soil, (b) ^{237}Np concentration in milk, and (c) ^{99}Tc concentration in air. Table 11-1 lists the full names of the participants.

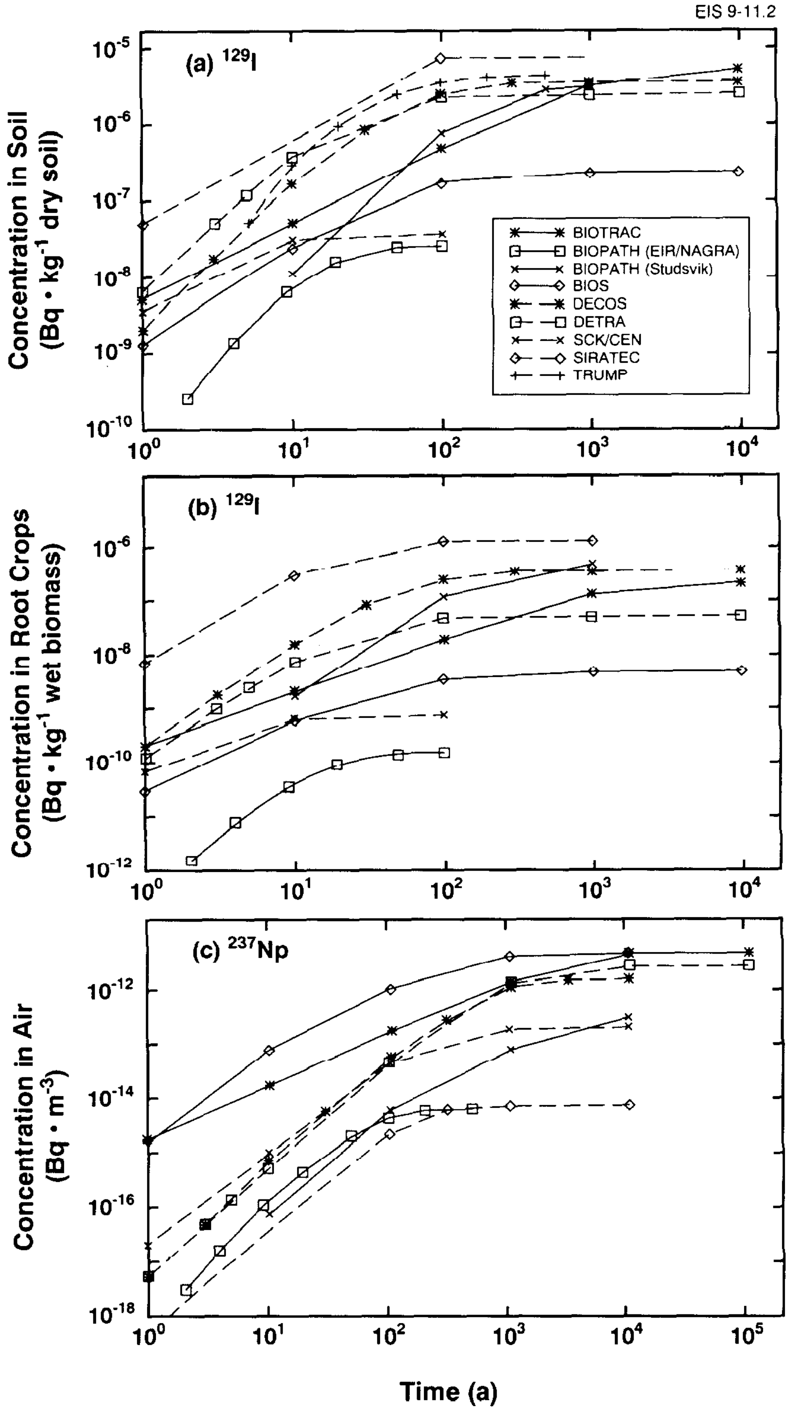


FIGURE 11-2: Results of Scenario B6a (Groundwater Discharge to a Terrestrial Zone—Generic Case). (a) ^{129}I concentration in soil, (b) ^{129}I concentration in root crops, and (c) ^{237}Np concentration in air. Table 11-1 lists the full names of the participants.

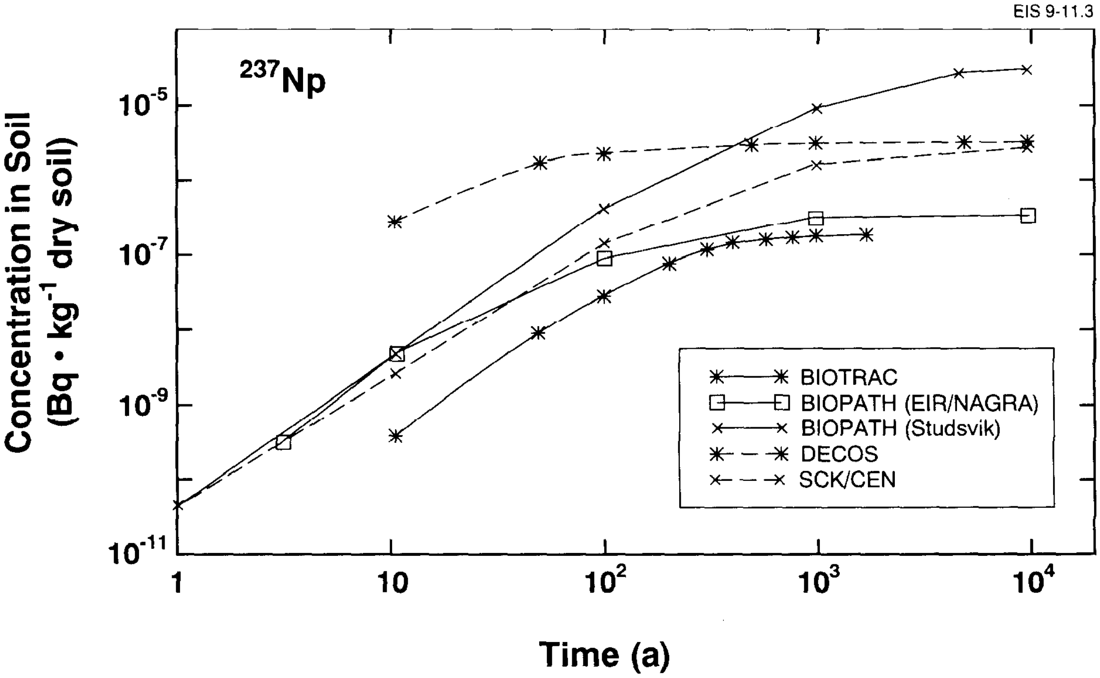


FIGURE 11-3: Results of Scenario B6b (Groundwater Discharge to a Terrestrial Zone—Site-Specific Case). ^{237}Np concentration in soil. Table 11-1 lists the full names of the participants.

results is the large variability in the magnitude of the uncertainties estimated by the various participants. This variability arises because the modellers calculated their uncertainties in a number of different ways. They accounted for different sources of uncertainty, assigned different values to the sources they considered, and used different propagation techniques to obtain the final uncertainty in their predictions. A second point to note is that the uncertainties tend to be large. This is not an unexpected result. Uncertainties should be of the same order of magnitude as the variation in results when different models are applied to a common situation (Section 1.5.7).

A number of sources contributed to the uncertainty in the BIOMOVS results. The vague definitions of the scenarios often made interpretation difficult, leading to differences in the conceptual models used and in their mathematical formulation. Modellers had to supply most of the parameter values needed for the calculations, and differences arose in choosing appropriate values for the scenarios. Finally, many participants tried to account for uncertainties associated with environmental and cultural change over the long time frame of the scenarios. Although our understanding of the relevant processes and pathways may be good for present conditions, the uncertainties increase as calculations are extended further and further into the future.

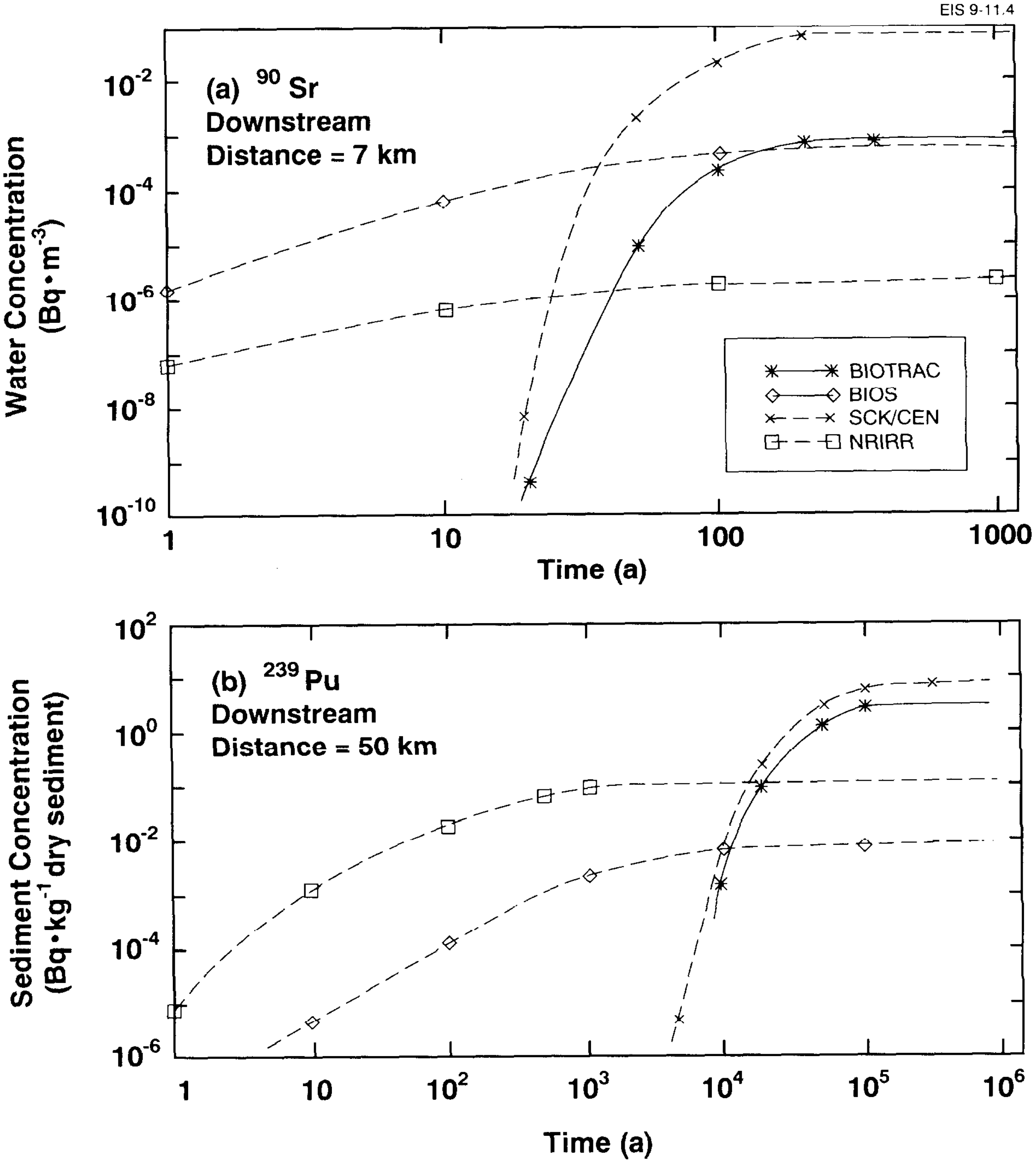


FIGURE 11-4: Results of Scenario B7 (Groundwater Discharge to a River). (a) ^{90}Sr concentration in river water 7 km downstream from the source, and (b) ^{239}Pu concentration in sediment 50 km downstream from the source. Table 11-1 lists the full names of the participants.

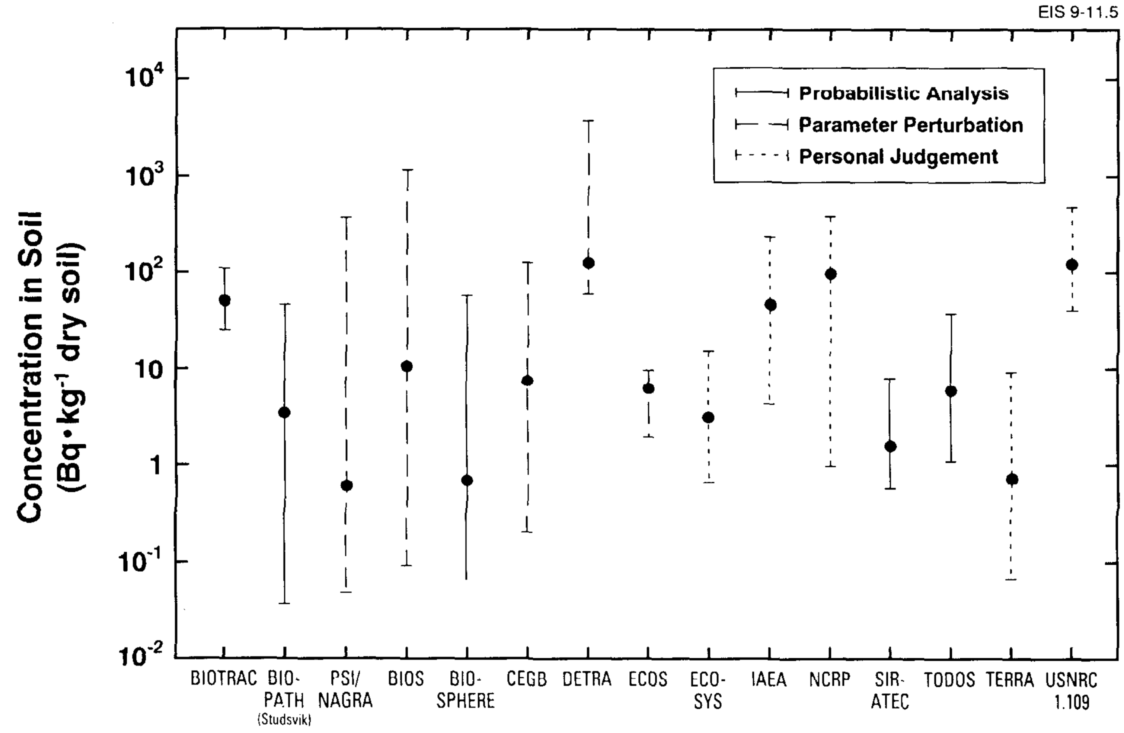


FIGURE 11-5: Uncertainty Estimates for ^{99}Tc Concentration in Soil at Steady State for Scenario B2. Table 11-1 lists the full names of the participants.

The results in Figures 11-5 and 11-6 show that, in general, the error bars estimated by the various participants overlap. This indicates that the predictions of the various models are similar when uncertainties are taken into account. Moreover, with few exceptions, all of the participants considered the same processes and treated them mathematically in a similar way using compartment models. This reflects the origins of these models in nuclear fuel waste management programs (Section 11.2). The predictions were discussed intensively at several meetings of the participants, who were able to reconcile most differences as being the result of different interpretations of the scenarios or different choices for parameter values. Therefore, despite the large variation in results, and the large uncertainties in the predictions, the BIOMOVS study has shown qualitatively that there is a common understanding of the processes and pathways of importance in nuclide transport through the biosphere. BIOTRAC is included in this conclusion. BIOMOVS has shown that our model is consistent with those of others worldwide, and that our predictions are comparable with theirs. The uncertainties are likely smaller when BIOTRAC is used to assess the disposal concept since that situation will be well-defined and the parameter values can be chosen to reflect the site being assessed.

One further aspect of model intercomparison has involved our specific-activity model for tritium (Section 2.5.1). Predictions of our model were

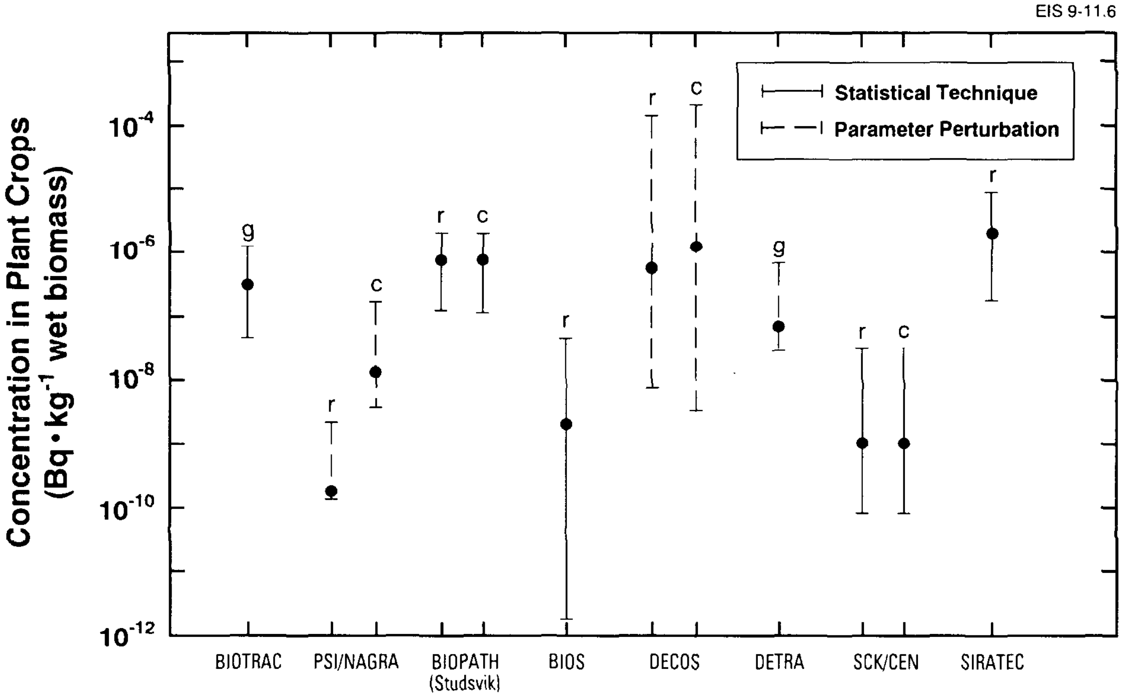


FIGURE 11-6: Uncertainty Estimates for ^{129}I Concentration in Crops at Steady State for Scenario B6a. Root crops = r, cereals = c, and generic crops = g. Table 11-1 lists the full names of the participants.

compared with those of UNSCEAR (1982) and NCRP (1979), and with the pre-closure model PREAC (Russell 1993) for a number of chronic exposure scenarios. The agreement was very good in all cases (Zach and Sheppard 1992), lending confidence to this part of BIOTRAC.

11.6 CONSERVATISM

BIOTRAC was developed to model as realistically as possible the transport of nuclides released from an underground nuclear fuel waste vault through the biosphere and the consequences to the environment and to humans. However, our knowledge of the relevant processes and parameter values is not perfect, and the uncertainties compound when predictions are made for times far into the future. We have made assumptions to ensure that consequences are overestimated or conservative in areas where our knowledge is limited for now, and where realistic models cannot be formulated or validated (Section 1.5.6).

Conservatism has been used prudently in many aspects of BIOTRAC. Doses are calculated for members of a critical group who are expected to receive the greatest exposure received by any humans because of their location and lifestyle. The critical group is assumed to have access to, and draw all of their resources from, only those parts of the biosphere that are most highly contaminated. Their assumed behaviour with regard to water sources, irrigation, choice of building materials and use of lake sediments is conservative. The use of the critical group concept compensates to some extent for the uncertainties associated with the long time frame of the assessment. As indicated in Section 1.5.4, the arguments presented here are not only relevant for human protection, but also for environmental protection involving other organisms (Chapter 13).

We believe that the four submodels of BIOTRAC are individually fairly realistic. The most conservative is probably the atmosphere submodel (Chapter 7). The processes by which material is suspended into the atmosphere are complex and not fully understood, and have been simulated using simple but conservative models. When the submodels are combined, an additional measure of conservatism is introduced because extra nuclide mass is added to the system (Section 9.3). Inventories of donor compartments are not always reduced when nuclides migrate to another compartment. This leads to an effective generation of mass, and an overestimate of the consequences. Finally, when it is uncertain whether or not a process with potential to produce a consequence will occur, we assume conservatively that it does. For example, we assume that nuclides always discharge from the geosphere to a terrestrial zone (Section 4.4.3), although this process may be uncommon in nature.

We have seen in Chapters 4 to 8 that conservative values and PDFs have been adopted for many parameters. For example, the values chosen for soil depth (Section 6.5.1.2) and man's total energy need (Section 8.5.6.1) lead to overestimates of concentrations and doses. Furthermore, parameters such as the plant/soil concentration ratio (Section 8.5.1.1) and the aquatic concentration ratio for fish (Section 8.5.1.3) are assumed to be lognormally distributed with very large GSDs. Consequences calculated using values from the upper ends of these distributions will be disproportionately large, and may result in very conservative concentration and dose estimates.

Overall, we believe that the concentrations and doses predicted by BIOTRAC will overestimate the true values. The magnitude of the overestimate is difficult to quantify, but we believe it is not excessive. Considerable effort has been spent on studying the processes and parameters that contribute most to dose, and these have been modelled as realistically as possible. Conservatism appears more frequently in the less important pathways, which have been less intensively studied and are accordingly less well understood. Conservatism has been used as a tool to compensate for uncertainties where they exist in our understanding, and has resulted in a model that will not underestimate consequences.

Conservatism is expressed in BIOTRAC in many separate instances, and this has a cumulative effect. Furthermore, the consequences estimated by BIOTRAC also include any conservatism in the vault and geosphere models. The consequences estimated by the combined SYVAC3 system model are likely substantial overestimates.

11.7 QUALITY ASSURANCE PROCEDURES

11.7.1 Quality Assurance of the Model

From the outset of the NFWMP in 1978, the work on which BIOTRAC is based has been subject to an informal quality assurance (QA) program guided by well-established scientific methodologies. A judicious mix of field surveys and experimental studies was undertaken to address specific aspects related to nuclide movement through the biosphere (Iverson et al. 1982, Zach 1985a, Zach et al. 1987). These studies were carefully designed, and used appropriate equipment and sampling procedures to gather the relevant data. Both sample analysis and data analysis were generally done using standard, reproducible procedures, and were well-documented. Most of the work has been published in the open scientific literature, which also provided a source for much data and information, and guidance as to what studies needed to be done. BIOTRAC and its parameter values have evolved directly from the understanding and data gained through our own studies and from the literature.

AECL policies and procedures have ensured that all the work has been done by fully trained personnel, well qualified for the tasks. Furthermore, all of the publications have been subjected to formal internal review and approval processes. The EIS and all the primary references were written to conform to specific QA procedures developed for these documents.

A formal QA program to guide waste management activities was initiated in 1990 (AECL 1990). This was done in response to stimuli both within and outside AECL (AECB 1987, TAC 1987) as the value of QA programs to complex projects became more generally recognized. This program has not been fully applied thus far but it will be in the future. The program provides a framework for the planned and disciplined consideration of all the elements that influence the quality of waste management activities and products. It defines responsibilities and sets out procedures for applying QA principles to planning, executing and documenting all the work activities, including those associated with data gathering and interpretation, and with model development and application. It ensures that requirements for waste management activities and products are satisfied in a systematic way, and that the results and documentation are traceable to the source information filed in a records management office.

11.7.2 Code Verification

An important element of QA is code verification, the process of demonstrating that a computer code is a proper representation of a conceptual model, and that the mathematical equations expressing the model are correctly encoded and solved.

A highly structured approach to code development was adopted to ensure that the BIOTRAC code in SYVAC3 performed as intended. This approach involved a number of stages: task specification, which included the preparation and formal review of data flow diagrams and a data dictionary describing the BIOTRAC model equations (Page-Jones 1980); code design, in which the sequence of calculations to be made was subdivided into modules, and the features and interactions of each module were described; and coding, which involved the preparation and preliminary testing of the computer code for all the modules. In addition, standard forms were used to pass parameter values from the researchers to the coders to ensure that the correct data were input into the BIOTRAC code (Stephens et al. 1989). A fuller account of these procedures is given by Goodwin et al. (1994).

The final step in code verification is testing. We have subjected the BIOTRAC code to extensive testing at every stage of its development. Each submodel was tested individually when it was first installed in SYVAC3. The numerical output for a number of input data sets was compared against known solutions obtained using either hand calculations or an independently developed computer code. The test simulations for the surface water and soil submodels included cases with constant source terms for which analytical solutions to the convolution integrals are available for checking (Sections 5.3.3 and 6.3.6). The data sets were chosen to test model performance for all the important nuclides and exposure pathways. Wherever the output of the code differed from the known solutions, the cause of the difference was identified and corrected.

Similar procedures were employed to test the full BIOTRAC code to ensure that the interfaces between the submodels and between the biosphere and the geosphere models had been properly implemented. In addition, time series of environmental concentrations and doses were plotted for all exposure pathways for the most important nuclides. These and similar results from the sensitivity analysis (Chapter 10) were examined qualitatively to ensure that they met intuitive expectations based on our understanding of the overall system.

The entire biosphere model is mathematically quite simple. The atmosphere, and food-chain and dose submodels (Chapters 7 and 8) are expressed as multiplicative chains. The surface water and soil submodels (Chapter 5 and 6), although somewhat more complex, are still relatively straightforward. Accordingly, all components of the model can be readily verified using the methods described above. All the testing was done by personnel who were not involved in producing the code, and who were challenged to find errors in it. This has included spot-checking of the surface water and soil submodel codes by the TAC. On the basis of all these tests, we believe that the code of BIOTRAC for the postclosure assessment is an exact representation of its mathematical formulation, and that it is working as intended.

11.8 SYSTEMS VARIABILITY ANALYSIS APPROACH

The formulation of BIOTRAC in probabilistic terms provides a degree of confidence in its prediction that would not be possible using a deterministic approach. We have assigned lognormal distributions with large GSDs to many of the parameters. The wide range of values considered accounts for uncertainty in the data, for spatial variability across the Canadian Shield in Ontario, and for temporal variability during interglacial conditions (Section 1.5.7). All the reasonable biosphere states, ecological conditions, chemical conditions and human activities are likely represented in the thousands of model simulations with variable combinations of parameter values. This in turn is reflected in our concentrations and dose estimates for the postclosure assessment (Goodwin et al. 1994).

The systems variability analysis approach also allows a quantitative estimate to be made of the uncertainty in the dose predictions. The distribution of results gives an estimate of the range of possible doses, and defines a probability with which a given dose could occur. The arithmetic mean of the doses predicted in each simulation provides a conservative measure of disposal facility performance in the postclosure phase (Section 2.7.4).

The systems variability analysis approach therefore enhances confidence in BIOTRAC by ensuring that all reasonable conditions have been taken into account, and by indicating the range over which consequences are likely to occur.

12. THE IMPACT OF CONTINENTAL GLACIATION ON NUCLIDE TRANSPORT THROUGH THE BIOSPHERE

12.1 THE NATURE OF GLACIATION

Over the past two to three million years, the Canadian Shield has undergone repeated continental glaciation (Hays et al. 1976). The glacial oscillations are hypothesized to be controlled by the flux of solar radiation received at the top of the atmosphere, which varies in response to changes in the earth's orbital parameters (Milankovitch 1941). Conditions favourable to periodic glaciation are expected to persist throughout at least the next one million years and cause between 10 and 30 glacial advances during this period (Shilts 1984). The ice volume record shows a general trend toward increased extent of continental ice with time (Shackleton and Opdyke 1973), so it is likely that a vault located on the Canadian Shield would be covered with ice during future advances. The exact timing of the next glaciation is uncertain, and external forces such as anthropogenic increases in atmospheric CO_2 concentrations may affect the onset. However, it is likely that the next glaciation may occur about 20 000 a from now (Eronen and Olander 1990).

At their largest extent, ice sheets several kilometres thick will cover the ground. Thus, the biosphere will undergo catastrophic changes during each advance and retreat of the ice. Temperatures and precipitation regimes, the volume and pattern of surface water flow, geomorphology, ecology and human cultural practices will all vary profoundly during each glacial cycle. Old Shield faults may be rejuvenated by the crustal stresses imposed by cyclic glacial loading and unloading (Sanford et al. 1985). Soils, vegetation cover and drainage systems will undergo a characteristic pattern of succession following the retreat of the ice (Pielou 1991). Surface water flow, the nature of glacial deposits, and soil and vegetation types in the vicinity of the discharge zone may be quite different after each glacial retreat.

Glaciation is likely the most severe transitional process that will affect the biosphere in the far future in relation to a nuclear fuel waste disposal facility in Canada (Section 3.3). This process has also been of interest in other northern countries such as Finland (Eronen and Orlander

1990), Sweden (Lindbom and Boghammar 1991, Vallander and Eurenivs 1991) and Great Britain (Goodess et al. 1991), and its impact on nuclide transport must be assessed. This is difficult to do, partly because there has been little research regarding the effects of glaciation on our society and the environment in general. We initiated our research by means of a major workshop on transitional processes (Heinrich 1984), developed an assessment methodology (Davis 1986), and solicited an independent analysis by external consultants from McGill University (Elson and Webber 1991). Here we synthesize this information and use BIOTRAC to assess the impact of glaciation on the dose to members of the critical group. We assume that humans are a general indicator species and that our results are also relevant for other biota (Chapter 13).

12.2 DISCRETE STATE ASSESSMENT APPROACH

We investigated a number of different methods for taking the effects of glaciation on the biosphere into account (Davis 1986). We decided on a discrete-state approach, in which the continuous range of possible future biospheres is broken down into a small number of distinct steady-state units (Davis 1986). The effects of glaciation on nuclide transport were evaluated by performing a separate, time-independent assessment of some of these states, assuming that each state persists throughout the entire BIOTRAC simulation period. Glacially induced transient events that cannot be resolved in a particular state have also been identified and evaluated.

The alternative to the discrete-state approach is to try to predict the future state of the biosphere, and nuclide transport through it, in a fully time-dependent model. Computer models have been developed in the United States (Petrie et al. 1981), France (BRGM 1985) and Great Britain (Frizelle 1986) to simulate the long-term evolution of natural disposal environments, taking into account such processes as climatic change, glaciation, erosion and faulting. The validity of this approach has not been established. Moreover, it is not clear that credible results can be obtained from a predictive model given the poor understanding of the causes and timing of glaciations, and of their effects on the biosphere and on nuclide transport through it. The development of a time-dependent model for the assessment of the Canadian nuclear fuel waste disposal concept would be particularly difficult because we are not assessing a specific site.

The discrete-state approach provides a practical and credible framework for the thorough evaluation of the role of glaciation in nuclide transport through the biosphere. Rather than attempting to predict the future, we have used historical data from past glaciations and information from regions of the earth that are currently undergoing glaciation to define characteristic biosphere states. It is easier to define representative parameter values for a few states than to deduce a time-dependent sequence of values extending through several glacial cycles. The discrete-state approach is also conservative. If large consequences are associated with a particular state, the greatest impacts will occur when that state is assumed to persist throughout a simulation, rather than appearing intermittently as one of a sequence of states.

The discrete-state approach has been adopted in other countries to address environmental change (Grogan 1985; Jones 1986a, 1986b; Ashton 1988). Since the next ice advance is not expected within 10 000 a, mathematical models need not be used to evaluate glaciation in our assessment (AECB 1987). Nevertheless, models are helpful. The discrete-state approach provides reasoned arguments from which to evaluate the impact of glaciation on the potential dose to humans.

12.2.1 Definition of Glacial States

Glacial states have been identified for assessing other geological disposal systems using the Astronomical CLimatic INdex (ACLIN) (Kukla 1979, Stottlemyre et al. 1981). In the context of the Canadian NFWMP, the ACLIN system has been discussed by Findlay et al. (1984), Matthews (1984) and Elson and Webber (1991). We define four basic ACLIN states, which can occur in various sequences:

ACLIN 1 - interglacial,
ACLIN 2 - mild interstadial,
ACLIN 3 - cold interstadial, and
ACLIN 4 - full glacial (stadial).

The ACLIN 1 state is typical of present-day conditions and is, therefore, represented by the models and parameter values used for assessing the central group of scenarios (Section 1.5.1).

In addition, there are subsets of these four basic states, some of which are associated with transient events, such as the onset of glaciation. The detailed characteristics of each of these states, relative to our assessment of the Canadian Shield environment, are discussed by Elson and Webber (1991). A brief summary of the major features distinguishing climate states throughout the cycle is given in Table 12-1 for a location near Sioux Lookout, Ontario, which only serves as an example.

TABLE 12-1

CLIMATIC FEATURES OF THE ACLIN STATES FOR AN EXAMPLARY LOCATION NEAR SIOUX LOOKOUT, ONTARIO

Mean Annual Conditions	ACLIN State			
	1	2	3	4
Temperature (°C)	1.5 to -0.5	-1 to -5	-6 to -10	-12 to -15
Precipitation (cm)	57 to 62	42 to 46	26 to 30	
Wind Speed ($m \cdot s^{-1}$)	4	4	6.4	

Note: Data summarized from Elson and Webber (1991).

The most important features of the states are that ACLIN 1 (our present state) is the warmest and wettest, ACLIN 2 and 3 are colder and drier, and ACLIN 4 is a fully glaciated, essentially uninhabitable, environment.

12.2.2 Transient Events

Because glaciation is a continuing process of climate evolution, there are many transient events associated with the glacial cycle. Elson and Webber (1991) describe transient events that occur over relatively short time periods, typically less than 2000 a. Most of these processes are associated with either glacial advance or retreat as the climate changes from one ACLIN state to another. We have been unable to identify any processes that could increase nuclide transport through the biosphere to humans during a glacial advance. Events during a retreat must be more carefully considered. For example, the frozen environment of ACLIN 4 may decrease water flow in the biosphere and in the upper parts of the geosphere, and allow gaseous nuclides such as ^{14}C and ^{129}I to accumulate at the ice/ground interface. These nuclides may be rapidly released to the biosphere as the ice melts. However, the most striking feature of a glacial retreat is the vast amount of meltwater involved, which has resulted in the formation of huge lakes in past glaciations (Pielou 1991). The increased dilution would likely more than compensate for any increase in nuclide release to the biosphere by trapped gases or other mechanisms, and concentrations and doses would remain at or below the levels predicted for other parts of the glacial cycle.

12.3 PREDICTION OF GLACIATION EFFECTS ON NUCLIDE TRANSPORT

Elson and Webber (1991) developed a method to derive meteorological parameter values for different glacial states for an arbitrary location on the Canadian Shield. They listed representative values for the four ACLIN states, identified the pathways by which nuclides could reach humans at each stage of the glacial cycle, and discussed the associated parameter values. We have used this information to evaluate the impact of glaciation on the dose to humans.

12.3.1 Modifications to BIOTRAC to Consider Glaciation

Glaciation is not expected to create any major new transport pathways within a given glacial state. This means that BIOTRAC, in the form described in Chapters 4 to 8, can be used to assess the different glacial states, although some model and parameter value changes may be necessary. Suggested modifications to the biosphere model are outlined by Elson and Webber (1991). It should be pointed out that their recommendations are based on the biosphere model used in the second interim assessment (Mehta 1985), and therefore some of their suggestions are not relevant to BIOTRAC. Also, some of their data for ACLIN 1 do not correspond exactly to those used in BIOTRAC. Specifically, their compilation of climate data for ACLIN 1 are based on a demonstration site near Sioux Lookout, whereas our data for assessing the central group of scenarios represent a broader scope suitable for generic Canadian Shield sites in Ontario, including Sioux Lookout. However, the discrepancies are minor and do not affect the outcome of our glaciation assessment.

The lifestyle of the critical group may change with the glacial state. The present interglacial environment supports agriculture and a self-sufficient farming culture is assumed to thrive. ACLIN 2 should also permit such activity, with summer temperatures close to those of ACLIN 1, but with colder winters and less precipitation (Elson and Webber 1991). However, ACLIN 3 may not support agriculture as it is practised today; temperatures are too low, permafrost may be common, and there is much less precipitation. The critical group living in an ACLIN 3 environment may have to rely more on a hunter/gatherer existence typical of some northern Canadian aboriginal peoples. Alternatively, there may be extensive importation of food from warmer climates (a common practice now), or development of advanced technology for food production in cold climates. Food importation would decrease the dose to the critical group. It is not likely that technological advances in food production will increase nuclide transport to humans appreciably.

In the full glacial state, ACLIN 4, an extensive ice sheet would cover the potential disposal site. Humans and most other biota cannot survive in such an environment, so the nearest inhabited areas would lie far from the disposal facility at the ice margin, or in ice-free corridors and similar refugia (Pielou 1991). These areas would likely correspond to the ACLIN 3 state. Nuclide concentrations and doses would be very low in these areas because of the great distances involved, the dispersive power of the glacier and the reduced rate of water flow in the geosphere and biosphere.

We evaluated the effects of glaciation on nuclide transport by comparing doses for the ACLIN 1 and ACLIN 3 states. ACLIN 2 conditions, and therefore doses, can be expected to be intermediate between those for ACLIN 1 and ACLIN 3, and so need not be calculated. ACLIN 4 is assumed to be uninhabited and was not assessed explicitly.

In principle, predictions for ACLIN 1 are provided by the results for the central group of scenarios of the postclosure assessment (Goodwin et al. 1994), which represent present-day conditions. However, to effect a comparison, we had to change parts of the soil submodel (Section 12.3.2.3). Among other factors, this involved restricting the glaciation assessment to two soil types: organic and sand. Concentrations of nuclides in loam and clay soils tend to be intermediate. Thus, the glaciation assessment consists of the comparisons of the total dose to man, based on organic or sand soils, for the ACLIN 1 and ACLIN 3 states.

There is no evidence that parameter variability changes from state to state. Hence we decided to base the glaciation assessment on median parameter values. BIOTRAC, implemented in SYVAC3, can be readily run with median values. This simplification is consistent with the guidelines that disposal facility performance beyond 10 000 a can be assessed using reasoned arguments (AECB 1987). Therefore, probabilistic comparisons are not needed.

The analysis was simplified further by comparing results for ^{14}C, ^{129}I and ^{99}Tc only. These nuclides were among the greatest contributors to dose in the second interim assessment (Wuschke et al. 1985b) and the present assessment (Section 10.3.2.1, Goodwin et al. 1994). Technetium-99 is handled by BIOTRAC in the same way as the majority of nuclides, whereas both ^{14}C and ^{129}I exhibit special pathways and modelling approaches (Section 2.5). The results from these three nuclides are expected to be representative of all nuclides and to provide a reasonable approximation to the total dose.

For the assessment of ACLIN 3, we stipulate that the climate should correspond to that outlined by Elson and Webber (1991), which is similar to the present climate near Ennadai Lake, NWT (61°N, 100°W). It is difficult to find a true analog of an ACLIN 3 state today because higher latitudes have different daylight conditions than lower-latitude sites would have under continental glaciation.

We model our ACLIN 3 critical group on present-day northern cultures. The cold climate limits agriculture, and it is possible that the critical group will be a hunter/gatherer society, so we account for an increased reliance on wild game and fish. However, marginal agriculture may be possible, and we consider this because it results in conservatively high doses. We make no allowance for food importation or advanced food production technologies.

12.3.2 The Submodels and Parameter Values for the Glacial Assessment

For the ACLIN 1 state, we used the median parameter values given in Chapters 4 to 9 with few exceptions. In the following sections, we define and justify changes to the parameter values required for the glaciation assessment, particularly for the ACLIN 3 state. The parameter values that were adjusted are listed in Table 12-2 for both states. Parameters that were not changed are either insensitive to different climates, or are sufficiently conservative or uncertain that there is no basis for change.

12.3.2.1 Geosphere/Biosphere Interface

For the glaciation assessment, we consider the biosphere in isolation, without incorporating potentially glacially induced changes in the flow of nuclides from the geosphere, a topic addressed by Davison et al. (1994b). The input to BIOTRAC consisted of median time series of nuclide discharges, as calculated by the geosphere model for the postclosure assessment (Goodwin et al. 1994). We assume that the geosphere/biosphere interface model (Chapter 4) applies equally well to the ACLIN 1 and ACLIN 3 states. The only relevant parameter specifically belonging to the interface is the terrestrial fraction of the total discharge zone, δ (Section 4.5.1), which is assumed not to change.

The occurrence of transient events during glacial advance and retreat could affect the geosphere/biosphere interface. A glacial advance could change the hydraulic head conditions, perhaps altering the location of the discharge zone (Elson and Webber 1991). In this instance, we assume that the critical group will occupy the area around the new discharge zone, which is assumed to be identical to the original discharge zone (Figure 4-4) from a modelling point of view. During a glacial retreat, additional meltwater in the vicinity of the interface would likely dilute nuclides reaching the surface. Since this will decrease concentrations and doses, and since the amount of dilution is difficult to quantify, we have not explicitly considered it in the model modifications for ACLIN 3.

12.3.2.2 Surface Water Submodel

The surface water model described in Chapter 5 can be readily used for the glaciation assessment, and most of the parameter values were not changed for the ACLIN 3 state. However, the sedimentation rate, S_{sed} (Section 5.5.5), is generally lower in more northern lakes because of lower production rates of organic matter at lower temperatures (Elson and Webber 1991). Accordingly, we decreased our ACLIN 1 value of 0.16 kg dry sediment$\cdot m^{-2}\cdot a^{-1}$ to 0.044 $kg\cdot m^{-2}\cdot a^{-1}$ on the basis of paleolimnological postglacial data from 17 Canadian Shield lakes (Bird et al. 1992). The gaseous evasion rate for ^{14}C, η_I^C (Section 5.5.9), was decreased to 15% of its ACLIN 1 value, as shown in Table 12-2. This lower rate reflects a shorter ice-free season and slower diffusion rates (Thurber and Broecker 1970). Runoff, R (Section 5.5.4), was also decreased, as discussed in Section 12.3.2.6. The remaining parameter values of the surface water model (Section 5.5) were assumed to be representative of both the ACLIN 1 and ACLIN 3 states.

12.3.2.3 Soil Submodel

The soil model (Chapter 6) does not explicitly consider permafrost, which will likely be present in the ACLIN 3 state (Elson and Webber 1991). Thus, our model needed to be modified. Permafrost would limit the upward movement of nuclides in solution, essentially blocking soil contamination from groundwater discharge but, to be conservative, we have allowed for such discharge at the geosphere/biosphere interface (Section 12.3.2.1).

The existence of permafrost is to a large extent inconsistent with agriculture, but vegetation can thrive over permafrost. To support agriculture, irrigation would have to be practised to compensate for low precipitation in ACLIN 3 (Table 12-1). This may result in soil contamination through irrigation. Therefore, we used SCEMR1 to derive new values for $(Css)^i_{ID}$ and $(tss)^i_{ID}$ (Section 6.3.2) for both the ACLIN 1 and ACLIN 3 states to establish comparable regression models for predicting soil concentrations (Sheppard M.I. 1992).

We established such models for sand and organic soils (Section 6.5.1.1). Organic soils may be formed at the edge of the ice sheet as small lakes evolve into wetlands and bogs characterized by peat soils. Sand soils might predominate in outwash areas after glacial melts. The effective precipitation, Pe (Section 6.5.2.2), was reduced to 0.17 m water$\cdot a^{-1}$ for ACLIN 3 (Section 12.3.2.6). We assumed that chemical exchange in the soil is temperature-independent and so used the median soil solid/liquid partition coefficients, Kd^i (Section 6.5.3), for both ACLIN states.

A new climate representative of ACLIN 3 was defined by calculating the difference between the monthly mean temperatures at Geraldton (Environment Canada 1982b) and Ennadai Lake (Environment Canada 1982a), and by subtracting this difference from each day during a given month. This produced a new data set for ACLIN 3 comparable to that for ACLIN 1 (Section 6.5.2), with a mean annual air temperature of -8°C corresponding to a much shorter growing season in ACLIN 3 than in ACLIN 1. Appropriate ambient vapour pressures were determined for ACLIN 3 (Sheppard M.I. 1992).

TABLE 12-2

COMPARISON OF PARAMETER VALUES FOR ACLIN 1 AND 3 STATES

Parameter Definition	ACLIN	
	1	3
Sedimentation rate, S_{sed} ($kg \cdot m^{-2} \cdot a^{-1}$)	0.16	0.044
C^{14} evasion rate lake to air, η_l^c (a^{-1})	0.92	0.138
Mean annual air temperature (°C)	5	-8
Precipitation, P ($m \cdot a^{-1}$)	0.78	0.29
Runoff, R ($m \cdot a^{-1}$)	0.31	0.12
Soil Depth, Z_s (m)	1.0	1.0
Effective precipitation, Pe ($m \cdot a^{-1}$)	0.47	0.17
Probability of peat fuel use, PT	0.01	1.0
Domestic heating needs, FUELUS ($MJ \cdot s^{-1}$)	3.5×10^{-3}	5.6×10^{-3}
C^{14} evasion rate soil to air, η_s^c (a^{-1})	8.8	4.4
I^{129} evasion rate soil to air, η_s^I (a^{-1})	3.2×10^{-2}	1.6×10^{-2}
Wind speed, UCAV ($m \cdot s^{-1}$)	2.36	3.78
Yield for wood, Yb ($kg \cdot m^{-2}$)	10.5	0.89
Plant interception fraction for wood, rb	1.0	0.1
Plant yield, Y_j ($kg \cdot m^{-2}$)		
TE PLANT	0.8	0.068
TE MEAT	1.0	0.085
TE MILK	0.8	0.068
TE BIRD	1.2	0.102
Above-ground exposure time te_j (d)	50	18 250
Plant environmental halftime, tp (d)	12	18 250
Man's total energy need, En ($kJ \cdot d^{-1}$)	14 600	18 600
Food type energy weighting factor, Ycf_j		
TE PLANT	0.32	0.05
TE MEAT	0.26	0.49
TE MILK	0.36	0.01
TE BIRD	0.05	0.20
FW FISH	0.01	0.25

* ACLIN 1 values are used in the central group of scenarios (Goodwin et al. 1994) and ACLIN 3 values in the glaciation assessment (Section 12.3.3).

Note that the mean annual air temperature used for ACLIN 1 was selected to represent a typical year in terms of precipitation rate (Section 6.5.2.1, Sheppard M.I. 1992). Although the year was typical for precipitation, it was not for temperature, which was warmer than the 1.5 to -0.5°C indicated in Table 12-1. The resulting 13°C mean temperature difference between ACLIN 1 and ACLIN 3 (Table 12-2) reflects a large temperature change so that our assessment includes a correspondingly large difference in climate.

Gaseous emissions from soil are partly mediated by biological activity, which is temperature-dependent. Given a 13°C temperature difference between ACLIN 1 and ACLIN 3, and noting that many physiological functions are approximately halved with a 10°C temperature decrease (Lehninger 1975), we reduced the gaseous evasion rates, η_s^i, from soil to air to 1.6×10^{-2} a^{-1} for ^{129}I and to 4.4 a^{-1} for ^{14}C (Table 12-2).

The remaining parameters of the soil model (Section 6.5) are assumed to be representative of both the ACLIN 1 and ACLIN 3 states.

12.3.2.4 Atmosphere Submodel

The atmosphere model requires few modifications for the glaciation assessment; many of its parameter values are appropriate for both the ACLIN 1 and ACLIN 3 states. For example, the building parameters and those related to indoor air (Section 7.5) are similar for both states, either because they are independent of climate or because the values selected for interglacial conditions are conservative. However, specific ACLIN 3 values are needed for some parameters.

We assumed that peat burning for heating purposes is more common in ACLIN 3, and the probability of doing so, PT (Section 7.5.2.7), was assigned a value of 1.0 rather than 0.01 (Table 12-2) when organic soil is considered. Increased home heating is required in ACLIN 3, and we assumed that FUELUS, the amount of energy required to heat a single family dwelling (Section 7.5.2.1), increases linearly with heating-degree days below 18°C. Long-term averages at Sioux Lookout and Ennadai Lake are 6278 and 9961 heating-degree days (Environment Canada 1982d) and so FUELUS assumes a value of 5.6×10^{-3} $MJ \cdot s^{-1}$ for ACLIN 3, which is 1.6 times higher than for ACLIN 1.

Elson and Webber (1991) indicate that wind speeds will generally increase in the ACLIN 3 state by about a factor of 1.6. Increased wind speeds will enhance atmospheric dilution, thereby decreasing local air concentrations. For dispersion from a chimney (Equation (7.7)), we increased the annual average wind speed, UCAV (Section 7.5.3.1), by a factor of 1.6 to 3.78 $m \cdot s^{-1}$ for ACLIN 3. For dispersion from terrestrial and aquatic area sources, increased wind speeds were accommodated by adjusting UWGHT (Section 7.5.3.2) accordingly, using the new value of UCAV (Equation (7.45)).

Particle suspension, as defined by the terrestrial, ADL (Section 7.5.1.1), and aquatic, AADL (Section 7.5.12), dust loads, was left unchanged. We believe that the values chosen for interglacial conditions are appropriate for both ACLIN 1 and ACLIN 3, even though ACLIN 3 conditions are drier and windier.

A colder ACLIN 3 climate may have an increased frequency of stable atmospheric conditions over terrestrial regions. However, increased wind speeds may counterbalance this, resulting in no appreciable change in the average stability. To be conservative, we assumed that stable conditions, which maximize air concentration, prevail in ACLIN 3. These conditions are stability classes F and G (Amiro 1992b). By rederiving Equation (7.4) to determine the atmospheric dispersion factor, $(DISP)_T$, for these conditions, we found that

$$(DISP)_T = [9.27 \cdot A^{1/8} - 18]/UWGHT \quad . \qquad (12.1)$$

We assumed that stability conditions will not change appreciably over aquatic surfaces, with neutral conditions continuing to predominate. Therefore, Equation (7.5) remains unchanged and applies to both states.

All the other parameter values described in Chapter 7 can be applied to both the ACLIN 1 and the ACLIN 3 states.

12.3.2.5 Food-Chain and Dose Submodel

CALDOS (Chapter 8) can be readily used for the glaciation assessment, except for a few parameter value changes. Human anatomy and physiology are assumed to remain constant in the various glacial states, so many of the food-chain and dose parameters will not change. For example, the DCFs (Section 8.5.2) will not vary with climate. However, there are some parameters, mostly those related to diet and ecosystem type, that will vary.

The dominant native plant community in ACLIN 3 may be transitional between boreal forest and tundra, typical of the vegetation found near the treeline in northern Canada today. From the data on plant yields given by Elson and Webber (1991), we decreased the yield for wood, Yb (Section 8.5.8.2), to 0.89 kg wet biomass$\cdot$m^{-2} land, and the plant interception fraction for wood, rb (Section 8.5.7.2), to 0.1 for ACLIN 3 to reflect a sparsely forested open woodland.

We assumed that the critical group can sustain some agriculture or will rely on wild native plants for food and animal fodder. In either case, the colder climate will decrease yields in comparison with ACLIN 1. Following the recommendations of Elson and Webber (1991), we decreased plant yields, Y_j (Section 8.5.8.1), to 0.068, 0.085, 0.068 and 0.102 kg wet biomass$\cdot$m^{-2} soil for the food types TE PLANT, TE MEAT, TE MILK and TE BIRD respectively.

The critical group may rely more heavily on wild game in ACLIN 3 than in ACLIN 1, or perhaps a native northern species such as caribou may be the major meat source. Native animals may obtain a larger portion of their diet from perennial vegetation that could accumulate and retain nuclides over a longer period than an annual agricultural crop. For instance, lichens are known to retain nuclides efficiently, which can subsequently be transferred via caribou to humans (Whicker and Schultz 1982). To include this pathway, we increased the time of above-ground exposure, te_j (Section 8.5.3.2), for TE PLANT and TE MEAT, and the plant environmental halftime, tp (Section 8.5.3.3), to 50 a or 18 250 d for ACLIN 3. This reflects the long potential times over which some wild plants could accumulate and retain nuclides. A large value of tp will result in radioactive decay becoming a more important depletion term in Equation (8.10).

A colder environment might increase man's total energy need, En (Section 8.5.6.1), so we increased it from 14 600 kJ$\cdot$d^{-1} to 18 600 kJ$\cdot$d^{-1} for ACLIN 3. Also northern diets, typified by today's inhabitants, generally include a large proportion of meat and fish (wild game). Hence, we modified the food type energy weighting factors, Ycf_j (Section 8.5.6.4), to reflect this. The Ycf_j values for ACLIN 3 are TE PLANT = 0.05, TE MILK = 0.01, TE MEAT = 0.49, TE BIRD = 0.2 and FW FISH = 0.25.

The remainder of the food-chain and dose parameters were assumed to vary little between the two glacial states and were not altered.

12.3.2.6 Water Balance Parameters

Elson and Webber (1991) indicate that both precipitation, P, and runoff, R, should decrease in ACLIN 3, although the relationship given in Equation (9.14) still holds for effective precipitation, Pe (Section 6.5.2.2). Following their recommendations, we decreased P to 0.29 m water·a^{-1} and R to 0.12 m water·a^{-1}, so that Pe = 0.17 m water·a^{-1} for ACLIN 3 rather than 0.47 m·a^{-1} for ACLIN 1. This change assumes that glaciers are not actively retreating, which otherwise would increase R substantially. This assumption is conservative because melting would increase dilution.

As indicated in Section 12.3.2.3, irrigation is a potentially important source of contamination for both soil and plants, given the dry conditions in ACLIN 3. For the glaciation assessment, we consider irrigation of the garden only, as governed by the probability of irrigation, PI (Section 6.5.5.2), and the water switch, LW (Section 9.1.2), for both the ACLIN 1 and ACLIN 3 states. More irrigation water would be needed to maintain soil moisture at field capacity because of the drier conditions in ACLIN 3 (Section 6.3.7.2).

12.3.3 Assessment Results

The BIOTRAC results were used to compare the doses predicted for the ACLIN 1 and ACLIN 3 states on a ratio basis using the modified parameter values listed in Table 12-2. For both states, BIOTRAC was driven by the actual postclosure assessment median inputs from the geosphere. These inputs increase with time (Goodwin et al. 1994), so that the times of peak dose for each of the three nuclides considered were partially dependent on the release rates from the geosphere. The ratios of the ACLIN 3 to ACLIN 1 peak doses up to 100 000 a for ^{14}C, ^{129}I and ^{99}Tc are given in Table 12-3 for both organic and sand soils.

TABLE 12-3

DOSE RATIOS FOR MAN FOR ACLIN STATES 1 AND 3 FOR KEY RADIONUCLIDES

Ratio of Peak Annual Doses up to 10^5 a	^{129}I	^{14}C	^{99}Tc
ACLIN 3 / ACLIN 1 (organic soil)	0.45	1.6	6.5
ACLIN 3 / ACLIN 1 (sand soil)	1.62	5.7	0.70
Time of peak dose (a)	10^5	6×10^4	10^5

Note: ACLIN 1 corresponds to present-day interglacial conditions, represented by the central group of scenarios, and ACLIN 3 to cold interstadial conditions.

The ACLIN 3 environment produces slightly lower predicted doses than ACLIN 1 for organic soil and slightly higher doses for sand soil for ^{129}I (Table 12-3). The effect is opposite for ^{99}Tc and somewhat more extreme for organic soil, though ^{99}Tc doses were relatively insignificant. Doses from ^{14}C increase by up to a factor of six for ACLIN 3 compared with ACLIN 1, depending on the soil type. For ^{129}I and ^{14}C, the soil/plant/man pathway (Section 8.3.1.1) contributes the greatest portion of the dose in ACLIN 1, whereas the air/plant/man pathway, involving deposition of irrigation water to vegetation (Section 8.3.1.3), is most important in ACLIN 3. These differences are primarily caused by the use of more irrigation water containing nuclides in the drier ACLIN 3 state.

If we assume that the geosphere is unaffected by glaciation, BIOTRAC predicts that doses to humans in an ACLIN 3 environment would differ only slightly from those for present-day ACLIN 1 conditions. Doses from individual nuclides will vary slightly. However, with the total dose to man dominated by ^{129}I for times up to 10 000 a (Chapter 10, Goodwin et al. 1994), a critical group living in an ACLIN 1 or ACLIN 3 environment would be exposed to similar doses.

12.4 SUMMARY

Glaciation is a major trauma to the biosphere. The environmental consequences, as exhibited by ecosystem change to glaciation, are immense, and would greatly exceed perturbations caused by small amounts of nuclides originating from the vault. The cultural impact of glaciation on human development and habits would also be large, effectively dwarfing potential radiological consequences.

Regardless of the large physical impact of glaciation, we have quantitatively assessed the potential implications of glaciation on the dose to humans. Our comparative analysis indicates that the ACLIN 1 and ACLIN 3 states will likely result in similar total doses. The reason for this similarity is that glacially induced biosphere changes do not affect any of the highly sensitive parameters such as the lake/well-water switch, LW (Section 10.3). The intermediate ACLIN 2 state should result in doses that lie between those predicted for ACLIN states 1 and 3. ACLIN 4 is a full glacial state and humans are assumed not to inhabit the disposal facility region. The closest possible critical group would be at the edge of the ice sheet, effectively experiencing an ACLIN 3 environment, but with lower doses because of the large distance from the disposal facility location. Transient events associated with the evolution of the glacial cycle are not expected to introduce processes that could increase the dose to humans. The major transient process during a glacial melt is increased runoff, which should decrease local nuclide concentrations through flushing and dilution.

The central question addressed in this section is whether glaciation could lead to substantially increased doses to humans as far as the biosphere is concerned. Our analysis shows that this is unlikely because doses for various glacial states tend to be similar. To the extent that human doses also apply to other organisms, these conclusions can be broadened to include all biota. The question of the effects of glaciation on the combined vault, geosphere and biosphere system is addressed by Goodwin et al. (1994).

13. ENVIRONMENTAL PROTECTION

13.1 INTRODUCTION

13.1.1 The Approach to Environmental Protection

Protection of the environment focuses on maintaining the quality of water, soil and air, and on preventing adverse impacts on the biota from radiological and chemical contaminants to assure normal survival, reproduction and growth. Our assessment methodology for environmental protection involves three quantitative approaches to help evaluate potential effects of a disposal facility on the environment. These approaches include relating human safety criteria to potential environmental impacts, predicting nuclide concentrations in soil, water and air, and estimating radiological doses to several generic target organisms (Table 13-1). Each of these approaches uses predicted outputs of BIOTRAC and is consistent with the concepts outlined in Chapters 1 to 12. In Section 13.4, we also show how our dose assessment approach for generic target organisms can be applied to specific target species. We are concerned here only with the presentation of our methodology. The postclosure assessment results based on this methodology are presented by Goodwin et al. (1994).

Our three approaches lend themselves to direct or indirect environmental monitoring, an important aspect of our concept (Simmons et al. 1994). This is particularly true for nuclide concentrations in soil, water and air.

TABLE 13-1

TYPE OF IMPACTS CONSIDERED BY THE THREE APPROACHES USED FOR ASSESSING ENVIRONMENTAL EFFECTS

Method to Evaluate Impact	Chemical Toxicity	Radiological Toxicity	Quantitative Prediction From BIOTRAC	Ecological Organizational Level Addressed
Relation to Human Safety		X	Radiological Dose to Humans	Population
Nuclide Concentrations in Environment	X	X	Concentrations in Soil, Water and Air	Population Community Ecosystem
Dose Prediction to Biota		X	Radiological Dose to Generic Target Organisms (Plant, Mammal, Bird and Fish)	Individual

Doses to humans and other biota are difficult to monitor directly. However, these doses are a function of nuclide concentrations, and so can also be monitored through soil, water and air, and more directly through nuclide concentrations in plant and animal tissues. Given these concentrations, doses can be readily calculated and evaluated.

The primary effects of radiological and chemical toxicants occur at the molecular level. Depending on severity, the effects may then be expressed at various biological and ecological organizational levels from the cell to the individual to the ecosystem. For humans, the individual is of prime importance, and so we have adopted the critical group concept for assessing effects (Section 1.5.4). For non-human organisms, the welfare of the population is usually more important than the survival of the individual. However, if there are no biological effects on individuals, then there will likely be no impact on the higher ecological organizational levels such as populations, communities and ecosystems. Robustness generally increases with a higher level of organization, and many ecosystems exhibit a high degree of resilience (Beanlands and Duinker 1983). Therefore, although higher organizational groupings are complex and difficult to model, they should be implicitly protected if individual plants and animals, and their populations, are protected.

In Table 13-1, we have indicated the organizational level explicitly considered by each of our approaches. All of them apply to radiological toxicity, but only the prediction of nuclide concentrations in environmental media is used to evaluate chemical toxicity. Figure 13-1 demonstrates how radiation acts primarily at the molecular level and how these effects may propagate up organizational levels that exhibit increasing complexity and robustness. It also indicates that explicit protection of individual generic target organisms affords implicit protection at the higher levels.

Here, we present each of our approaches and their application to environmental protection. The output and potential impacts during the postclosure phase are described in Goodwin et al. (1994).

13.1.2 Protection of the Environment Related to Human Radiation Safety

The level of safety required for the protection of human individuals from radiological effects is thought likely to be adequate to protect other species, although not necessarily individual members of those species (ICRP 1977, AECB 1987). Recently, the ICRP (1991a) indicated that the standard of environmental control needed to protect man to the degree currently thought desirable will ensure that other species are not put at risk. However, it is possible that individual members of non-human species might be harmed occasionally, but not to the extent of endangering whole species or creating imbalances between species. This belief or assumption recognizes that mammals are the most radiosensitive group of organisms (Whicker and Schultz 1982) and that humans are very long-lived, so that latent effects have sufficient time to appear. The assumption has been examined by several expert committees using various calculations and modelling procedures (NCRP 1991, IAEA 1992, UNSCEAR 1992). Myers (1989) also carried out an evaluation with specific reference to geological disposal of Canadian nuclear fuel wastes. The assumption was supported in all of these

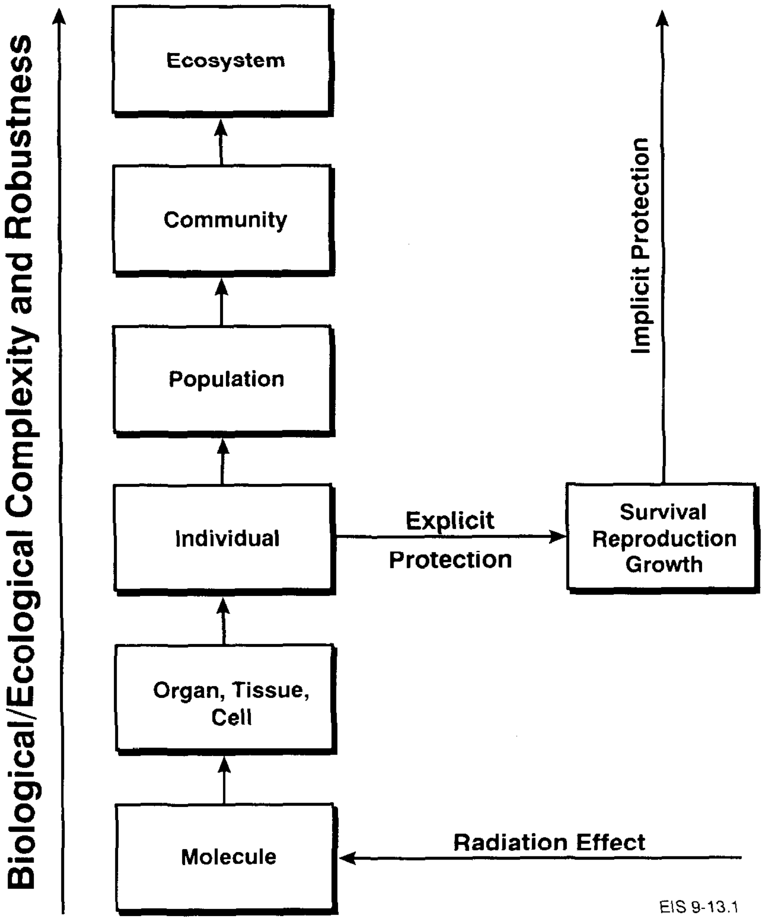


FIGURE 13-1: Schematic Representation of Rationale for Selecting Individual Generic Target Organisms for Assuring Radiation Protection at All the Ecological Organizational Levels

analyses and it was concluded that radiological protection of humans should, in general, also implicitly protect the environment.

The assumption that human protection is adequate to protect other species would only be valid if both humans and other biota inhabit the same part of the environment (Zach et al. 1993). In some instances, non-human organisms could be exposed to higher water, sediment, soil or air concentrations because of habitat or lifestyle differences, and so there could be an impact on certain species without a concomitant impact on humans (Thompson 1988). However, in our assessment, humans are assumed to inhabit the most contaminated parts of the environment (Section 1.5.4), and share the risk there with other biota. Therefore, humans are an appropriate indicator species for the postclosure assessment, and establishment of human protection helps to ensure that other species are protected as well.

13.1.3 Nuclide Concentrations Related to Regulatory Criteria and Baseline Data

Various agencies have established environmental criteria for chemically toxic nuclides and radionuclides (CWQG 1987, OME 1988, AEL 1991). These criteria are based on concentrations in a physical environmental media such as surface water, soil or air. BIOTRAC predicts nuclide concentrations in surface water (Chapter 5), soil (Chapter 6), and air (Chapter 7), and these concentrations can be readily compared with the criteria to demonstrate acceptability.

Where regulatory criteria are unavailable, the predicted concentrations can be compared with environmental baseline data for each nuclide from the Canadian Shield. These baseline data include estimates of mean background concentrations and variability in these concentrations (Amiro 1992a, 1992c). The predicted concentrations can also be compared with known concentrations where some environmental effect has been observed, or with concentrations where no effect was seen (Amiro 1992a).

BIOTRAC estimates concentrations in several abiotic media or compartments. The biota living in and around these media are dependent on their quality so that the assessment of protection of the biota is directly tied to predictions of nuclide concentrations by BIOTRAC.

13.1.4 Radiological Doses to Non-Human Organisms

For humans, the calculation of radiological dose is one of the main end points of the postclosure assessment because dose can be rigorously related to risk (Section 1.2.3). Doses can also be calculated for non-human organisms based on the transport pathways modelled in BIOTRAC. However, a rigorous methodology does not exist to relate radiological dose to risk for non-human biota. Despite this, predicted doses can be related to observed doses where effects have been measured in the field or laboratory, or to doses where no effects could be detected (e.g., Amiro and Dugle 1985, Dugle 1986, Rose 1992, Zach et al. 1993). Doses resulting from multiple pathways for many radionuclides are additive so that the dose is an integrated measure of the total potential impact. This is the same as in humans (Section 8.3.3).

In Section 13.3, we outlined the model for calculating doses to four generic non-human target organisms. This model is an integral part of BIOTRAC, and it interfaces smoothly with the four submodels presented in Chapters 5 to 8. The model's output provides a quantitative measure of the potential radiotoxicity that could affect non-human biota. In Section 13.4, we show how our approach for generic organisms can be applied to specific terrestrial and aquatic target species.

13.2 NUCLIDE CONCENTRATIONS IN THE ENVIRONMENT

As discussed briefly in Section 13.1.2, BIOTRAC predicts concentrations in compartments related to the transport of nuclides from the vault to humans and to other organisms. These compartments can reflect components of either natural ecosystems or systems greatly modified by human activities. Biota can be exposed to nuclides in both cases; for example, domestic animals could inhabit agricultural land, whereas wild animals such as moose could inhabit a forest environment. By comparing the nuclide concentrations in ecosystem components predicted by BIOTRAC with regulatory criteria, standards or guidelines, or even with background concentrations on the Canadian Shield, we can evaluate the environmental acceptability of the disposal concept.

To simplify the broad range of environments that can be defined for organisms, we assume that all biota inhabit the immediate discharge zone of the disposal facility, where water, sediment, soil and air are potentially most contaminated (Section 1.5.4). The corresponding nuclide concentrations for determining environmental quality in each compartment are given in Table 13-2. We assume that the lake-water concentration is more appropriate than the well-water concentration for assessing environmental quality affecting non-human biota. We use either the soil or lake sediment concentration, whichever is greater for any soil type (Section 6.5.1.1). This reflects the possibility that biota can inhabit lake sediments, a drained lake bottom, or colonize an agricultural area that was developed using dredged sediments. Air concentrations are calculated to reflect the total concentration from all sources that an organism could experience outdoors (Section 7.3). This includes anthropogenic sources such as release of contaminated smoke from a chimney.

TABLE 13-2

NUCLIDE CONCENTRATIONS FOR ASSESSING ENVIRONMENTAL QUALITY

Compartment	Concentration	Equation or Table
Surface Water	C_l^i	5.6
Sediment or Soil	C_{se}^i or C_s^i, whichever is greater for any soil type	5.15, Table 6-6
Air (outdoor only)	C_a^i	7.11, 7.12, 7.13, 7.14, 7.16, 7.19, 7.20, 7.21, 7.22, 7.23, 7.26, 7.27, 7.28, 7.30, Table 7-1

These conservative assumptions for assessing general environmental quality aim to protect plant and animal populations, communities, and ecosystems (Table 13-1). Many natural plants and animals would experience lower concentrations than those predicted by BIOTRAC. For example, the most contaminated soils will likely occur where contaminated groundwater is used for irrigation (Section 6.3.7.2), but this would mostly affect domestic plants and animals.

BIOTRAC is a probabilistic model and the predicted nuclide concentrations are based on a large number of simulations incorporating variability and uncertainty in the parameters (Section 1.5.7). Nuclide concentrations in surface water, soil and air are time-dependent, and we use the maximum concentrations experienced during a simulation (e.g., the maximum concentrations calculated within 10 000 a). These concentration maxima, even if they occur at different times, are then averaged over the total number of simulations to give mean nuclide concentrations. These concentrations are then used to evaluate environmental quality in a general way that includes all biota (Goodwin et al. 1994).

13.3 THE MODEL OF RADIOLOGICAL IMPACTS ON NON-HUMAN BIOTA

13.3.1 Target Organisms and Selection of Nuclides

Non-human biota can inhabit any of the physical compartments modelled in BIOTRAC. These biota include plants, animals and microorganisms living in water, sediment, soil and air as part of aquatic and terrestrial ecosystems. We do not and cannot explicitly evaluate individual radiological doses for each plant and animal species on the Canadian Shield. This is not only because of the large effort involved, but also because the detailed ecological, physiological and radiological data for many species are unknown. Furthermore, without a specific site it is difficult to decide on the most relevant species. Fortunately, it is possible to define suitable generic target organisms that represent a broad range of species and habitats, and we have defined four such organisms (Figure 13-2).

1. A plant with nuclide uptake characteristics similar to a broad range of terrestrial vascular plants. This would include many grasses, herbs and trees. The plant can be immersed in contaminated soil, air and water, so it reflects the external exposures received by both terrestrial and aquatic species.

2. A mammal most similar to a herbivore in its eating habits. Typical species would include caribou, moose, beaver and meadow vole. The mammal can be immersed in contaminated air, soil and water, thereby reflecting external exposures received by terrestrial land, soil-burrowing and aquatic mammals. To a large extent, most mammalian predators are also included here.

3. A bird most similar to a terrestrial species that eats seeds and fruit. Typical species would include the ruffed grouse, song sparrow, and evening grosbeak. The bird can be immersed in air, soil and water, corresponding to a wide range of terrestrial and aquatic species. Thus, waterfowl would also be included to some extent.

4. A fish representing a wide range of free-swimming (pelagic) and bottom-feeding (benthic) species. This would include characteristics of diverse species such as lake trout, walleye, northern pike, lake whitefish and white sucker. The fish can be immersed in either water or sediment.

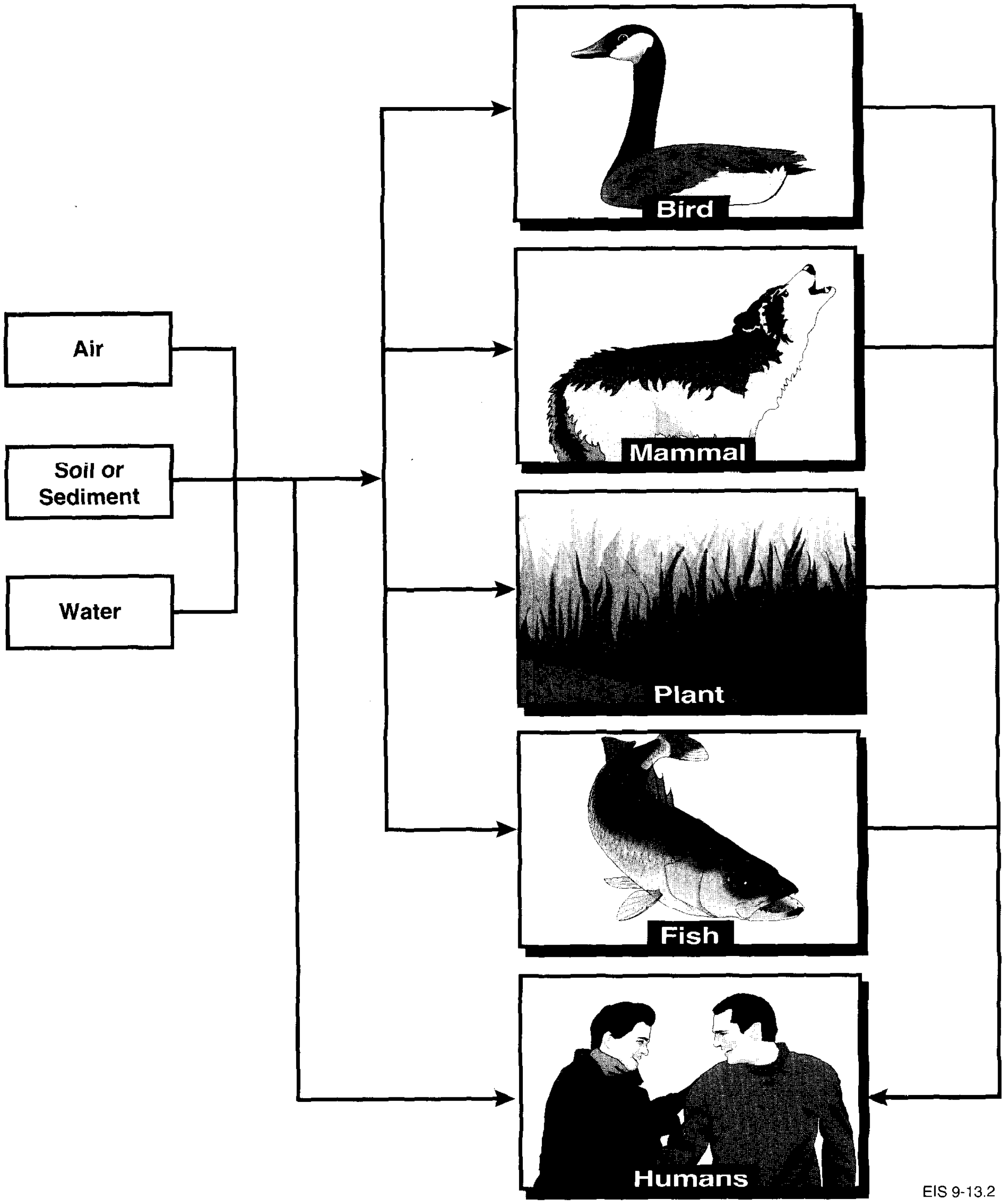


FIGURE 13-2: Schematic Representation of the Exposure Situation for the Four Generic Target Organisms and Humans in Relation to Each Other and the Three Main Physical Biosphere Compartments (Water, Soil or Sediment, and Air)

The radiological dose to each of these target organisms is predicted so that the doses can be compared with known responses exhibited by the corresponding target groups.

Our four generic target organisms do not explicitly include all the major groups of organisms that occur on the Canadian Shield. Among others, invertebrates, amphibians and reptiles are conspicuously missing largely because of the lack of appropriate data, which is a result of the tendency to study organisms that are of direct importance to humans. Some of the missing groups may partially overlap with our target organisms, e.g., amphibians with fish and reptiles with birds. We have no indication that the missing groups of organisms are particularly radiosensitive (Whicker and Schultz 1982). As noted in Section 13.3.2, in terms of external exposure all organisms are included regardless of group.

In the two interim assessments (Wuschke et al. 1981, Wuschke et al. 1985a) and in the present assessment (Section 10.3, Goodwin et al. 1994), almost all of the radiological dose to man during postclosure results from ^{129}I. The next two most important nuclides are ^{14}C and ^{99}Tc. The importance of these three nuclides relative to all the others (Table 1-1) is largely a function of the magnitude of the flux from the geosphere to the biosphere. Hence, these three nuclides should also cause most of the dose to non-human biota. Therefore, we selected ^{14}C, ^{129}I and ^{99}Tc to calculate the total radiological impact on the target organisms. These three nuclides were also used in the sensitivity analysis of BIOTRAC (Chapter 10) and in our glaciation assessment (Chapter 12).

13.3.2 Qualitative Description of the Model

The model is fully integrated into BIOTRAC and is analogous to the food-chain and dose submodel, CALDOS (Chapter 8). Nuclide concentrations in lake water, soil and air are calculated as described in Chapters 5 to 7. These concentrations are used to calculate doses from exposure of the target organism to external radiation sources by using DCFs. The concentrations in surface water, soil and air are also used to calculate nuclide concentrations in the tissues of the target organisms through food-chain transfer. These concentrations are then used to calculate doses from exposure to internal sources with the help of DCFs. These methods are very similar to those in CALDOS and use the same basic methodology expressed in Equation (8.1) and in Section 8.2.

Food-chain transfer is considered for internal exposure. Specific pathways are included whereby nuclides are transferred from water, soil and air to the biota. The plant is a primary producer and can receive nuclides from direct uptake from soil and from deposition onto the leaves from the atmosphere and from aerial irrigation water. The fish can take up nuclides from the water and indirectly from sediments. The terrestrial animals (mammals and birds) are primary consumers and can accumulate nuclides from ingestion of water, vegetation and soil. Higher levels of consumers such as predators are also included, to a certain extent, in the broad PDFs of our transfer coefficients (Section 8.5.1, Zach and Sheppard 1992).

For external exposures, we assume that plants can be immersed in air, soil and water. This reflects a broad range of terrestrial and aquatic plants, and also considers exposure of the roots, shoots, leaves and reproductive parts. External exposures of mammals and birds reflect a broad range of animals. We include immersion in water, soil, air and vegetation, and therefore encompass all terrestrial animals, semi-aquatic animals and animals inhabiting shallow waters. For example, external radiological doses to diverse animal groups such as owls, ducks, frogs, snakes, wolves, otter and deer are considered. Fish can receive an external radiation dose from water and from sediments. The inclusion of a broad range of biota in these external exposure calculations is possible through the use of conservative assumptions in the application of the external DCFs. We conservatively allow the organisms to live simultaneously in each of the relevant habitats so that the occupancy factor for each is unity and need not be considered explicitly.

Our model calculates whole-body doses, except for internal exposure of animals to ^{129}I. This nuclide is exceptional because iodine tends to accumulate in the thyroid gland (Section 2.5.3); hence we consider the dose to that gland.

13.3.3 Mathematical Formulation of the Model

As in the case of CALDOS, all the nuclide concentrations are expressed in becquerels (Bq), whereas the corresponding unit in the other submodels of BIOTRAC is the mole (Section 8.3). The use of Bq simplifies the application of the DCFs, which conventionally are expressed in radioactivity units.

The radiological dose for the target organisms is calculated in units of grays per year ($Gy \cdot a^{-1}$). This differs from the units used for humans ($Sv \cdot a^{-1}$) because of the radiation quality factor, Q. This factor accounts for the biological effectiveness of various radiation types (Section 1.2.3, Zach and Sheppard 1992) and is not needed when calculating doses for ^{14}C, ^{129}I and ^{99}Tc because it would have a value of one (ICRP 1977).

In Section 8.3 describing CALDOS, many of the equations include a term that accounts for ingrowth of nuclides from precursors. This only applies to nuclides with half-lives between 1 d and 20 a (Table 2-1). Thus, it is not considered here because the half-lives of ^{14}C, ^{129}I and ^{99}Tc are much longer (Table 1-1). More importantly, none of these nuclides is a daughter product and all have stable progeny.

Also in Section 8.3, many of the food-chain equations include an exponential term for radiological decay during a holdup time. This holdup time allows for some decay of radionuclides before they are ingested by humans. This term is not included in the equations for calculating dose to non-human organisms because these organisms usually consume food and water directly without delay. This is conservative for some animals that store food, such as beavers, squirrels and blue jays.

In the equations of our model, soil concentrations of nuclides are designated by C^i_s, which relates to the soil model (Table 6-6). However, soil may also be based on sediment concentrations, $C^i_{s\bullet}$, as calculated in Equation (5.15). The probability of sediment use as soil is given in Section 6.5.5.4. Several equations involve domestic water for which the nuclide concentration is designated by C_w^i. Domestic water may be derived from the bedrock well, C_{ww}^i (Equation 4.18), or the lake, C_l^i (Equation 5.6). The choice of the water source is explained in Section 9.1.2. Note that the rate of deposition of nuclides to vegetation, D_b^i, occurs in several equations and is defined in Equations (7.41) and (7.43).

13.3.3.1 Internal Exposure Pathways

Internal Exposure of Plants

Plants can accumulate nuclides from the soil via root uptake and from aerial deposition. We conservatively assume that the plant is located in the garden, which has the greatest soil nuclide concentration (Section 6.3.7) and also receives irrigation water, often from a potentially contaminated well (Section 6.3.7.2).

The internal dose to plants from nuclide i, DBi_p^i (Gy·a^{-1}), is given by

$$DBi_p^i = \left\{C_s^i \cdot B_v^i + \left[D_b^i \cdot (r_j/Y_j) \cdot \left[1 - \exp(-\lambda_E^i \cdot te_j)\right]/\lambda_E^i\right]\right\} \cdot (DFB_p^i)_I \qquad (13.1)$$

where $(DFB_p^i)_I$ is the dose conversion factor for internal exposure of plants to nuclide i ((Gy·a^{-1})/(Bq·kg^{-1} wet biomass)).

The other symbols are defined in Equations (8.2) and (8.8).

Internal Exposure of Terrestrial Animals

Terrestrial animals, specifically the target mammal and bird, can accumulate nuclides internally through ingestion of contaminated food, water and soil. We assume that the inhalation dose is negligible compared with the ingestion dose for the three nuclides modelled.

The internal dose to terrestrial animals from nuclide i, DBi_b^i (Gy·a^{-1}), is given by

$$\begin{aligned} DBi_b^i = \Bigg[&\left[C_s^i \cdot Bv^i \cdot F_j^i \cdot Qf_j\right] \\ &+ \left\{D_b^i \cdot \left[r_j/Y_j\right] \cdot F_j^i \cdot Qf_j \cdot \left[1 - \exp\left[-\lambda_E^i \cdot te_j\right]\right]/\lambda_E^i\right\} \\ &+ \left[C_s^i \cdot F_j^i \cdot Qs_j\right] + \left[C_w^i \cdot F_j^i \cdot Qdw_j\right]\Bigg] \cdot (DFB_b^i)_I \end{aligned} \qquad (13.2)$$

where DBi_b^i refers to either mammals, DBi_M^i, or birds, DBi_B^i,

$(DFB_b^i)_I$ is the internal dose conversion factor for nuclide i for either mammals, $(DFB_M^i)_I$, or birds, $(DFB_B^i)_I$ ((Gy·a^{-1})/(Bq·kg^{-1} wet biomass)), and

j designates the food types (Section 8.1) TE MEAT for mammals and TE BIRD for birds.

All the other symbols in Equation (13.2) follow those in Equations (8.6), (8.12), (8.14) and (8.16). The food types TE MEAT and TE BIRD relate to terrestrial animal transfer coefficients (Section 8.5.1) and ingestion rates (Section 8.5.5).

Internal Exposure of Fish

As described in Section 8.3.1.7, fish inhabiting the discharge lake may become contaminated through ingestion of food and sediment, and through osmotic exchange of fluids. The dose to fish from nuclide i from internal exposure, DBi_F^i ($Gy \cdot a^{-1}$), is given by

$$DBi_F^i = C_l^i \cdot B_j^i \cdot (DFB_F^i)_I \qquad (13.3)$$

where $(DFB_F^i)_I$ is the dose conversion factor for internal exposure of fish to nuclide i (($Gy \cdot a^{-1}$)/($Bq \cdot kg^{-1}$ wet biomass)).

All the other symbols in Equation (13.3) are defined in Equation (8.18). The concentration ratio, B_j^i, refers to j = FW FISH and includes a variety of sources for intakes by fish within its broad PDF (Section 8.5.1.3).

13.3.3.2 External Exposure Pathways

The target organisms can be immersed in four different media, depending on their habitat. The relevant media are given in Table 13-3 for each target organism. The radiation sources for external exposure are similar to those for humans, although the exposure situation is different to reflect a broad range of plants and animals.

TABLE 13-3

SOURCES OF EXTERNAL EXPOSURE FOR GENERIC TARGET ORGANISMS

Source	Organism			
	Fish	Plant	Mammal	Bird
Water Immersion (W)	X	X	X	X
Air Immersion (A)		X	X	X
Soil or Sediment Immersion (S)	X	X	X	X
Vegetation Immersion (V)			X	X

Immersion in Water

All the target organisms are assumed to be immersed in water. Fish spend their whole lives in water; plants can be aquatic or have their roots submerged in soil pore-water; and a large range of animals can spend a portion of their time swimming, wading or lying in water.

For plants, the dose from nuclide i from immersion in water, $(DBe^i_P)_W$ ($Gy \cdot a^{-1}$), is given by

$$(DBe^i_P)_W = C^i_W \cdot (DFB^i_P)_W \qquad (13.4)$$

where C^i_W is the concentration of nuclide i in domestic water ($Bq \cdot m^{-3}$ water), and

$(DFB^i_P)_W$ is the water immersion dose conversion factor for plants for nuclide i (($Gy \cdot a^{-1}$)/($Bq \cdot m^{-3}$ water)).

This means that the plant can be fully immersed in water originating from the bedrock well or from the lake. Full immersion includes exposure from irrigation.

For mammals, the dose from nuclide i from immersion in water, $(DBe^i_M)_W$ ($Gy \cdot a^{-1}$), is given by

$$(DBe^i_M)_W = C^i_W \cdot (DFB^i_M)_W \qquad (13.5)$$

where $(DFB^i_M)_W$ is the water immersion dose conversion factor for mammals for nuclide i (($Gy \cdot a^{-1}$)/($Bq \cdot m^{-3}$ water)).

As in the case of plants, mammals can be exposed to water originating from either the well or from the lake. So both domestic and wild mammals are fully considered.

For birds, the dose from nuclide i from immersion in water, $(DBe^i_B)_W$ ($Gy \cdot a^{-1}$), is given by

$$(DBe^i_B)_W = C^i_W \cdot (DFB^i_B)_W \qquad (13.6)$$

where $(DFB^i_B)_W$ is the water immersion dose conversion factor for birds for nuclide i (($Gy \cdot a^{-1}$)/($Bq \cdot m^{-3}$ water)).

Birds are also exposed to well or lake water, so that both wild and domestic birds are included.

For fish, the dose from nuclide i from immersion in water, $(DBe^i_F)_W$ ($Gy \cdot a^{-1}$), is given by

$$(DBe^i_F)_W = C^i_l \cdot (DFB^i_F)_W \qquad (13.7)$$

where $(DFB^i_F)_W$ is the water immersion dose conversion factor for fish for nuclide i (($Gy \cdot a^{-1}$)/($Bq \cdot m^{-3}$ water)).

Fish are only exposed to lake water.

Immersion in Soil or Sediment

Plants are rooted in soil and they may also accumulate soil particles on foliage from aerial deposition; terrestrial animals may lie or burrow in soil or be covered with contaminated soil particles; and bottom-feeding fish can be essentially immersed in sediments. Therefore, we assess the external exposure of all the target organisms to soil or sediment (Section 6.5.5.4).

For plants, the dose from nuclide i from immersion in soil, $(DBe^i_P)_S$ $(Gy \cdot a^{-1})$, is given by

$$(DBe^i_P)_S = C^i_s \cdot (DFB^i_P)_S \tag{13.8}$$

where C^i_s is the concentration of nuclide i in the root zone of the garden $(Bq \cdot kg^{-1}$ dry soil), and

$(DFB^i_P)_S$ is the soil immersion dose conversion factor for plants for nuclide i $((Gy \cdot a^{-1})/(Bq \cdot kg^{-1}$ dry soil)).

This means that the plant is exposed to the field with the greatest nuclide concentration (Section 6.3.7), which may be enhanced through contaminated well or lake water from irrigation (Table 6-6). This allows the evaluation of doses to irrigated plants and is likely conservative for all other plants.

For mammals, the dose from nuclide i from immersion in soil, $(DBe^i_M)_S$ $(Gy \cdot a^{-1})$, is given by

$$(DBe^i_M)_S = C^i_s \cdot (DFB^i_M)_S \tag{13.9}$$

where C^i_s is the concentration of nuclide i in the root zone of the forage field $(Bq \cdot kg^{-1}$ dry soil), and

$(DFB^i_M)_S$ is the soil immersion dose conversion factor for mammals for nuclide i $((Gy \cdot a^{-1})/(Bq \cdot kg^{-1}$ dry soil)).

The forage field soil is used to reflect the habitat of most domestic and wild terrestrial mammals grazing in an agricultural area. Forest soils have either comparable or lower nuclide concentrations, so the selection of the forage field for wild animals is likely conservative.

For birds such as the burrowing owl, the dose from nuclide i from immersion in soil, $(DBe^i_B)_S$ $(Gy \cdot a^{-1})$, is given by

$$(DBe^i_B)_S = C^i_s \cdot (DFB^i_B)_S \tag{13.10}$$

where $(DFB^i_B)_S$ is the soil immersion dose conversion factor for birds for nuclide i $((Gy \cdot a^{-1})/(Bq \cdot kg^{-1}$ dry soil)).

The soil concentration used here is also based on the forage field, as in the case of mammals.

For fish, the dose from nuclide i from immersion in sediment, $(DBe^i_F)_S$ $(Gy \cdot a^{-1})$, is given by

$$(DBe^i_F)_S = C^i_{s\bullet} \cdot (DFB^i_F)_S \tag{13.11}$$

where $C^i_{s\bullet}$ is the concentration of nuclide i in the top 0.3 m of sediments $(Bq \cdot kg^{-1}$ dry sediment), and

$(DFB^i_F)_S$ is the sediment immersion dose conversion factor for fish for nuclide i $((Gy \cdot a^{-1})/(Bq \cdot kg^{-1}$ dry sediment)).

The calculation of $C^i_{s\bullet}$ is documented in Equation (5.15).

Immersion in Air

We assume that plants, mammals and birds can be immersed in air, but fish cannot (Table 13-3). In all cases, the organisms are exposed to outdoor air concentrations and all the concentrations of nuclide i in air, C^i_a, refer to the sum from all the outdoor contributions (Section 7.3.7).

For plants, the dose from nuclide i from immersion in air, $(DBe^i_P)_A$ $(Gy \cdot a^{-1})$, is given by

$$(DBe^i_P)_A = C^i_a \cdot (DFB^i_P)_A \tag{13.12}$$

where $(DFB^i_P)_A$ is the air immersion dose conversion factor for plants for nuclide i $((Gy \cdot a^{-1})/(Bq \cdot m^{-3}$ air)).

For mammals, the dose from nuclide i from immersion in air, $(DBe^i_M)_A$ $(Gy \cdot a^{-1})$, is given by

$$(DBe^i_M)_A = C^i_a \cdot (DFB^i_M)_A \tag{13.13}$$

where $(DFB^i_M)_A$ is the air immersion dose conversion factor for mammals for nuclide i $((Gy \cdot a^{-1})/(Bq \cdot m^{-3}$ air)).

For birds, the dose from nuclide i from immersion in air, $(DBe^i_B)_A$ $(Gy \cdot a^{-1})$, is given by

$$(DBe^i_B)_A = C^i_a \cdot (DFB^i_B)_A \tag{13.14}$$

where $(DFB^i_B)_A$ is the air immersion dose conversion factor for birds for nuclide i $((Gy \cdot a^{-1})/(Bq \cdot m^{-3}$ air)).

Immersion in Vegetation

Terrestrial animals can be surrounded or immersed in vegetation during much of their lives. For example, many birds nest and roost in vegetation, many small mammals burrow and feed in vegetation, domestic animals may bed on straw, and forest dwellers are surrounded by trees and other plants. We assess the dose from external exposure to vegetation that has been contaminated from a variety of sources. We assume that exposure to contaminated vegetation is unimportant for fish because most of their external dose should come directly from water and sediment, and water forms an efficient shield against radiation penetration from remote sources.

We do not calculate external exposure to a plant from other plants. This situation is implicitly included in our DCF, which conservatively assume that all the radiation is absorbed by the plant (Section 13.3.5.3, Amiro 1992c). Therefore, we only calculate external exposure for mammals and birds from vegetation (Table 13-3).

Plant concentrations are calculated using the methods outlined in Sections 8.3.1.1 and 8.3.1.3. Vegetation can receive nuclides from the soil through root uptake and through aerial deposition. We assume that the animals inhabit the forage field. This field is rarely irrigated, and if so, only by lake water, so its plant concentrations tend to be lower than those in the garden. The forage-field plant concentrations are similar or greater than those in the woodlot, so the use of the forage field is reasonable and likely conservative. The forage field is also used for soil immersion of animals (Equations 13.9 and 13.10).

The concentrations in vegetation are calculated using both soil and air pathways, and the resulting dose is given by

$$(DBe_b^i)_V = \left\{ C_s^i \cdot Bv^i + D_b^i \cdot (r_j/Y_j) \cdot \left[1 - \exp(-\lambda_E^i \cdot te_j) \right] / \lambda_E^i \right\} \cdot (DFB_b^i)_V \qquad (13.15)$$

where $(DBe_b^i)_V$ is the dose from nuclide i for mammals, $(DBe_M^i)_V$, or birds, $(DBe_B^i)_V$, from immersion in vegetation ($Gy \cdot a^{-1}$), and

$(DFB_b^i)_V$ is the dose conversion factor for nuclide i for mammals, $(DFB_M^i)_V$, or birds, $(DFB_B^i)_V$, from external exposure to vegetation ($(Gy \cdot a^{-1})/(Bq \cdot kg^{-1}$ wet biomass)).

Determination of Total External Doses

The total external doses, $(DBe_b^i)_{Tot}$ ($Gy \cdot a^{-1}$), for each of our four target organisms are calculated by summing the contributions from all the sources (Table 13-3).

Therefore the total external dose for nuclide i for fish is given by

$$(DBe_F^i)_{Tot} = (DBe_F^i)_W + (DBe_F^i)_S \quad . \qquad (13.16)$$

The total external dose for nuclide i for plants is given by

$$(DBe_P^i)_{Tot} = (DBe_P^i)_W + (DBe_P^i)_S + (DBe_P^i)_A \quad . \qquad (13.17)$$

The total external dose for nuclide i for mammals is given by

$$(DBe_M^i)_{Tot} = (DBe_M^i)_W + (DBe_M^i)_S + (DBe_M^i)_A + (DBe_M^i)_V \quad . \qquad (13.18)$$

The total external dose for nuclide i for birds is given by

$$(DBe_B^i)_{Tot} = (DBe_B^i)_W + (DBe_B^i)_S + (DBe_B^i)_A + (DBe_B^i)_V \quad . \tag{13.19}$$

13.3.3.3 Calculation of Total Radiological Doses

The total dose, $(DB_b)_{TT}$ $(Gy \cdot a^{-1})$, for each target organism is simply the sum of all external and internal doses from all nuclides.

The total dose to fish is given by

$$(DB_F)_{TT} = \sum_{i=1}^{m} \left[((DBe_F^i)_{Tot} + DBi_F^i) \right] \quad . \tag{13.20}$$

The total dose to plants is given by

$$(DB_P)_{TT} = \sum_{i=1}^{m} \left[((DBe_P^i)_{Tot} + DBi_P^i) \right] \quad . \tag{13.21}$$

The total dose to mammals is given by

$$(DB_M)_{TT} = \sum_{i=1}^{m} \left[((DBe_M^i)_{Tot} + DBi_M^i) \right] \quad . \tag{13.22}$$

The total dose to birds is given by

$$(DB_B)_{TT} = \sum_{i=1}^{m} \left[((DBe_B^i)_{Tot} + DBi_B^i) \right] \quad . \tag{13.23}$$

Note that the number of nuclides, m, has a value of 3 to account for ^{14}C, ^{129}I and ^{99}Tc.

13.3.4 Interfaces

The model for predicting doses to the four generic target organisms interfaces smoothly with the surface water (Chapter 5), soil (Chapter 6) and atmosphere (Chapter 7) submodels of BIOTRAC (Figure 13-3). It also interfaces with CALDOS (Chapter 8) and uses the same parameters to calculate nuclide transfer through the food chain. Hence the interfaces outlined in Figure 13-3 are similar to those depicted in Figure 8-3. The model requires concentrations of nuclides in well water, surface water, soil, sediment and air as inputs. It also requires rates for aerial deposition and irrigation. Because the model for non-human biota and CALDOS are very similar, the linkages and the source of input parameters discussed in Section 9.4 for CALDOS are also relevant here.

Doses are calculated for ^{14}C, ^{129}I and ^{99}Tc, hence processes involving only these nuclides need to be included. Radiological doses to the four generic

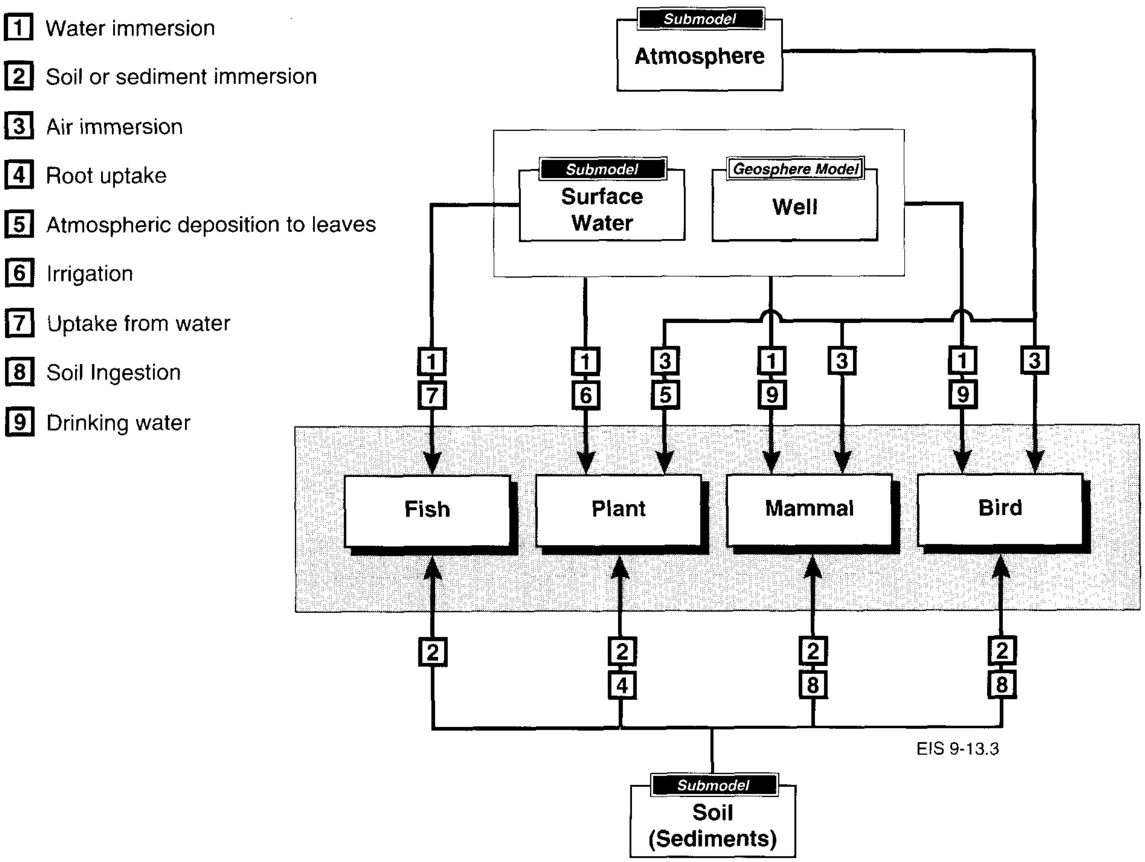


FIGURE 13-3: Interfaces Between the Radiological Dose Model for Non-Human Organisms (Shaded) and the Four Submodels of BIOTRAC. The numbered pathways relate to transfer mechanisms and exposure situations.

target organisms are the primary output of the model to help assure protection of the environment in the postclosure phase (Goodwin et al. 1994).

13.3.5 Parameters

13.3.5.1 Parameters Documented Elsewhere

Most parameters in the model are defined elsewhere. The values and PDFs of most of the parameters are documented in Chapters 5 to 8. The only exceptions are a time parameter, te_i, described in Section 13.3.5.2, and the DCFs, discussed in Section 13.3.5.3.

13.3.5.2 Time of Exposure for Terrestrial Food, te_j (d)

In CALDOS, the time of exposure for terrestrial food types, te_j, was selected to reflect an agricultural situation where humans harvest an annual crop (Section 8.5.3.2). The values selected are conservative for that situation. However, they would not be necessarily conservative for the ingestion of perennial plants by wild animals. For example, caribou may feed on lichens, which are long-lived and could be exposed to contaminants for many years. This situation is also recognized in the assessment of the impact of glaciation (Chapter 12), where te_j was set to 50 a or 18 250 d (Section 12.3.2.5). In Equations (13.1), (13.2) and (13.15), te_j is also set to 18 250 d for j = TE PLANT, TE MEAT and TE BIRD. This reflects the potentially long exposure time of vegetation eaten by wild mammals and birds, and also assumes that a plant can be continually irradiated from internal nuclides for 50 a.

13.3.5.3 Dose Conversion Factors

DCFs are used to calculate radiological doses for the four target organisms from nuclide concentrations. The nuclides can be either internal or external to the organisms. The DCFs were established by Amiro (1992c) for each of the target organisms. Internal DCFs are used in Equations (13.1) to (13.3); external DCFs are used in Equations (13.4) to (13.15).

Only three nuclides are considered and separate DCFs are required for each of them. The DCFs include doses resulting from both photon (γ-radiation) and electron (β-radiation) emissions. There are no α-particle emissions from ^{14}C, ^{129}I or ^{99}Tc, but they could be considered in a similar way as the other types of radiations. However, a radiation quality factor, Q, would be required for α-radiation to account for its greater potential for doing harm. The DCF values are listed in Table 13-4.

Internal Dose Conversion Factors $(DFB^i_P)_I$, $(DFB^i_M)_I$, $(DFB^i_B)_I$, $(DFB^i_F)_I$
$((Gy\cdot a^{-1})/(Bq\cdot kg^{-1}$ wet biomass))

The DCFs for internal exposure are conservatively high; they assume that all the emitted radiation energy is absorbed by the organism (Amiro 1992c). This allows the DCFs to be used for organisms of any size, although they predict conservatively high doses for small organisms that may only absorb a small part of the energy. This assumption is reasonable for nuclides such as ^{14}C and ^{99}Tc that emit only electron radiation. In the case of ^{14}C, the value given in Table 13-4 is identical to the value for humans, $DF^{c'}$, based on specific activity considerations (Section 8.5.2.1). This is because in both cases all the energy emitted is absorbed. For ^{129}I, the DCFs for non-human biota are higher than the corresponding value, DF^I, for humans (Section 8.5.2.1) because of more conservative assumptions.

For nuclides that emit photons, our DCFs are slightly conservative, probably by less than an order of magnitude, depending on the size of the organism. Of the three nuclides considered, this only applies for ^{129}I.

The DCFs for internal exposure of animals to ^{129}I, $(DFB^I_F)_I$, $(DFB^I_M)_I$, and $(DFB^I_B)_I$, are greater than $(DFB^I_P)_I$ to account for the concentration of

TABLE 13-4

DOSE CONVERSION FACTORS FOR NON-HUMAN GENERIC TARGET ORGANISMS

Parameter	Nuclide		
	^{14}C	^{99}Tc	^{129}I
$(DFB^i_P)_I$, $(DFB^i_M)_I$, $(DFB^i_B)_I$, $(DFB^i_F)_I$			
Internal Exposure			
Plants	2.5×10^{-7}	5.11×10^{-7}	4.46×10^{-7}
Terrestrial Animals and Fish	2.5×10^{-7}	5.11×10^{-7}	4.46×10^{-6}
$(Gy \cdot a^{-1})/(Bq \cdot kg^{-1}$ wet biomass)			
$(DFB^i_P)_W$, $(DFB^i_M)_W$, $(DFB^i_B)_W$, $(DFB^I_F)_W$			
Water Immersion			
Terrestrial Organisms	6.51×10^{-12}	8.62×10^{-11}	1.19×10^{-10}
Fish	1.38×10^{-10}	2.43×10^{-10}	2.70×10^{-10}
$(Gy \cdot a^{-1})/(Bq \cdot m^{-3}$ water)			
$(DFB^i_P)_A$, $(DFB^i_M)_A$, $(DFB^i_B)_A$			
Air Immersion	6.01×10^{-9}	7.96×10^{-8}	5.73×10^{-8}
$(Gy \cdot a^{-1})/(Bq \cdot m^{-3}$ air)			
$(DFB^i_F)_S$, $(DFB^i_P)_S$, $(DFB^i_M)_S$, $(DFB^i_B)_S$			
Soil/Sediment Immersion	9.77×10^{-9}	1.29×10^{-7}	1.79×10^{-7}
$(Gy \cdot a^{-1})/(Bq \cdot kg^{-1}$ dry)			
$(DFB^i_M)_V$, $(DFB^i_B)_V$			
Vegetation Immersion	6.01×10^{-9}	7.96×10^{-8}	5.73×10^{-8}
$(Gy \cdot a^{-1})/(Bq \cdot kg^{-1}$ wet biomass)			

Note: I = internal exposure; A = air, S = soil or sediment, V = vegetation and W = water immersion; and B = bird, F = fish, M = mammal and P = plant target organisms.

iodine in the thyroid gland (Amiro 1992c). This compensates for not considering the thyroid explicitly in the transfer pathways in Equations (13.2) and (13.3). The resulting calculated doses therefore evaluate the dose received by the thyroid. This is the only specific organ dose calculated for non-human organisms; all the other doses are calculated for the whole body. For humans, the thyroid gland is also of sole importance for ^{129}I (Section 8.5.2.1, Zach and Sheppard 1992).

External Dose Conversion Factors for Water Immersion, $(DFB^i_P)_W$, $(DFB^i_M)_W$, $(DFB^i_B)_W$, $(DFB^i_F)_W$
$((Gy \cdot a^{-1})/(Bq \cdot m^{-3}$ water))

The DCFs for water immersion of plants and terrestrial animals are based on Holford (1989) (Amiro 1992c). Here, we assume that the organism is immersed in a semi-infinite, uniformly contaminated medium, with the body centroid located at a depth of 0.1 m below the water surface. We estimated radiological doses from electron emissions at 70 μm below the epidermal surface; internal parts of the organism will receive a lesser dose. Photon emissions are not as easily attenuated, and we use the calculations of Holford (1989) at the epidermal surface. The resulting DCFs (Table 13-4) are greater than those used for humans (Table 8-4). This is largely because the human DCFs correspond to the whole body dose (effective dose equivalent (Section 1.2.3)), whereas the DCFs for non-human biota correspond to the dose at the body surface.

The external dose conversion factor for fish, $(DFB_F^i)_W$, is based on values reported by NRCC (1983) as explained by Amiro (1992c).

External Dose Conversion Factors for Air Immersion, $(DFB_P^i)_A$, $(DFB_M^i)_A$, $(DFB_B^i)_A$

$((Gy \cdot a^{-1})/(Bq \cdot m^{-3}$ air))

The DCFs for air immersion are also based on Holford (1989) (Amiro 1992c). Here, we assume that the organism is immersed in a semi-infinite, uniformly contaminated medium, with the body centroid located 1 m above a ground surface. Radiological doses from electron emissions are estimated at 70 μm below the epidermal surface; internal parts of the organism will receive a lesser dose. Photon emissions are not as easily attenuated, and we use the calculations of Holford (1989) at the epidermal surface. The resulting DCFs (Table 13-4) are greater than those used for humans (Table 8-4), as explained for the DCFs for water immersion.

External Dose Conversion Factors for Soil and Sediment Immersion, $(DFB_P^i)_S$, $(DFB_M^i)_S$, $(DFB_B^i)_S$, $(DFB_F^i)_S$

$((Gy \cdot a^{-1})/(Bq \cdot kg^{-1}$ dry soil or sediment))

Fish are assumed to be immersed in sediment, and all the other organisms are assumed to be immersed in either soil or sediment used as soil (Section 6.3.7.3). To establish these DCFs, Amiro (1992c) used several conservative assumptions and based the soil and sediment immersion DCFs on the water immersion geometry used by Holford (1989). The resulting DCFs include a modification for soil bulk density, and therefore have different values and units than those for water immersion (Table 13-4).

External Dose Conversion Factors for Immersion in Vegetation, $(DFB_M^i)_V$, $(DFB_B^i)_V$

$((Gy \cdot a^{-1}/(Bq \cdot kg^{-1}$ wet biomass))

Amiro (1992c) used several conservative assumptions and based the DCFs for immersion in vegetation on the air immersion geometry used by Holford (1989). The resulting DCF values for vegetation and air immersion are identical (Table 13-4), even with different units. This is because we assumed a total vegetation density of 1 kg wet biomass$\cdot m^{-3}$ air on the basis of a plant yield of 1 kg wet biomass$\cdot m^{-2}$ soil (Section 8.5.8) and a plant height of 1 m (Amiro 1992c).

13.3.6 Model Validation

Our model for calculating doses to the four generic non-human biota is analogous to CALDOS and uses similar concepts and transfer functions. Therefore, to a large extent, it has received validation similar to CALDOS (Section 8.6). However, the use of this type of assessment model for non-human biota has not been validated because little work has been done in this general area.

13.3.7 Discussion

13.3.7.1 Assumptions

A number of assumptions were made in deriving the model to estimate radiological doses to generic non-human target organisms. In this section, we restate, evaluate and review the main assumptions and discuss their effects on the predictions of the model.

1. *The transfer of nuclides into and through the food chain is assumed to be linear.* This assumption is analogous to that made for calculating nuclide transfer to humans, as discussed in Section 8.7.1.

2. *The complex processes responsible for nuclide transfer into and through the food chain can be described using simple transfer coefficients.* Again, the reasoning here is the same as that for human food-chain transfer (Section 8.7.1).

3. *Steady-state conditions are assumed.* This is reasonable for the relatively short time scales that affect nuclide transfer through the food chain to various organisms. Further discussion of this assumption is given in Section 8.7.1.

4. *Recycling of nuclides is accounted for implicitly.* The discussion in Section 8.7.1 for human food chains is also relevant here.

5. *Radiological doses are calculated for four generic target organisms.* These organisms represent a broad range of biota receiving external exposures from contaminated water, soil (or sediment), air and vegetation. The calculations assume that an organism is immersed in each of the relevant media, and the total external dose is obtained by summing over media and nuclides. This conservatively allows plants and animals to spend all of their time in several habitats simultaneously so that diverse natural histories are included. Internal exposure of the target organisms is more specifically related to certain groups of plants and animals. The transfer coefficients used to estimate nuclide transfer through the food chain were largely developed for domestic plants and animals. However, they include all the wild animals with similar feeding habits and all the native plants with similar nuclide uptake characteristics. The fish represents a broad range of freshwater species, and the mammal and bird represent most terrestrial animals that are primary consumers. Zach and Sheppard (1992) discuss the scope of our target organisms in relation to some of the model parameters in more detail.

6. *Only three nuclides are used in the dose calculations.* These three nuclides, ^{14}C, ^{129}I and ^{99}Tc, have been shown to contribute almost all of the dose to humans (Goodwin et al. 1994) and for this reason were also used in the sensitivity analysis (Chapter 10) and the glaciation assessment (Chapter 12). The impact of these three nuclides is related primarily to the relative flux from the geosphere so they should also dominate the radiological dose to non-human organisms. Neglecting the other nuclides will not significantly underestimate the doses to our target organisms.

7. *Organisms in higher trophic levels such as secondary or tertiary consumers are not modelled.* Humans, the only secondary consumer modelled explicitly by us (Chapter 8), can be used as good indicator species representing a broad range of large mammalian secondary consumers, such as bear and wolf. Trophic levels beyond the primary consumer are not modelled explicitly for non-human biota. However, bioconcentration up the food chain has not been shown to be an important mechanism for the three nuclides considered, or for most other nuclides (Zach and Sheppard 1992). Furthermore, the broad PDFs used for all of our transfer coefficients (Section 8.5.1) account for a wide range of transfer mechanisms. This allows consumers at high levels in the food chain to be included either because there is no bioconcentration or because the amount of bioconcentration is encompassed within our parameter distribution. It is likely that animals such as owls, osprey, raven, wolf, mink, and a broad range of invertebrates are all included.

8. *Complex ecosystem interactions and other ecological interrelationships are not modelled.* The model does not account for ecological relationships, nor does it consider the impact of ionizing radiation on the ecosystem. The model is unidirectional because we assume that the biosphere is in a steady state and that nuclides are transferred through the food chain without feedback from potential effects. This assumption is valid provided radiological doses are too low to cause any environmental changes. If the doses to biota are large enough to cause an impact, then the ecosystem might be altered. Given the stringent radiological dose criteria for humans (AECB 1987), substantial ecosystem impacts are very unlikely.

9. *The transfer coefficients apply mostly to domestic plants and animals, but the broad range also includes wild plants and animals.* The nuclide uptake characteristics for most domestic plants are similar to many native plant species (Zach and Sheppard 1992). In most cases, annual agricultural plants have greater transfer coefficients than perennial native species. Thus, our model is conservative in that most native species such as trees will have transfer coefficients in the lower range of the distributions used in our model. All of our animal transfer coefficients have wide PDFs, and thus include a wide variety of domestic and wild animals. The water, food and soil ingestion rates used for mammals and birds are based mostly on cattle and chickens respectively (Section 8.5.5). These are also broadly distributed parameters in our model and should include many wild species. Zach and

Sheppard (1992) discuss the broad applicability of our transfer coefficients and ingestion rates in detail.

10. *The model is comprehensive but it does not explicitly treat all possible exposure pathways.* A similar assumption is also made for CALDOS (Section 8.7.1). We have ignored inhalation by terrestrial animals, assuming that it is relatively unimportant for ^{14}C, ^{129}I and ^{99}Tc. This pathway might be important for some nuclides such as radon and its daughters in the case of burrowing animals. However, the nuclide flux to the biosphere is totally dominated by the three nuclides that we have selected, so the total dose is not underestimated when other nuclides are not considered. Other minor pathways are either considered implicitly, or else our conservative assumptions likely compensate for their exclusion.

13.3.7.2 Evaluation

Our model for calculating radiological doses to generic non-human target organisms can be used to give a quantitative measure of the predicted impacts of the disposal concept on plants and animals. The model employs methods similar to those used for estimating radiological doses to humans (Chapter 8). As an assessment model, it simplifies many complex processes and relies on distributed parameter values to include a wide range of pathways and organisms.

The model was designed specifically for the postclosure assessment and therefore has some unique features. For example, doses are calculated for only ^{14}C, ^{129}I and ^{99}Tc. They are the three most important nuclides that are predicted to reach the biosphere from the underground vault (Goodwin et al. 1994). If the source term in the vault or the transport mechanisms in the geosphere were to change, other nuclides might become more important.

The model evaluates radiological dose to four target organisms. These organisms are generic, representing different groups of biota: plants, mammals, birds and fish. The external exposure component of the model applies to all plants, animals and microorganisms because it was developed using conservative methods for large organisms. These methods overestimate doses to smaller organisms such as bacteria and insects. The internal exposure component of the model applies to a more narrow range of organisms, but still reflects many species of fish, plants and terrestrial animals.

Ecological relationships are not modelled explicitly, although nuclide transfer in the food chain is modelled. The effects of radiation on ecological relationships would only have to be assessed if the dose estimates are sufficiently great to cause adverse effects on individual organisms. For example, if the model predicts doses of the order of several $Gy \cdot a^{-1}$, then we would expect such detrimental effects (Myers 1989). The ecological impact, including population dynamics, interrelationships among plant and animal communities and energy flow, would need to be assessed using a different modelling approach. On the other hand, if dose predictions are low, such that no effects on individuals are likely, then a significant impact at higher organizational levels would be unlikely. This means that populations, communities and ecosystems would be implicitly protected (Figure 13-1).

We conclude that the model provides a quantitative measure of radiological doses that could be experienced by a variety of non-human organisms. These doses can be compared with known effects of ionizing radiation on non-human biota to ensure protection of the environment.

13.4 CALCULATION OF DOSES TO SPECIFIC TARGET SPECIES

Our methodology for evaluating and assuring environmental protection (Section 13.1.1) is very comprehensive and includes generic target organisms. However, if predicted doses to these organisms are relatively high, it may become desirable to calculate radiological doses to specific target species in addition to doses to generic target organisms (Section 13.3). Such calculations may also be of interest when assessing specific sites during siting of a potential nuclear fuel waste disposal facility.

The objective here is to show how such calculations could be carried out, and to identify the data and models needed to do so, particularly for food-chain transfer. We have selected a terrestrial example with the wolf, and an aquatic example with a top predator, such as the osprey, common loon or otter, as the ultimate target species. These top predators can be considered flagship species the protection of which will also assure the protection of other species upon which they depend, and the environment in general.

The terrestrial example is based on the work by Zach et al. (1989) on the transfer of fallout ^{137}Cs in boreal food chains. The aquatic example is based on environmental studies of uranium mining in northern Saskatchewan and on pollutant studies in the Great Lakes. These examples were selected because of the availability of transfer data. The radionuclides involved are not important in the postclosure assessment (Goodwin et al. 1994); however, they do help to demonstrate the methodology needed for calculating doses to specific target species in a realistic manner using the proper ecological setting.

Given radionuclide concentrations for target species, radiological doses can be calculated with the help of DCFs (Section 13.3.3). The doses can then be evaluated in the same manner as those for our four generic target organisms as discussed in Section 13.1.4.

13.4.1 Terrestrial Boreal Food Chain - Wolf

Figure 13-4 summarizes a boreal terrestrial food chain and includes the relevant parameter values needed for calculating doses to primary producers (plants), primary consumers (moose, white-tailed deer and beaver) and a secondary consumer (wolf). The plant/soil concentration ratio, Bv^i (Section 8.5.1.1), is based on several soil types and plant species, and it can be represented by a lognormal PDF. The same is true for all the terrestrial animal transfer coefficients, F_j^i (Section 8.5.1.2). We have pooled various soil types and plant species here for simplicity. Average values are given for the feed or forage ingestion rates, Qf_j (Section 8.5.1.2), for these animals. No data are given for the relative portion of

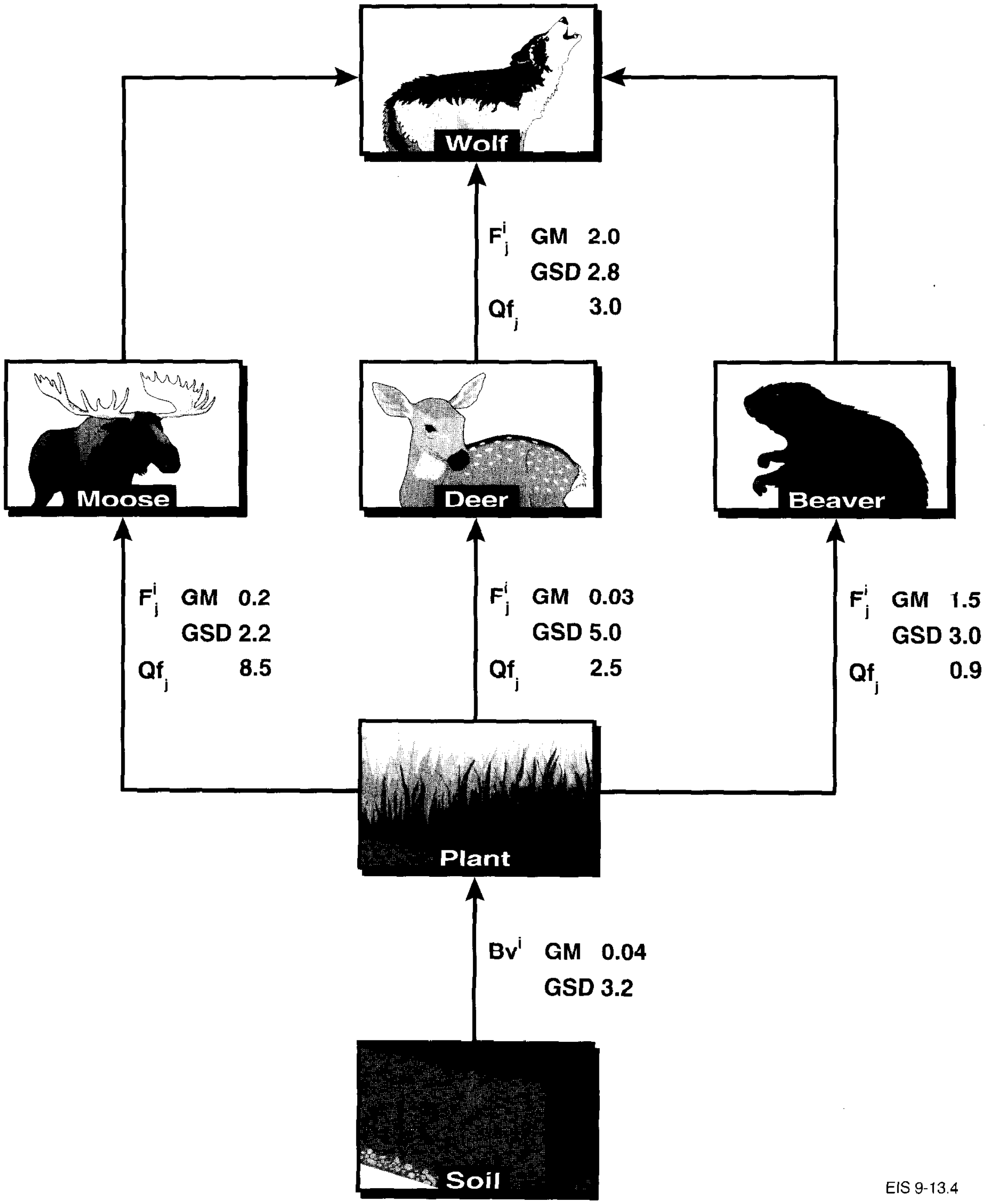


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FIGURE 13-4: Terrestrial Food-Chain Transfer of ^{137}Cs from Soil to Wolf. Soil concentrations can be predicted by BIOTRAC and the parameter values for Bv^i ($Bq \cdot kg^{-1}$ wet biomass/$Bq \cdot kg^{-1}$ dry soil), F_j^i ($d \cdot kg^{-1}$ wet biomass) and Qf_j ($kg \cdot$wet biomass$\cdot d^{-1}$) can be used to calculate concentrations in biota for predicting radiological doses.

moose, deer and beaver in the diet of wolves. This would have to be determined on the basis of a specific site.

Figure 13-4 includes many simplifications. We have not shown the different soil types and plant species involved in the study and for which data are available (Zach et al. 1989). Again such details could be considered explicitly in a site-specific application. Also not shown are transfers that may originate from water and air. In Figure 13-5 we show a more complete picture, but without parameter values. Such values are largely unavailable and the missing information would have to be determined through further studies or covered through conservative assumptions (Section 2.7). Note, however, that BIOTRAC calculates integrated water, soil and air concentrations (Chapters 5 to 7).

Concentrations of ^{137}Cs were relatively low in all the organisms studied by Zach et al. (1989). There was some biomagnification at the level of the wolf, with tissue concentrations up to 6.5 times higher than in the prey. The potential effects of this were not investigated and none would be expected at the prevailing low radiation levels.

13.4.2 Aquatic Food Chain - Osprey

Aquatic food chain models that can be used to evaluate environmental protection have been published by Thomann (1981), Swanson (1983, 1985) and Thomann and Connolly (1984). Compartment models usually require contaminant concentrations in water and sediment to predict concentrations in biota including algae, invertebrates, forage fish and predatory fish.

Water and sediment concentrations can be calculated by the surface water model of BIOTRAC (Chapter 5). The models of Thomann (1981) and Swanson (1985) are well adapted to the highly specific food chain on the Canadian Shield. Swanson's data describe the distribution of ^{210}Pb, ^{226}Ra and ^{TOTAL}U in the biota, water and sediments of lakes in the Beaverlodge Lake area of northern Saskatchewan, whereas Thomann (1981) models the transfer of ^{137}Cs, ^{239}Pu and PCB in Great Lakes fish.

We added a final compartment to the structure of Thomann's (1981) model to represent the highest consumer in the food chain. This compartment would represent the osprey, common loon or otter receiving radionuclides from the ingestion of fish, water or sediment. The general structure of the food webs modelled is given in Figure 13-6. Individual compartments can represent generic organisms (e.g., forage fish) or specific organisms (e.g., fathead minnow), with the selection of appropriate data values and uncertainties.

Waite et al. (1988, 1989, 1990) and Joshi et al. (1989) document the movement of several radionuclides from mine tailings in a bay of Lake Athabasca, and evaluate concentrations in biota, doses and toxicological effects. Despite relatively high dose rates to kidney and gonad caused by high radionuclide concentrations in bone and gut, no significant toxicological or pathological effects were identified in pike or whitefish from the contaminated bay when compared with fish from a reference site (Waite et al. 1990). Swanson (1983, 1985) used the data on radionuclide concentrations in biota in the Beaverlodge Lake area to estimate radiological doses to the

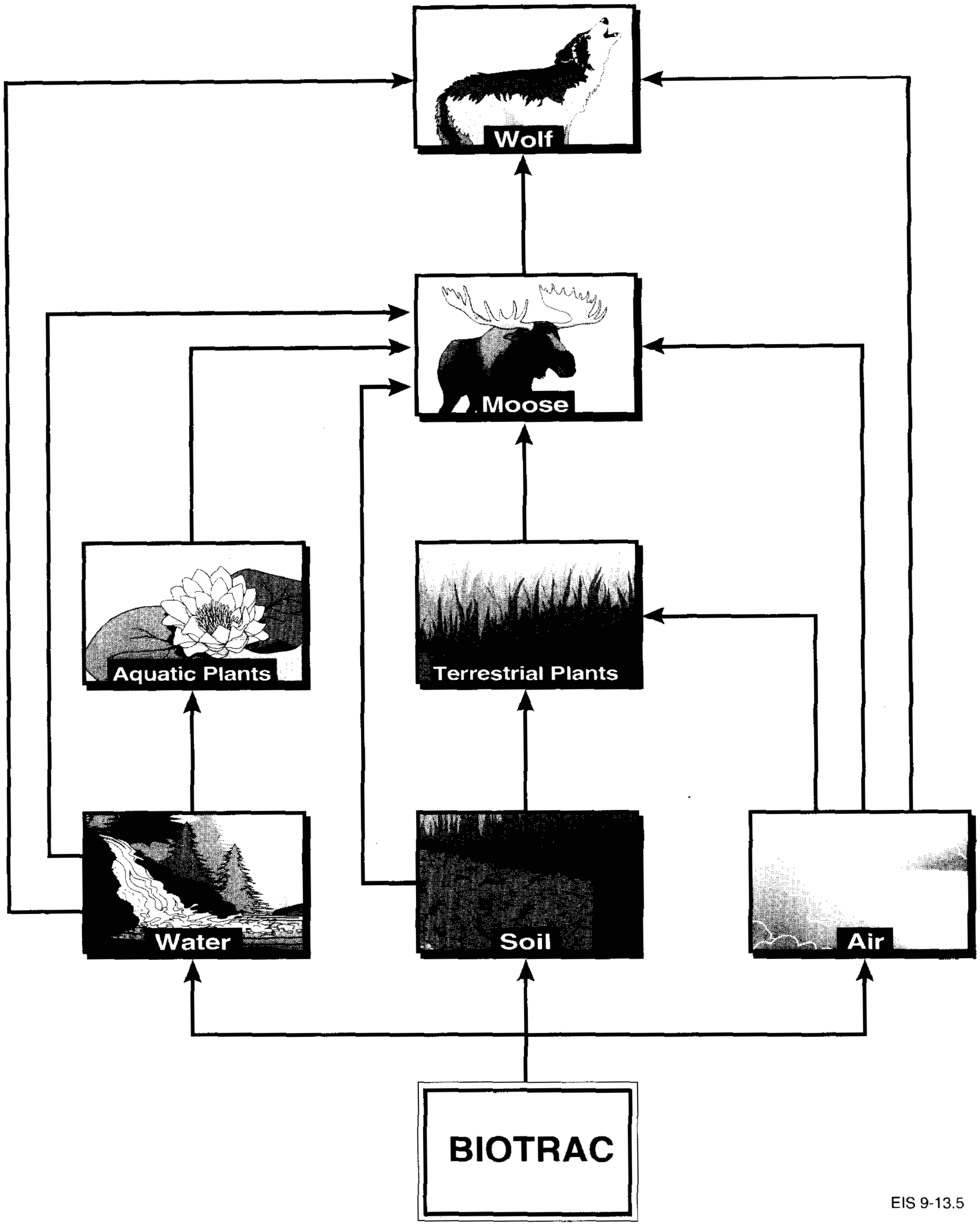

FIGURE 13-5: Transfer Pathways from Water, Soil and Air to Plants, Moose and Wolf

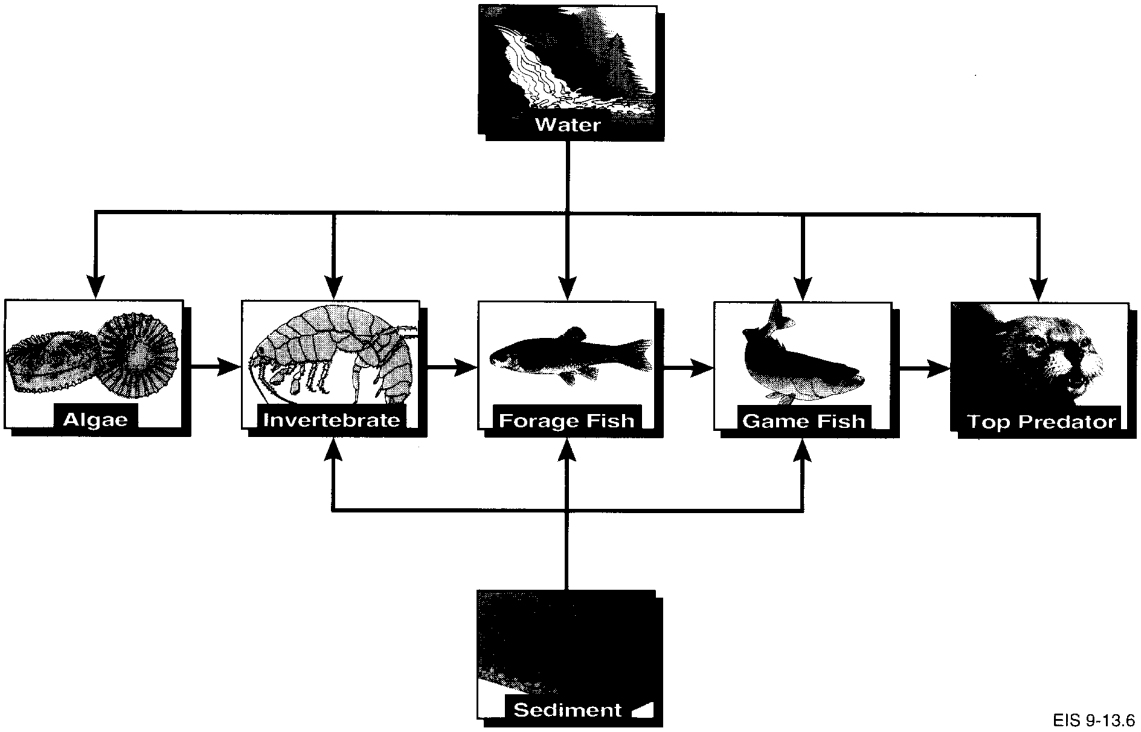

FIGURE 13-6: Transfer Pathways from Water and Sediment to Algae, Invertebrates, Fish and Top Predators

upper trophic levels caused by the accumulation of ^{210}Pb, ^{226}Ra and ^{TOTAL}U. Although Swanson's (1985) study does not provide explicit estimates of the rate of transfer of nuclides through the food chain, Thomann's (1981) paper provides data for ^{137}Cs and ^{234}Pu, and shows how these data can be used for other nuclides. Together, these studies provide good examples of the rigorous approach necessary to estimate radionuclide concentrations and dose to specific aquatic organisms, and to organisms feeding wholly or partly in aquatic ecosystems.

13.4.3 Conclusions

We have shown how radiological doses to specific target species can be calculated in a realistic manner. Such calculations require a large amount of data, particularly on food-chain transfer, and there are gaps in the data available. These would have to be bridged by selecting conservative, yet reasonable parameter values. It would be impossible to collect data for protected species such as the osprey and common loon. For these reasons, we rely on the three approaches outlined in Section 13.3 for the postclosure assessment. These approaches are well supported by data and, taken together, address environmental protection in an integrated and holistic way.

13.5 SUMMARY AND CONCLUSIONS

Protection of the environment is one of the central goals for the safe disposal of nuclear fuel wastes. In this chapter, we define three quantitative end points for evaluation of potential impacts of the waste nuclides entering the biosphere from an underground vault. These include the evaluation of human radiological doses and the use of humans as an indicator species for other biota, the prediction of nuclide concentrations in various environmental compartments, and the prediction of radiological doses to several generic target organisms. These end points are derived rigorously and form a good basis for evaluating potential impacts on the environment to ensure its protection. The end points are also suitable for monitoring purposes, particularly nuclide concentrations in surface water, soil and air.

14. SITE-SPECIFIC APPLICATION

Concept assessment is not concerned with selecting a site for a disposal facility and so there is no specific site. However, it is necessary for us to develop and demonstrate a suitable assessment methodology and establish that technically suitable disposal sites likely exist in Canada (Federal Environmental Assessment Review Panel 1992, AECL 1994a). Potential sites would be identified through a screening and detailed evaluation procedure (Davison et al. 1994a) with full public participation (Greber et al. 1994). Site evaluation would involve detailed assessments using a methodology such as SYVAC. The present generic version of BIOTRAC would be modified for such assessments to make it correspond to specific sites. In this chapter, we describe the changes required to make the model and its parameter values site-specific. In all cases, the changes are relatively minor and are readily achievable. We discuss the need to consider additional pathways and processes and different modelling approaches in a site-specific assessment. We identify the parameters that would have to take on site-specific values, and show how such values can be obtained. Finally, we describe how a site-specific model would provide the opportunity for an improved treatment of some transport processes.

Even at a specific site, the state of the biosphere will likely change over the next 10 000 a or more (Chapter 3), and temporal variations must still be considered in a site-specific model. Given the location and nature of a specific site, new pathways or modelling approaches might be used to allow for the possibility of environmental change. Means of PDFs for parameters that change slowly with time could reflect present-day conditions at a specific site, but means for more rapidly varying parameters would have to be based on expected values over the course of time. The variability of site-specific parameters would not include spatial variability over the entire Canadian Shield area in Ontario, but would have to reflect the uncertainty associated with environmental change, and so would be similar to our generic values. The long time frame of the assessment would therefore mean that the differences between generic and site-specific versions of BIOTRAC would be limited.

14.1 GEOSPHERE/BIOSPHERE INTERFACE

The geosphere model used for the postclosure assessment is based on a specific location, the WRA (Section 4.1). Accordingly, the techniques already developed to study and model subsurface structure and hydrogeology are available for application to a potential disposal facility site (Davison et al. 1994a). The information that is presently passed from the geosphere to the biosphere model (Section 4.4) would be available in a site-specific assessment. This would include the identification of the water body into which the nuclides would emerge, the locations and areas of localized discharge zones within the water body, and predicted water and nuclide flows out of the geosphere. This information would be obtained using a variety of geotechnical and modelling techniques to investigate and document the groundwater flow system (Davison et al. 1994a) and surface evidence of deep groundwater discharge (Stephenson et al. 1992). Investigation of the stratigraphy of the area through coring and sonar studies would reveal the nature and depth of overburden and sediment layers beneath and adjacent to the water body. Because both the geosphere and biosphere models would treat the same location, the hydrological parameters that they hold in common would be consistent.

The approach to modelling the geosphere/sediment interface described in Section 4.4.1 would likely not change in a site-specific assessment. The geosphere/surface water interface (Section 4.4.2) would also not change, although given a specific location it would be possible to determine which discharges contribute to the nuclide load in the water body accessed by the critical group and other biota. Information on local surface hydrology may make it possible to establish whether or not discharge would occur to terrestrial areas (Section 4.4.3), and the size of these areas. It would then be unnecessary to assume that an arbitrary fraction of the discharge emerges beneath terrestrial areas (Section 4.5.1). Moreover, it may be possible through site-specific studies to determine the amount of water moving laterally and downward through areas of terrestrial discharge. This amount could be added to the water volume used to dilute the nuclide flux (Equation 4.8) to produce a more realistic estimate of pore-water concentrations to drive the soil model. The bedrock well model developed for our assessment (Section 4.4.4) could be augmented by an overburden well if such wells were common at a specific site.

The parameters appearing in the interface models may take on values at a specific location that are different from those observed at the WRA. The subsurface hydrology of the site would determine the location and depth of domestic wells. The probability that a well would serve as the water source would be determined by the hydrology, the availability of other water sources, water quality and existing practices of water use in the area. The stable iodine and carbon concentrations in near-surface groundwater that would be used in the dosimetry models for ^{14}C and ^{129}I (Sections 8.3.1.11 and 8.3.1.12) would be determined using standard methods. The remaining parameters are discussed in the geosphere model report (Davison et al. 1994b). They will not be considered further here, except to note that they could all be obtained at a specific location using standard techniques.

14.2 SURFACE WATER SUBMODEL

As noted in the previous section, the discharge water body could be determined at a specific site, as could the location of localized discharge zones within it. If the nuclides emerge into a lake (Section 4.4.2), the model described in Chapter 5 would be used in the assessment. If the discharge were to a wetland, predictions would be made using an appropriate wetland model, perhaps a combination of a lake and a soil model. If the discharge occurred to a river or stream, an appropriate model would be developed, or the nearest downstream lake would be modelled, for the reasons given in Section 5.1. In the case of multiple discharge locations, each location would be evaluated separately, and the assessment could be focussed on the location for which health and environmental impacts were the largest.

Given the topography, climate and soil conditions at a specific location, it might be possible to model the nuclide input to the lake through runoff from soil in a less conservative way than is done now (Figure 4.8). Similarly, information on circulation patterns in the lake could lead to a more realistic description of nuclide mixing. The assumption of uniform, instantaneous mixing could then be dropped, and water concentrations calculated as a function of position in the lake. Similar factors could be considered if nuclides were to discharge to a wetland, river or stream. Given a specific surface water body, it might also be possible to address lake evolution in a more realistic manner (Section 14.7).

The parameter values and PDFs derived here for BIOTRAC (Section 5.5) would not necessarily apply to a specific water body, but site-specific values could be readily defined. Once the discharge water body has been identified, the catchment area, lake area, and lake mean depth could be easily measured. Information on runoff could be obtained from historical records, which are available for many locations (Environment Canada 1985), or measured directly using routine monitoring methods. The sediment accumulation rate and the thickness of mixed sediment would be determined by examining individual sediment cores (Appleby and Oldfield 1978, Robbins 1978). Appropriate literature values for the water/sediment transfer rates of nuclides could be chosen, based on the observed trophic status and water chemistry of the lake. These values could be further refined for key nuclides through site-specific experimental work, using methods described by Lerman (1979), Hesslein et al. (1980) and Cornett and Ophel (1986). Site-specific studies could also lead to adjustments in the ^{14}C gaseous evasion rate, and provide the data to define evasion rates for other gaseous nuclides, such as ^{129}I.

14.3 SOIL SUBMODEL

The mechanistic model SCEMR1 would continue to serve as the basis for calculating soil concentrations at a specific location (Section 6.3.1). SCEMR1 would be run using soil and climate data characteristic of the site and a new set of regression equations developed for incorporation into SYVAC (Section 6.3.2). Given the topography, climate and soil conditions of the site, it might be possible to include processes such as soil erosion and runoff in the model, and to calculate the associated lateral movement of nuclides. Similarly, this information might allow more explicit modelling of recycling processes, including root uptake and the subsequent return of nuclides to the soil through vegetative decay (Section 6.5.5.1). The nature or location of the discharge zones may make it unnecessary to treat the situation in which nuclides reach the soil from below with contaminated groundwater (Section 14.1). Similarly, soil and climate conditions will determine the frequency with which the garden and forage field are irrigated, as well as the duration of irrigation.

In order to run SCEMR1 for a specific location, a variety of soil and climate data would have to be gathered (Section 6.5). Soil depths would be established by measuring water table levels. Soil types in the area would be surveyed, and the major soil series identified. Parameters such as dry bulk density, field moisture capacity, saturated hydraulic conductivity and cation exchange capacity, as well as the soil moisture characteristic curve, would be determined for each major series. The sorption and degassing behaviour of the various nuclides would also be established for each series. Some of this information could be extracted from the literature once the soil type was known; the rest could be obtained experimentally using routine methods (Buckman and Brady 1969, Thibault et al. 1990). Values for gaseous evasion rates could be obtained experimentally by measuring the nuclide flux from the soil to the atmosphere, or by using a mass balance approach (Sheppard M.I. et al. 1991). As much of the experimental work as possible would be done in the field. Where field tests are not practical, soil cores would be taken for analysis in the laboratory.

The meteorological data required to run SCEMR1 include daily values of precipitation, solar radiation, air temperature, wind speed and vapour pressure for a year that represents the long-term average climate of the site. Long-term records of wind speed, precipitation and temperature are available at many locations on the Canadian Shield (Environment Canada 1982a, 1982b). Solar radiation and vapour pressure are measured less commonly, but some information is available. The required meteorological inputs for a specific location may therefore exist in the literature. If the historical data are in any way insufficient, the instrumentation and methodologies exist to allow site-specific values of all the climatological parameters to be determined by direct observation. This would involve continuous recording over a number of years to determine representative values for the site.

The majority of the parameters used in SCEMR1 are not distributed and would be assigned fixed values representative of the site. Only the effective precipitation, soil solid/liquid partition coefficient, soil depth, and gaseous evasion rate appear as distributed parameters in the response function formulation of the soil model (Section 6.3.3). PDFs for each of these parameters could be defined through a measurement program to reflect local conditions.

14.4 ATMOSPHERE SUBMODEL

The atmosphere submodel described in Chapter 7 would remain essentially unchanged in a site-specific application. The availability of site-specific data could lead to improvements in two areas. First, the dispersion relations (Equations 7.4 and 7.5) for ground-level area sources could be modified to reflect local conditions of wind speed, atmospheric stability and topography. Secondly, the sizes and relative locations of the garden, forage field, woodlot and lake may be well enough known at a specific location to allow the lateral transfer of airborne nuclides between fields to be calculated. Atmospheric transport from more to less contaminated areas could be modelled in this way.

Site-specific values and PDFs for several model parameters (Section 7.5) would be established by direct observation. Atmospheric dust loads from both terrestrial and aquatic sources could be measured. A site-specific value for the aquatic iodine mass loading parameter could be obtained by measuring natural iodine concentrations in the local water body and in the air above it. Local values for the radon transfer coefficients from surface waters to air, and from soil to indoor air, could be determined (Emerson et al. 1973, George and Breslin 1980). Once the soil characteristics are known, a site-specific value for the radon emanation rate from soils could be calculated. Gaseous evasion rates for the volatile nuclides (Section 2.5) from soil and surface water could be determined using the methods indicated in the previous two sections. Characterization of the local forest ecosystem would lead to site-specific values for forest yield and the energy content of the wood. The soil survey mentioned in the previous section would define the amount of peat available for burning. Site-specific information on wind speeds and atmospheric stability, for use in defining the dispersion relations, would be available from historical data (Environment Canada 1982a, 1982b) or from direct measurements.

Site-specific values and PDFs for most of the other atmosphere model parameters would be similar to those specified for the generic assessment (Section 7.5). Parameters such as building size, infiltration rate and the frequency of agricultural fires are related to culture, and show little variation from place to place on the Canadian Shield. Average values for the dry deposition velocity also show little spatial variability. The large variability in this bulk parameter represents uncertainty, rather than spatial variability (Section 7.5.4.1).

14.5 FOOD-CHAIN AND DOSE SUBMODEL

The food-chain and dose submodel, CALDOS, includes all the exposure pathways that could result in an appreciable dose to humans. No additional pathways would likely have to be considered in a site-specific application, and no changes to the modelling approach would likely be needed.

The parameter values used in CALDOS (Section 8.5) reflect temperate conditions and the type of agriculture and animal husbandry practised on the more southern portions of the Canadian Shield in Ontario. Given a specific location, the parameter values may have to be modified to reflect local conditions. The location, climate and soil conditions of the site would determine the type of plants that can be grown. This information may influence the choice of PDFs for parameters such as the plant/soil concentration ratio, plant yield, and nutrient and water contents of food types. The variation assigned to both the plant/soil concentration ratio and the aquatic transfer coefficient in the generic assessment (Section 8.5.1) may be substantially reduced at a specific location. The critical group may not be able to achieve self-sufficiency in agricultural produce at a very rocky or swampy site. In this case, imported, uncontaminated food could be assumed to make up part of the diet.

Modifications to CALDOS would be most extensive if the selected location is far north on the Canadian Shield where conventional agriculture is difficult and relatively unimportant. This would be analogous to the ACLIN 3 state discussed in the glaciation assessment (Section 12.3.2.5). In this case, the model may have to place more emphasis on subsistence hunting and gathering especially by aboriginal peoples. Additional food types representing wild plants (berries and wild rice) and wild animals (venison, upland game and waterfowl) may have to be explicitly introduced to supplement the five types already considered. New parameter values would have to be specified for these food types because they are not cultivated or raised domestically.

For plants, this would involve primarily yields, plant/soil concentration ratios and nutrient contents. Some information on these parameters for native species is already available in the literature (Sheppard M.I. and Thibault 1983, Zach et al. 1989), and the remainder could be obtained through further studies. For animals, values would be required for the nuclide transfer coefficients, for ingestion rates of forage, water and soil, and for the amount of food the animals would yield on harvesting. Although this type of information is difficult to obtain for wild species, some data do exist (Swanson 1985, Lowe and Horrill 1986, Swanson and Richert 1987, Clulow 1988, Zach et al. 1989), and more could be determined. Much of the available information on wild plant and animal foods of humans has already been incorporated in CALDOS.

The proportion of the human diet made up by the various food types would have to be redefined for a northern population, using, for example, data published by Nutrition Canada (1977). Imports of uncontaminated food might become very important. Cultural parameters such as food holdup times, occupancy factors and water demand per capita might also change. The PDFs may have to reflect values appropriate to crops grown in greenhouses, or by other advanced production techniques. It may also be necessary to examine the factors associated with dose and risk predictions to ensure that these factors apply to isolated aboriginal populations. DCFs in general may have to be changed in accordance with recommendations by bodies such as the ICRP, as discussed in Section 1.2.3.

14.6 ENVIRONMENTAL EFFECTS

Our methodology developed for assessing environmental effects (Chapter 13) readily lends itself to site-specific applications. However, the methodology can be more focussed, given a specific site. Our methodology consists of three approaches: relating human radiological safety criteria to potential environmental impacts; comparing predicted nuclide concentrations in water, soil and air to regulatory criteria and guidelines, as well as to environmental baseline concentrations; and comparing predicted radiological doses to four generic target organisms with effects known to occur at various dose levels. We discuss changes for each of these methods in turn, noting that many additional changes are the same as those indicated for the food-chain and dose submodel (Section 14.5).

For the first of our approaches, there would be little direct change. However, improved dose prediction for humans in a site-specific application would also strengthen our environmental assessment because we use man as a sensitive indicator species of potential radiation effects (Section 13.1.2). The second approach could be strengthened by establishing site-specific environmental baseline concentrations. The required concentrations can be readily measured using proper sampling and analytical procedures. These baseline data could then be used in a similar manner as those for the entire Canadian Shield established for our concept assessment for comparison with BIOTRAC predictions (Section 13.1.3). Finally, given a specific site, it might be possible to define more specific target organisms for dose prediction, organisms that more closely reflect local conditions than the four broadly generic organisms established for our concept assessment (Section 13.3.1). This might be particularly important in the case of an extreme northern or southern site. We have shown how this can be accomplished in Section 13.4.

In a site-specific application, our approach for assessing environmental effects could be closely tailored to local conditions. This would considerably strengthen the assessments and help to ensure environmental protection.

14.7 ENVIRONMENTAL CHANGE

A site-specific application would provide the opportunity to incorporate environmental change more fully into the model (Chapter 3). The location and geomorphological history of the site would allow its future evolution to be inferred. The physical state of the biosphere through which the nuclides migrate could then be changed throughout a model simulation. A model of lake infilling and the transition to land could be developed once detailed information on the area, depth, sedimentation rate and circulation patterns in a discharge lake were known. Conditions of climate and drainage would indicate how local soils could be expected to evolve. The topography, climate and state of tectonic uplift or subsidence would determine the local rate of denudation, and the extent that river channels would be eroded. The nature of future climatic variations could be deduced with greater certainty once the location of the site was known. From all of this information, logical inferences could be made regarding the ecology, and human behaviour and cultural practices at the discharge zone.

A site-specific application would also allow glaciation to be treated more fully (Chapter 12). The location of the site would determine the ice depth, and the duration of ice cover to be expected during future glacial advances. A southerly site might experience repeated advances and retreats during successive interstadials, whereas a northerly site might be continuously ice-covered. The location of the site would also determine the amount of isostatic depression caused by the ice and the extent of possible surficial fracturing and downwarping. The local topography would determine how much glacial erosion could be expected at the site, and indicate the effect of the glacier on local drainage systems. It may be possible to estimate the depth and composition of deposits that would be left at a specific location by a retreating glacier. Finally, the characteristics of the site may provide an indication of how effectively a glacier would disperse the nuclides that might have accumulated in the biosphere during the previous interglacial state, and whether nuclides would continue to discharge at the site following glacial retreat.

Site-specific information such as this could be used to produce a model to predict the state of the environment over time in the vicinity of a particular discharge zone. A number of such models are currently under development internationally (Petrie et al. 1981, BRGM 1985, Frizelle 1986). Predictions of nuclide migration through an evolving environment would be more realistic than predictions for a steady-state system, which must be used for the generic assessment (Chapter 12).

14.8 DISCUSSION

The conclusion to be drawn from the preceding sections is that the current generic version of BIOTRAC could be successfully modified for application at a specific location. Only minor changes would be required to make the model itself suitable for a site-specific assessment. Some site-specific data could be derived from the literature, and the rest could be directly measured at the site using currently available techniques. Model predictions for a specific site would very likely fall within the range of predictions of the generic model.

At a specific location, it would be possible to take advantage of local information to refine the model in several ways. In particular, a site-specific application would provide the opportunity to improve our models of terrestrial discharge, environmental change, and the lateral transport of nuclides by wind and water. This would lead to improved concentration and dose predictions as the conservative assumptions required for the generic assessment were replaced by more realistic approaches. Furthermore, the variability in many parameters could be reduced because it would no longer be necessary to account for spatial variability over the entire Canadian Shield in the PDFs. However, even in a site-specific application the use of conservative parameter values would remain important. The effects of all this would likely be lower and less variable concentrations and doses than those predicted by BIOTRAC for our generic postclosure assessment (Goodwin et al. 1994).

Site-specific applications of our models lie in the future and depend on the acceptability of the disposal concept. The current high level of interest in the environment, and nuclear fuel waste management in particular, will continue to stimulate research relevant to the behaviour and transport of nuclides in the biosphere, and the effects of chemical and radiological toxic substances on humans and other biota. Consequently, existing databases, models and other assessment methodologies will improve continuously. Clearly, the intention is to keep abreast and to contribute to these improvements in order to ensure that site-specific assessment results would be of the highest quality. Some of our ongoing studies are listed in Appendix F.

15. SUMMARY AND CONCLUSIONS

15.1 SUMMARY OF THE MODEL

In this report, we have developed and documented the biosphere model, BIOTRAC, used to evaluate the environmental and health impacts of the postclosure phase of the concept for disposal of Canada's nuclear fuel waste.

We have shown how the available data from field and experimental studies, and current theoretical understanding, have been synthesized into a model capable of predicting both nuclide concentrations throughout the biosphere and radiological doses to humans and other biota. This report is one of nine primary references (Figure 1-4) to support the EIS for the scientific review and public hearings of the disposal concept.

BIOTRAC was developed specifically to assess the postclosure impacts associated with a disposal facility, taking into account the information requested by the Federal Environmental Assessment Review Panel (1992) guidelines and the regulatory guidelines put forward by the AECB (1987). The important processes and pathways associated with transport from an underground source over very long times were identified through rigorous scenario analyses. These processes and pathways were modelled using a systems variability analysis approach directed by the executive code SYVAC3, which allows BIOTRAC to be linked with the vault and geosphere models (Figure 1-3), and which provides a way to quantify the variability and uncertainty in model predictions. The model was made generic because potential facility locations cannot be searched for before the disposal concept has been accepted (Joint Statement 1981). The suite of parameter values sampled at the beginning of each simulation represents no particular site, but one of a range of possible Canadian Shield sites in Ontario. The model and parameter values can, however, be readily modified to treat a specific location. Doses are calculated for a critical group of people living in the Canadian Shield environment who receive the greatest exposure because of their location, lifestyle and diet. Doses are also calculated for generic biota that inhabit the same location of potentially highest exposure.

The model was developed to provide valid predictions over a period of about 10 000 a during which time interglacial conditions are assumed to persist. Reasoned arguments supported by some simple calculations were used to show that beyond 10 000 a, when processes such as glaciation might begin to induce environmental change, doses will not suddenly or dramatically increase.

To ensure that computer requirements did not become impractical, the various transport processes were modelled in as simple and efficient a manner as possible, consistent with accuracy and completeness. In areas where knowledge was lacking and where realistic models could not be formulated or validated, we made conservative assumptions to ensure that consequences would be overestimated. Although BIOTRAC was developed specifically for the postclosure phase of the concept, it is consistent with the preclosure assessment model, PREAC, given the different aims of the two models and the different types of releases they address.

BIOTRAC was designed to estimate nuclide concentrations in the environment and doses to humans and other biota from nuclides escaping from an underground vault. It achieves this by simulating the transport of nuclides from the point where they discharge from the geosphere through the biosphere to various organisms. Once in the biosphere, transport is modelled by considering four separate but closely linked compartments representing surface water, the soil, the atmosphere and the food chain. These compartments are the main submodels of BIOTRAC; a fifth compartment, focusing on environmental protection, is closely related to the four compartments, particularly to the food-chain compartment.

BIOTRAC is driven by the output of the geosphere model. The primary point of nuclide discharge is through compacted sediments to the lake; however, we assume that a small portion of each of three discharge zones underlies a terrestrial area. These areas form four fields that are available to the critical group and other biota. Another primary point of nuclide discharge may be a domestic bedrock well drilled into the contaminant plume. Interface models couple the geosphere and the biosphere at each of these contact points. Concentrations for compacted sediments are calculated on the assumption that the flow through them is advection-dominated and that nuclides in the flow are partitioned between the solid and liquid phases. The predicted nuclide flow out of the compacted sediments and out of the well is used directly to drive the surface water submodel. Some of the parameters required in the surface water submodel were assigned fixed values representative of the WRA to ensure consistency with the site-specific geosphere model. The soil submodel for a groundwater source is driven by the nuclide concentration in the lowest of four soil layers. The concentration is calculated from the nuclide flow out of the geosphere using a mass balance equation. Finally, nuclide concentrations in the water taken from a bedrock well drilled into the groundwater plume are calculated with the help of a two-dimensional analytical model that is part of the geosphere model.

The surface water submodel is formulated as coupled, time-dependent mass balance equations for nuclide concentrations in the water column and mixed sediment on top of the compacted sediments. The model is driven by the nuclide flow out of the geosphere, and takes into account the processes of flushing, dilution, mixing, sedimentation, gaseous evasion, and radioactive decay and ingrowth. Nuclide inputs to the lake via runoff and atmospheric deposition, losses from suspension of particulate nuclides into the air, and resuspension of nuclides from sediment to the water column, are all treated implicitly.

The prediction of soil concentration is based on the SCEMR1 model, a mechanistic model that provides realistic estimates of nuclide migration through the soil profile. SCEMR1 is a one-dimensional, time-dependent model that calculates daily water flows between four soil layers using the Darcy equation and the equation of continuity. Nuclides introduced into the soil profile may be advected downward by leaching, or upward by capillary rise. The nuclide concentration in each soil layer is calculated using a mass balance equation that takes account of sorption and advection, and that assumes uniform, instantaneous mixing of the nuclides within the layer. The output of SCEMR1 is the time-dependent contribution to nuclide concentration in the soil root zone for each of three contamination pathways: groundwater discharge, irrigation and atmospheric deposition. In BIOTRAC, SCEMR1 results for these pathways are approximated by a simple analytical expression that depends on the steady-state root-zone concentration and on the time to steady state. Values for these variables are available from a regression analysis of SCEMR1 results. The analytical expressions used to write the mass balance equations for the root zone take account of gaseous evasion, cropping losses, and radioactive decay and ingrowth. The solutions provide the root-zone soil concentrations for each contamination pathway by which a field can become contaminated. The soil concentration in each field is then found by summing over the pathways.

The transport equations defining the surface water and soil submodels are solved by the response function/convolution approach used throughout SYVAC3 to treat time-dependent systems.

The atmosphere receives its nuclides indirectly via suspension from contaminated surface water, soil and vegetation. A number of different suspension mechanisms, both natural and anthropogenic, are treated in the atmosphere submodel, including the suspension of particulate nuclides from terrestrial and aquatic sources, the evasion of gases from terrestrial and aquatic sources, and the release of nuclides when biomass is burned. The models chosen to simulate each mechanism reflect our theoretical understanding of the process and the quantity and quality of the available data. In some cases, we assume that air concentrations are directly proportional to the nuclide concentration in the source compartment. The proportionality constants, which take the form of mass loading parameters, account for a number of suspension mechanisms collectively, and also account for dispersion. For other mechanisms, it is possible to formulate models to predict the nuclide flux to the atmosphere, which can be combined with a dispersion model to calculate air concentrations. For a ground-level area source, we based our dispersion model on the trajectory simulation approach. In all cases, the models assume that air concentrations adjust instantaneously to changes in the concentration of the source compartment. Separate indoor and outdoor concentrations are calculated for nuclides, taking into account the suspension mechanisms in which a nuclide can participate.

The atmosphere submodel also predicts the rate at which nuclides are deposited from the air to soil and vegetation. We use the concept of deposition velocities to model the dry deposition process, and the washout ratio approach to treat wet deposition.

The food-chain and dose submodel, CALDOS, traces nuclide movement from the physical compartments of the biosphere through the food chain to humans, and calculates doses from both internal and external exposure pathways. Doses from each pathway are predicted using simple multiplicative chain models that assume steady-state conditions. The model takes account of such transport processes as root uptake, contamination of plant surfaces by atmospheric deposition and irrigation, transfer through terrestrial animals, losses from plant surfaces by environmental processes, and radioactive decay and ingrowth. The internal exposure pathways modelled for man are the ingestion of contaminated plants, animals, water and soil; the ingestion of animals and fish that have consumed contaminated plants, water and soil; and the inhalation of contaminated air. The external pathways modelled for man are immersion in contaminated air, immersion in contaminated water, exposure to contaminated soil and exposure to contaminated building materials. Man's food and water ingestion rates, and his inhalation rate, are calculated in a self-consistent way in the model from his total energy need, his diet, and the nutritional contents of the foods in his diet. Other resources of the critical group, including number of livestock, field areas and water demand, are also calculated in an internally consistent manner.

Alternative approaches to transport modelling and dose calculation are used for a few nuclides with special properties. Internal tritium doses are calculated using a specific-activity model because tritium is very mobile in the environment. A limited specific-activity model is also used for ^{129}I because the internal ^{129}I doses are dominated by the thyroid gland, and the iodine content of the thyroid is regulated metabolically. The internal doses of both ^{129}I and ^{14}C are limited by the specific activity of these nuclides in the groundwater discharging to the biosphere. Many of the transport and exposure pathways need not be considered for the noble gases, which do not accumulate in the biosphere. Short-lived daughter radionuclides with half-lives less than one day are assumed to be in secular equilibrium with their precursors throughout the biosphere; the contribution of these nuclides to dose are accounted for through their precursors.

BIOTRAC includes a model for calculating radiological doses to four generic target organisms (plants, mammals, birds and fish). This model is closely related to the four submodels of BIOTRAC, particularly CALDOS. Thus, the model primarily receives input from the surface water, soil and atmosphere submodels. Nuclides are then traced through the food chain to the target organisms. Appropriate internal and external exposures are taken into account for each of them. The predicted doses can be compared with doses at which certain effects have or have not been observed, as published in the literature. The model for assessing doses to non-human biota is one of three approaches described by us to evaluate environmental effects. One of the two other approaches use man as a sensitive indicator species, and the other uses baseline data on nuclide concentrations in the Canadian Shield environment that can be compared with BIOTRAC predictions.

Sensitivity analyses were carried out for BIOTRAC and for each of the four submodels separately to help identify the most important nuclides, pathways and model parameters. Results also helped to establish confidence by indicating that the models performed as expected. Mass balance studies showed that BIOTRAC creates some nuclide mass, as expected, and that this has little influence on dose predictions for man.

Values and PDFs for the many parameters appearing in BIOTRAC were set only after careful appraisal of the available data, and of the role of each parameter in the model. In general, the parameters were distributed to account for uncertainties in the data or the model, and to reflect the variability that could be encountered in space or time on the Ontario portion of the Canadian Shield. Where possible, the PDF type and attributes for a given parameter were determined by statistical analysis of the available data. Where the data were insufficient for this purpose, the PDFs were set using reasoned arguments, taking into account all the available information. The PDFs were truncated or correlated, as necessary, to prevent unreasonable values, or combinations of values. Fixed values were assigned only to parameters that showed relatively little variation, that had little impact on the calculated consequences, and for which a clearly conservative value could be identified. Most of the values were drawn from the literature, but some were supplied by our own research program. Where possible, the values used were annual averages based on data from the Canadian Shield. For each parameter, we demonstrated how the available data were used to construct a suitable distribution for use in BIOTRAC.

BIOTRAC does not currently allow parameters defining the physical state of the biosphere to vary throughout a simulation. Environmental change is therefore not explicitly modelled, but it is accounted for in other ways. The effect of fluctuating processes on predicted consequences is handled implicitly through the use of distributed parameter values, as long as it can be assumed that concentrations and doses respond rapidly to changes in the parameter values. Our PDFs reflect primarily spatial variability across the Shield, which probably exceeds the temporal changes experienced at any one site during currently prevailing interglacial conditions. Of the many long-term transitional processes that could affect the biosphere over the lifetime of the disposal facility, only human activities and glaciation (including glacially induced faulting and succession in a glacial regime) have the potential to alter the rate of nuclide transfer to the biosphere and its biota. Because of the uncertainty involved in predicting future human culture and technology, we have not attempted to account for anthropogenic effects in the model. The exception to this is human intrusion in the form of a bedrock well drilled into the groundwater plume. Many of the changes that could occur in human culture and lifestyle are effectively taken into account through the critical group concept.

Glaciation was assessed by using a modified version of BIOTRAC to calculate doses to man for conditions representative of a cold interstadial climate, and by qualitatively evaluating a number of glacially induced pathways. This work demonstrated that doses throughout the glacial cycle would be comparable to those calculated by BIOTRAC for present interglacial conditions. Finally, we account for succession following retreat of the ice by modelling, in a rough way, the end point of lake infilling and use of sediment as soil.

15.2 ASSUMPTIONS

As a model of nuclide movement through the biosphere, BIOTRAC is a representation of a very complex system. As such, it involves a large number of assumptions that allow the system to be reduced to a manageable level for modelling purposes. The assumptions associated with the four individual submodels have been discussed in detail in Chapters 5 to 8; the assumptions concerning the interface model and the model focusing on environmental protection are documented in Chapters 4 and 13. Here we examine some of the major assumptions that run throughout BIOTRAC as a whole, and discuss their effect on the predicted concentrations and doses.

1. *Nuclide behaviour in the biosphere is well enough understood that the consequences of a used-fuel disposal facility can be reliably assessed.* The science of biosphere modelling is founded on a number of mature disciplines, each of which has been studied for more than a century. A vast amount of information, accumulated over several decades in many countries, is available on nuclide transport and dosimetry. Research programs established in Canada and abroad over ten years ago have helped to solve the special challenges raised by nuclear waste management. Results from all of these studies have been used to develop mathematical models, which have been shown through validation against observational data to reproduce the essential features of nuclide transport through the biosphere on short time scales. We therefore have sufficient understanding to

assess the impacts of a disposal facility. The level of confidence that should be placed in BIOTRAC predictions is discussed further in Section 15.3.

2. *The data required to run the model are available.* Since BIOTRAC is implemented probabilistically, the database must support the construction of PDFs for many model parameters that reasonably reflect the probabilities that the parameters will assume particular values. As used in BIOTRAC, the PDFs are meant to account for all sources of uncertainty in the data and in the model, as well as spatial variability across the Ontario portion of the Canadian Shield and temporal variability during interglacial conditions, represented by current conditions. For most parameters, there are too few data available to separate these different contributions to the probability of different possible values. Furthermore, it is not clear that the different contributions can be combined into a single PDF in a statistically meaningful way.

   In the face of these difficulties, we adopted a number of different measures (Stephens et al. 1989) to ensure that the PDFs used in the model will not result in an underestimate of concentrations and doses. We made no attempt to separate the different contributions to the range of possible values of a given parameter; instead, we constructed the PDFs from the pool of all the relevant data, which should reflect all the sources of uncertainty. Statistical and theoretical arguments, analogs and expert opinion were all used to aid in defining the PDFs. Wherever doubt existed concerning a particular parameter, we adopted values or PDFs that we believe are conservative. In particular, the PDFs for the more variable parameters, and those with the greatest impact on consequences, were chosen to be lognormal. The tails of these distributions extend beyond the most extreme observed values of the parameters, and so allow for conditions in space or time that have not yet been observed. PDFs constructed in this way should provide a comprehensive representation of the distribution of possible parameter values. When used in BIOTRAC, they should not result in an underestimate of the consequences that could arise from the disposal facility.

3. *The processes governing nuclide movement through the biosphere can be modelled using simple transfer coefficients.* For example, sedimentation in lakes is simulated using a first-order rate constant, sorption processes in soil and sediments are described by solid/liquid partition coefficients, the suspension of particulate material into the atmosphere is modelled using a dust-loading parameter, and the uptake of nuclides through plant roots is simulated by concentration ratios. In each of these cases, a simple transfer coefficient is used to describe the net effect of a process, or set of processes, that in reality are extremely complex and may involve interacting physical, chemical and biological components. We do not have a complete theoretical understanding of most of these processes; similarly, the data required for detailed modelling are sometimes unavailable. For these reasons, it is impractical to simulate each process individually, and to attempt to do

so would result in large uncertainties. On the other hand, transfer parameters are empirically based. They can be readily measured in the field and the laboratory, and their values implicitly account for all the processes and effects. At our present level of understanding, models based on transfer coefficients perform as well or better than process-oriented models. Transfer coefficients are widely used, and are the commonly accepted approach in assessment modelling.

4. *The varying environmental behaviour of different chemical species of the same nuclide can be accounted for through the parameter distributions.* In general, on reaching the biosphere, a given nuclide could be in any one of a number of different chemical forms, which may vary considerably with respect to mobility. For some nuclides, chemical form can also influence health effects. Moreover, the chemical form could change as the nuclide moves through the biosphere and encounters different environmental conditions. For the most part we do not take account of the chemical form when modelling the transfer of a given nuclide. For volatile nuclides, we include additional pathways that are unique to the gaseous form, and so model the most mobile form. But for the remaining nuclides, we assume that the behaviour of different chemical species can be accounted for through the parameter distributions. In most cases this is likely a good assumption. The PDFs are formed by considering all the relevant field and experimental data, and so include values representative of the different chemical forms present in the biosphere. In this way, we cover the range of chemical behaviour of a given nuclide without having to specify its particular chemical form.

5. *The effects of some complex processes on nuclide transport can be implicitly accounted for by not depleting the nuclide inventory in the source compartment when nuclides are transferred from one compartment to another.* For example, only 5% of the nuclides taken up by plants in the model is permanently lost from the soil; the remaining 95% is effectively returned, accounting for recycling of the nuclides through waste products and the decay of the organisms involved. Similarly, soil concentrations are not reduced when contaminated particles are suspended into the atmosphere. This implies that atmospheric particulate material derived from the soil need not be included in the depositional flux when soil concentrations arising from atmospheric deposition are calculated. As a final example, the nuclide flow used to drive the surface water model is not depleted by the nuclide discharge to terrestrial areas. This accounts implicitly for the nuclides that would reach the lake with runoff water flowing over soil.

   Recycling, runoff and the suspension/deposition cycle are complex, time-dependent processes that are strongly site-specific. They are difficult to model realistically when the topography of the site, the predominant wind directions and the relative locations of the fields and the water body are unknown. By not reducing nuclide inventories in donor compartments, we can model these processes simply and conservatively for both donor and receptor compartments. If a mass balance were enforced, a transfer parameter that produced a conservative result in one compartment could underestimate the concentration in the other, and possibly underestimate doses. The cost of treating these processes as we do is that nuclide mass is generated by the model. However, the amounts created are small and do not significantly increase the predicted concentrations and doses.

6. *Feedback of potential effects on the system modelled are not taken into account.* Our model is unidirectional in the sense that there is no feedback from potential effects caused by the release of nuclides to the biosphere. If there were effects, they might change agricultural and ecological systems, and this might necessitate model changes. Our assumption is valid as long as there are no such disruptive effects. Given the many broadly distributed parameters in our model, it is reasonable to assume that minor effects are accounted for, and so the absence of any effects is not a prerequisite for model validity. Furthermore, the rigorous regulatory criteria and guidelines for environmental and health protection ensure that there will not be any appreciable disruptive effects.

7. *For the most part, the model does not account for seasonal changes in the biosphere, or for the effects of such changes on nuclide transfer.* In particular, most processes and parameter values in the model do not reflect winter conditions. This generally results in conservative predictions. Because biological activity slows down in winter, and because the soil and most water bodies are frozen, nuclides are less mobile than they are in summer. Winter conditions would also reduce atmospheric suspension and deposition so that the snow melt would not cause a sudden flush of nuclides. However, we assume that many processes, such as gaseous evasion from soils and lakes, are active throughout the year, with parameter values based on summer conditions. For other parameters, such as the plant interception fraction, we adopt values that are representative of the largest values that could occur throughout the year. Many processes that are unique to the winter season tend to reduce concentrations and doses. For example, snow cover would shield humans and many other biota from contaminated soil and lessen their external dose from ground exposure. Similarly, melting of the snowpack in the spring would leach nuclides out of the soil and reduce the root-zone concentration. It is therefore conservative to ignore such processes. For processes that remain active throughout the year, and that can be simulated using a linear model (e.g., sedimentation in lakes), the use of annual average parameter values provides a valid description of the process on an annual basis.

8. *The long-term evolution of the biosphere, as a result of natural or anthropogenic effects, is not modelled explicitly.* We do account, however, for environmental and cultural change in other ways. Our approach to these issues is discussed in Section 15.1.

15.3 VALIDITY OF THE MODEL

Complete experimental validation of BIOTRAC is not possible. The observational data against which to compare model predictions do not exist for an underground source, even over short periods of time. The data are obviously not available for comparison over the full duration of the postclosure assessment period. In the face of this difficulty, we have used a variety of other approaches to establish BIOTRAC's credibility, as discussed for each submodel separately in Chapters 5 to 8 and for BIOTRAC as a whole in Chapter 11.

The surface water, soil and atmosphere submodels have received some experimental validation. The predictions of the soil and surface water models were found to agree well with observations from experiments that lasted over several years, and that dealt with a variety of nuclides under various conditions representative of the Canadian Shield. These models therefore appear to address, in an appropriate way, the processes important for predicting nuclide concentrations in soils and surface waters, at least over short periods of time. Similarly, the dispersion relationships used in the atmosphere submodel were derived from a model that agrees well with experimental data.

We have subjected BIOTRAC and its parameter values to a rigorous qualitative evaluation. A detailed scenario analysis helped to identify the features, events and processes that needed to be included in the model. The development of each submodel began with a thorough literature review to extract the latest and best information on the relevant processes, and on associated modelling approaches and parameter values. These were combined with findings from our own research program to develop an appropriate model for assessing the performance of an underground disposal facility. We have incorporated international experience in biosphere modelling through periodic meetings with groups from various countries and international organizations. The model, the data and the overall research program have been continually exposed to peer review through publication in the open literature, formal independent reviews, and review by the TAC. Through these approaches, we have demonstrated that BIOTRAC reflects the accumulated body of knowledge on nuclide transport through the biosphere, and that it follows current scientific practices for models designed to assess geological disposal systems. The similarity among assessment models worldwide, and the agreement between their predictions in model intercomparison studies, suggest that a consensus exists regarding the processes and pathways of importance, and the way in which they should be modelled.

We have ensured the validity of the model in a number of other ways. We have used natural analogs of nuclide transfer in the biosphere to determine parameter values for use in BIOTRAC, and to develop approaches to modelling transfer over long periods of time. From the outset, the work on which BIOTRAC is based has been subject to an informal quality assurance program guided by well-established scientific principles. A thorough sensitivity analysis of the model has increased our understanding of its behaviour, and has shown that it responds as we expected on an intuitive basis. In areas where our knowledge is lacking, and where realistic models cannot be formulated or validated, we have made assumptions to ensure that consequences are not underestimated. Examples of conservatism can be found in all aspects of the model, including its underlying philosophy, its input data, and its formulation. The use of a systems variability analysis approach provides quantitative estimates of the variability and uncertainty in the predictions, and increases the confidence concerning the probability of high concentrations and doses. Finally, we are reviewing the model on an ongoing basis, and incorporating new data from the literature and our own work to keep it up-to-date (Appendix F).

15.4 CONCLUSIONS

On the basis of the evidence presented in this document, we believe that BIOTRAC and its parameter values provide a state-of-the-art tool suitable for the postclosure phase of the assessment of the concept for disposal of Canada's nuclear fuel waste. From our careful work and validation efforts, we believe that the model provides a satisfactory description of nuclide movement through the biosphere, and that its predictions provide adequate estimates of the resulting environmental concentrations and doses to humans and to various biota. This is important to ensure the long-term protection of the environment and humans from potential chemical and radiological toxic effects resulting from the release of any nuclides from the disposal facility. We believe that the many assumptions made in formulating the model and setting parameter values are justified, and lead, on balance, to conservative predictions. The model will continue to be updated as new information becomes available.

We conclude that BIOTRAC provides a suitable and satisfactory description of nuclide behaviour in the biosphere, and that it will not underestimate doses to humans and other biota when used to assess the concept for disposal of Canada's nuclear fuel waste.

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APPENDIX A

LIST OF ACRONYMS, NAMES AND ABBREVIATIONS

ACLIN	Astronomical CLimatic INdex
AECB	Atomic Energy Control Board
AECL	Atomic Energy of Canada Limited
AEL	Angus Environmental Limited
AES	Atmospheric Environment Service
AMS	American Meteorological Society
BEIR	Committee on the Biological Effects of Ionizing Radiation (United States)
BIOMOVS	BIOsphere MOdel Validation Study
BIOTRAC	BIOsphere TRansport And Consequence model
BRGM	Bureau de Recherches Géologiques et Minières (France)
CALDOS	CALculation of DOSe food-chain and dose model
CEC	Cation Exchange Capacity
CMHC	Canada Mortgage and Housing Corporation
CNFWMP	Canadian Nuclear Fuel Waste Management Program
COG	CANDU Owners Group
CSA	Canadian Standards Association
CWQG	Canadian Water Quality Guidelines
DCF	Dose Conversion Factor
DSMA	Dilworth Secord Meagher and Associates Ltd.
EARP	Environmental Assessment Review Process
EEC	European Economic Communities
EIS	Environmental Impact Statement
ELA	Experimental Lakes Area
EMR	Energy Mines and Resources Canada
ET	EvapoTranspiration
EURATOM	European Atomic Community (EEC)
EWAM	Energy Water Air Model for humans
FEARO	Federal Environmental Assessment Review Office
FOOD III	Food chain model, Version III
FW FISH	Freshwater fish food type
GEONET	GEOsphere NETwork model
GM	Geometric Mean
GSD	Geometric Standard Deviation
HRI	Holcomb Research Institute
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
IIASA	International Institute of Applied Systems Analysis
IUR	International Union of Radioecologists
MNR	Ministry of Natural Resources (Ontario)
MOTIF	Model Of Transport In Fractured porous media
MREG	Multiple REGression

NAGRA	NAtionale Genossenschaft für die Lagerung Radioaktiver Abfälle (Switzerland)
NAPS	National Air Pollution Surveillance network
NAS	National Academy of Sciences (United States)
NEA	Nuclear Energy Agency (OECD)
NCRP	National Council on Radiation Protection (United States)
NFWMP	Nuclear Fuel Waste Management Program
NWT	Northwest Territories
OECD	Organisation for Economic Co-operation and Development
OEPCB	Ontario Economics and Policy Coordination Branch
OME	Ontario Ministry of the Environment
PCBs	Polychlorinated Biphenyls
PDF	Probability Density Function
PREAC	Preclosure Radiological Environmental Assessment Code
QA	Quality Assurance
SAS	Statistical Analysis Systems
SCEMR1	Soil Chemical Exchange and Migration of Radionuclides model, Revision 1
SD	Standard Deviation
SENSYV	SENsitivity for SYstem Variability analysis
STP	Standard Temperature and Pressure
SYVAC3	SYstems Variability Analysis Code - Generation 3
TAC	Technical Advisory Committee
TE BIRD	Poultry and egg food type
TE MEAT	Mammalian meat food type
TE MILK	Milk and dairy product food type
TE PLANT	Terrestrial plant food type
TEHM	Terrestrial Ecosystem Hydrology Model
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
URL	Underground Research Laboratory
USEPA	United States Environmental Protection Agency
USNRC	United States Nuclear Regulatory Commission
WL	Whiteshell Laboratories
WRA	Whiteshell Research Area
WHO	World Health Organization
YT	Yukon Territory

APPENDIX B

LIST OF SYMBOLS

Symbol	Symbol Description	Unit
a	regression constant used to calculate the volume of irrigation water applied to the garden and forage field	unitless
a_G, b_G, c_G, d_G, e_G, f_G, g_G	regression constants used to calculate the steady-state root-zone soil concentration in the groundwater case	unitless
a_{ID}, b_{ID}, c_{ID}, d_{ID}, e_{ID}, f_{ID}, g_{ID}, h_{ID}, l_{ID}, m_{ID},	regression constants used to calculate the steady-state root-zone soil concentration in the irrigation/deposition case	unitless
A	nominal area of an agricultural field, sediment layer, etc.	m^2
A_d	catchment area	m^2
A_{DZ}	total area of discharge zone	m^2
A_e	area of field e	m^2
A_F	area of the forage field	m^2
$A_{F1,2,3,4}$	areas of the agricultural fields (peat bog, garden, forage field and woodlot respectively)	m^2
A_I	total activity of ^{129}I per unit mass of man's thyroid gland	$Bq \cdot kg^{-1}$ thyroid
A_l	area of the lake	m^2
A_P	area of peat bog required to provide sufficient peat to heat a home	m^2 or $m^2 \cdot a^{-1}$

continued...

Symbol	Symbol Description	Unit
A_{sed}	area of the lake bottom covered by sediments	m^2
A_T	area of terrestrial contamination	m^2
A_{TD}	area of arable terrestrial discharge, bottom soil layer or shallow soil associated with a given discharge zone	m^2
$A_{TD1,2,3}$	areas of arable terrestrial discharge associated with each discharge zone, ranked according to pore-water concentration	m^2
A_V	area of the garden	m^2
A_W	forest area required to supply sufficient wood fuel to heat a home	m^2 or $m^2 \cdot a^{-1}$
AADL	aquatic atmospheric dust load	m^3 water$\cdot m^{-3}$ air
ADL	terrestrial atmospheric dust load	kg dry soil$\cdot m^{-3}$ air
AF	atmospheric pathway in which particles are suspended via agricultural fires	unitless
AG	atmospheric pathway in which gases are suspended from aquatic sources	unitless
AIML	aquatic iodine mass loading parameter	m^3 water$\cdot m^{-3}$ air
AP	atmospheric pathway in which particles are suspended from aquatic sources	unitless
ATC^{Rn}	aquatic transfer coefficient for radon	$m \cdot s^{-1}$

continued...

Symbol	Symbol Description	Unit
b	regression constant used to calculate the volume of irrigation water applied to the garden and forage field	unitless
B_j^i	aquatic concentration ratio for nuclide i and food type j = FW FISH	m^3 water·kg^{-1} wet biomass or L water·kg^{-1} wet biomass
Bc	carbon content of soft tissue in man's body	kg carbon
BH	building height	m
Bs	mass of soft tissue in man's body	kg soft tissue
Bv^i	plant/soil concentration ratio for nuclide i	(mol·kg^{-1} wet biomass)/ (mol·kg^{-1} dry soil) or (Bq·kg^{-1} wet biomass)/ (Bq·kg^{-1} dry soil)
BVOL	building volume	m^3
BW	building width	m
c	regression constant used to calculate the volume of irrigation water applied to the garden and forage field	unitless
C	nuclide concentration in an environmental compartment that acts as a source of contamination for the food chain	unitless
C_a^i	concentration of nuclide i in air	mol·m^{-3} air or Bq·m^{-3} air
$(C_a^i)_{AF}$	air concentration of nuclide i from agricultural fires	mol·m^{-3} air

continued...

Symbol	Symbol Description	Unit
$(C_a^i)_{AG}$	air concentration of nuclide i suspended from the lake as a gas	$mol \cdot m^{-3}$ air
$(C_a^i)_{AP}$	air concentration of nuclide i suspended from the lake as particles	$mol \cdot m^{-3}$ air
$(C_a^i)_{EF}$	air concentration of nuclide i caused by home-heating fires	$mol \cdot m^{-3}$ air
$(C_a^i)_I$	concentration of nuclide i in indoor air	$Bq \cdot m^{-3}$ air
$(C_a^i)_{IGW}$	air concentration of nuclide i released indoors as a gas from domestic water	$mol \cdot m^{-3}$ air
$(C_a^i)_k$	air concentration of nuclide i via pathway k	$mol \cdot m^{-3}$ air
$(C_a^i)_{LF}$	air concentration of nuclide i from forest and land-clearing fires	$mol \cdot m^{-3}$ air
$(C_a^i)_O$	concentration of nuclide i in outdoor air	$Bq \cdot m^{-3}$ air
$(C_a^i)_{TG}$	air concentration of gaseous forms of nuclide i from terrestrial sources	$mol \cdot m^{-3}$ air
$(C_a^i)_{TP}$	air concentration of nuclide i suspended from terrestrial areas as particles	$mol \cdot m^{-3}$ air
$(C_a^{Rn})_{IGS}$	indoor air concentration of radon caused by diffusion from soil	$mol \cdot m^{-3}$ air
C_b^i	concentration of nuclide i in biomass (crops, trees or wood fuel)	$mol \cdot kg^{-1}$ wet biomass

continued...

Symbol	Symbol Description	Unit
C_D^i	concentration of nuclide i in irrigation water required to produce a flux to the soil surface equal to the flux from atmospheric deposition	$mol \cdot m^{-3}$ water
C_{ds}^i	concentration of nuclide i in compacted sediments	$mol \cdot kg^{-1}$ dry sediment
C_{gw}^i	concentration of nuclide i in groundwater	$Bq \cdot m^{-3}$ water or $mol \cdot m^{-3}$ water
C_{gw}^{sC}	concentration of stable carbon in groundwater	kg $^{12}C \cdot m^{-3}$ water
C_{gw}^{sI}	concentration of stable iodine in groundwater	kg $^{127}I \cdot m^{-3}$ water
C_I^i	concentration of nuclide i in the irrigation water	$mol \cdot m^{-3}$ water
C_{ibm}^i	concentration of nuclide i in inorganic building materials	$Bq \cdot kg^{-1}$ dry weight
C_j^i	concentration of nuclide i in environmental compartment j	$Bq \cdot kg^{-1}$ dry weight
$(C_j^i)_k$	concentration of nuclide i in environmental compartment j from transfer via pathway k	unitless
C_l^i	concentration of nuclide i in lake water	$mol \cdot m^{-3}$ water
C_{pw}^i	concentration of nuclide i in pore water of compacted sediments or soil	$mol \cdot m^{-3}$ water
C_R^G	ratio of ^{14}C to total carbon in groundwater	unitless
C_s^i	concentration of nuclide i in a soil layer or in peat	$mol \cdot kg^{-1}$ dry soil or $Bq \cdot kg^{-1}$ dry soil or $mol \cdot kg^{-1}$ dry peat

continued...

Symbol	Symbol Description	Unit
$(C^i_s(t))_D$	time-dependent concentration of nuclide i in the soil root zone for a deposition contamination	$mol \cdot kg^{-1}$ dry soil
$(C^i_s)_G$	steady-state concentration of nuclide i in the soil root zone from groundwater contamination	$mol \cdot kg^{-1}$ dry soil
$(C^i_s(t))_G$	time-dependent concentration of nuclide i in the soil root zone from groundwater contamination	$mol \cdot kg^{-1}$ dry soil
$(C^i_s(t))_I$	time-dependent concentration of nuclide i in the soil root zone from irrigation water contamination	$mol \cdot kg^{-1}$ dry soil
C^i_{se}	concentration of nuclide i in the top 0.3 m of mixed and compacted sediments	$mol \cdot kg^{-1}$ dry sediment
$C^i_{s,e}$	annual average soil concentration of nuclide i in the root zone of field e	$mol \cdot kg^{-1}$ dry soil
C^i_{sed}	concentration of nuclide i in the mixed sediment layer	$mol \cdot kg^{-1}$ dry sediment
C^i_{sol}	concentration of nuclide i on the sediment or soil solids	$mol \cdot kg^{-1}$ dry sediment or soil
C^H_w	concentration of hydrogen in water	g $hydrogen \cdot m^{-3}$ water
C^i_w	concentration of nuclide i in domestic water (lake water or well water)	$mol \cdot m^{-3}$ water or $Bq \cdot m^{-3}$ water
C^i_{wbm}	concentration of nuclide i in wooden building material from root uptake and atmospheric deposition	$Bq \cdot kg^{-1}$ dry biomass

continued.

Symbol	Symbol Description	Unit
C^i_{ww}	well-water concentration	$mol \cdot m^{-3}$ water
Cec	carbohydrate fuel value	$kJ \cdot g^{-1}$ carbohydrate
CEC	soil cation exchange capacity	$mol \cdot kg^{-1}$ dry soil
CM^i_p	maximum plant concentration equal to the concentration that would occur if all nuclides in the soil were taken up	$Bq \cdot kg^{-1}$ wet biomass
Cmw	carbohydrate metabolic water yield	m^3 water $\cdot g^{-1}$ carbohydrate
Co	carbohydrate STP oxygen combustion value	$m^3\ O_2 \cdot g^{-1}$ carbohydrate or $L\ O_2 \cdot g^{-1}$ carbohydrate
C(p)	Mallows statistic for multiple regression	unitless
$Cpw_{1,2,3}$	pore-water concentration in the lowest soil layer among the three terrestrial discharge zones, ranked according to pore-water concentration	$mol \cdot kg^{-1}$ dry soil
Cpw^i_j	concentration of nuclide i in pore water in soil layer j	$mol \cdot kg^{-1}$ water
Cs^i_j	concentration of nuclide i in soil layer j	$mol \cdot kg^{-1}$ dry soil
$(Css)^i$	normalized steady-state concentration of nuclide i in the soil root zone	$(mol \cdot kg^{-1}$ dry soil$)/(mol \cdot L^{-1}$ water$)$
$(Css)^i_G$	normalized steady-state root-zone soil concentration for nuclide i in the groundwater case	$(mol \cdot kg^{-1}$ dry soil$)/(mol \cdot L^{-1}$ water$)$

continued...

Symbol	Symbol Description	Unit
$(Css)^i_I$	normalized steady-state root-zone soil concentration for nuclide i in the irrigation case	$(mol \cdot kg^{-1}$ dry soil)/$(mol \cdot L^{-1}$ water)
$(Css)^i_{ID}$	normalized steady-state root-zone soil concentration for nuclide i in the irrigation/ deposition case	$(mol \cdot kg^{-1}$ dry soil)/$(mol \cdot L^{-1}$ water)
Cym_j	carbohydrate content of food type j	g carbohydrate$\cdot kg^{-1}$ wet biomass
d	regression constant used to calculate the volume of irrigation water applied to agricultural fields	unitless
D	amount of energy absorbed from radiation in human tissue	unitless
$(D^c)_U$	man's maximum total internal dose from ^{14}C	$Sv \cdot a^{-1}$
D^{H3}	man's total internal dose from tritium	$Sv \cdot a^{-1}$
D^i	total rate of deposition of nuclide i to underlying surfaces	$mol \cdot m^{-2} \cdot d^{-1}$ or $mol \cdot m^{-1} \cdot s^{-1}$
$(D^i)_A$	man's internal dose from inhalation of nuclide i	$Sv \cdot a^{-1}$
$(D^i)_{AI}$	man's external dose from immersion in air contaminated by nuclide i	$Sv \cdot a^{-1}$
$(D^i)_G$	man's external dose from ground contaminated by nuclide i	$Sv \cdot a^{-1}$
$(D^i)_S$	man's internal dose from ingestion of soil contaminated by nuclide i	$Sv \cdot a^{-1}$

continued...

Symbol	Symbol Description	Unit
$(D^i)_W$	man's internal dose from ingestion of drinking water contaminated by nuclide i	$Sv \cdot a^{-1}$
$(D^i)_{WI}$	man's external dose from immersion in water contaminated by nuclide i	$Sv \cdot a^{-1}$
D^I	man's total internal dose from ^{129}I	$Sv \cdot a^{-1}$
$(D^I)_U$	man's upper limit to the internal dose from ^{129}I	$Sv \cdot a^{-1}$
D_b^i	rate of deposition of nuclide i to vegetation	$mol \cdot m^{-2}$ soil$\cdot d^{-1}$ or $Bq \cdot m^{-2}$ soil$\cdot d^{-1}$
$(D_b^i)_D$	rate of atmospheric deposition of nuclide i to vegetation	$Bq \cdot m^{-2}$ soil$\cdot d^{-1}$ or $mol \cdot m^{-2}$ soil$\cdot d^{-1}$
$(D_b^i)_I$	rate of irrigation water deposition of nuclide i to vegetation	$Bq \cdot m^{-2}$ soil$\cdot d^{-1}$ or $mol \cdot m^{-2}$ soil$\cdot d^{-1}$
$(D_j^i)_{BM}$	man's external dose from exposure to building material j contaminated with nuclide i	$Sv \cdot a^{-1}$
$(D_j^i)_{LP}$	man's internal dose from ingestion of food type j = TE PLANT contaminated by leaf deposition with nuclide i	$Sv \cdot a^{-1}$
$(D_j^i)_{LPA}$	man's internal dose from ingestion of food types j = TE MEAT, TE MILK and TE BIRD contaminated by leaf deposition with nuclide i	$Sv \cdot a^{-1}$
$(D_j^i)_{RP}$	man's internal dose from ingestion of food type j = TE PLANT contaminated by root uptake with nuclide i	$Sv \cdot a^{-1}$

continued...

Symbol	Symbol Description	Unit
$(D_j^i)_{RPA}$	man's internal dose from ingestion of food types j = TE MEAT, TE MILK and TE BIRD contaminated by root uptake with nuclide i	$Sv \cdot a^{-1}$
$(D_j^i)_{SA}$	man's internal dose from ingestion of food types j = TE MEAT, TE MILK and TE BIRD contaminated by soil ingestion with nuclide i	$Sv \cdot a^{-1}$
$(D_j^i)_{WA}$	man's internal dose from ingestion of food types j = TE MEAT, TE MILK and TE BIRD contaminated by drinking water with nuclide i	$Sv \cdot a^{-1}$
$(D_j^i)_{WF}$	man's internal dose from ingestion of food type j = FW FISH contaminated with nuclide i	$Sv \cdot a^{-1}$
D_{ob}	overburden depth	m
D_s^i	rate of deposition of nuclide i from the atmosphere to the soil	$mol \cdot m^{-2}$ $soil \cdot d^{-1}$ or $mol \cdot m^{-2}$ $soil \cdot a^{-1}$
D_T	man's absorbed dose for an organ or tissue	Sv
D_w	well depth	m
$(DB_b)_{TT}$	total dose to non-human biota; b = fish (F), plant (P), mammal (M) or bird (B)	$Gy \cdot a^{-1}$
$(DBe_b^i)_A$	dose to non-human biota from immersion in air for nuclide i; b = plant (P), mammal (M), or bird (B)	$Gy \cdot a^{-1}$
$(DBe_b^i)_S$	dose to non-human biota from immersion in soil or sediment for nuclide i; b = fish (F), plant (P), mammal (M) or bird (B)	$Gy \cdot a^{-1}$

continued...

Symbol	Symbol Description	Unit
$(DBe_b^i)_V$	dose to non-human biota from immersion in vegetation for nuclide i; b = mammal (M) or bird (B)	$Gy \cdot a^{-1}$
$(DBe_b^i)_W$	dose to non-human biota from immersion in water for nuclide i; b = fish (F), plant (P), mammal (M) or bird (B)	$Gy \cdot a^{-1}$
$(DBe_b^i)_{Tot}$	total external dose to non-human biota for nuclide i; b = fish (F), plant (P), mammal (M), or bird (B)	$Gy \cdot a^{-1}$
DBi_b^i	dose to non-human biota from internal exposure for nuclide i; b = fish (F), plant (P), mammal (M) or bird (B)	$Gy \cdot a^{-1}$
DCF	dose conversion factor	various units
DD^i	rate of dry deposition of nuclide i from the atmosphere to underlying surfaces	$mol \cdot m^{-2} \cdot d^{-1}$
$DF^{c'}$	internal dose conversion factor for ^{14}C based on the specific-activity model	$(Sv \cdot a^{-1})/(Bq \cdot kg^{-1}$ soft tissue)
DF^{H3}	man's internal dose conversion factor for tritium	$(Sv \cdot a^{-1})/(Bq \cdot kg^{-1}$ soft tissue)
DF^I	man's internal dose conversion factor for ^{129}I	$(Sv \cdot a^{-1})/(Bq \cdot kg^{-1}$ thyroid)
DFa^i	man's air immersion dose conversion factor for nuclide i	$(Sv \cdot a^{-1})/(Bq \cdot m^{-3}$ air)
DFb^i	man's building material exposure dose conversion factor for nuclide i	$(Sv \cdot a^{-1})/(Bq \cdot kg^{-1}$ dry material)

continued...

Symbol	Symbol Description	Unit
$(DFB_b^i)_A$	dose conversion factor for air immersion of non-human biota for nuclide i; b = plant (P), mammal (M) or bird (B)	$(Gy \cdot a^{-1})/(Bq \cdot m^{-3}$ air)
$(DFB_b^i)_I$	dose conversion factor for internal exposure of nuclide i by non-human biota; b = fish (F), plant (P), mammal (M) or bird (B)	$(Gy \cdot a^{-1})/(Bq \cdot kg^{-1}$ wet biomass)
$(DFB_b^i)_S$	dose conversion factor for immersion of non-human biota in soil or sediment for nuclide i; b = fish (F), plant (P), mammal (M) or bird (B)	$(Gy \cdot a^{-1})/(Bq \cdot kg^{-1}$ dry soil or sediment)
$(DFB_b^i)_V$	dose conversion factor for immersion of non-human biota in vegetation for nuclide i; b = mammal (M), or bird (B)	$(Gy \cdot a^{-1})/(Bq \cdot kg^{-1}$ wet biomass)
$(DFB_b^i)_W$	dose conversion factor for water immersion of non-human biota for nuclide i; b = fish (F), plant (P), mammal (M) or bird (B),	$(Gy \cdot a^{-1})/(Bq \cdot m^{-3}$ water)
DFe^i	man's ingestion dose conversion factor for nuclide i	$Sv \cdot Bq^{-1}$
DFg^i	man's ground exposure dose conversion factor for nuclide i	$(Sv \cdot a^{-1})/(Bq \cdot kg^{-1}$ wet soil)
DFh^i	man's water immersion dose conversion factor for nuclide i	$(Sv \cdot a^{-1})/(Bq \cdot m^{-3}$ water)
DFi^i	man's inhalation dose conversion factor for nuclide i	$Sv \cdot Bq^{-1}$
DISP	atmospheric dispersion factor	$s \cdot m^{-1}$, $d \cdot m^{-1}$ $a \cdot m^{-1}$ or $s \cdot m^{-3}$

continued...

Symbol	Symbol Description	Unit
$(DISP)_A$	atmospheric dispersion factor for an aquatic area source at ground level	$s \cdot m^{-1}$
$(DISP)_B$	atmospheric dispersion factor for an elevated point source	$s \cdot m^{-3}$
$(DISP)_I$	atmospheric dispersion factor for indoor releases	$s \cdot m^{-3}$
$(DISP)_k$	atmospheric dispersion factor for pathway k	$s \cdot m^{-1}$ or $s \cdot m^{-3}$
$(DISP)_T$	atmospheric dispersion factor for a terrestrial area source at ground level	$s \cdot m^{-1}$
Dp	dose resulting from a particular exposure pathway	$Sv \cdot a^{-1}$
DW^i	rate of wet deposition of nuclide i from the atmosphere to underlying surfaces	$mol \cdot m^{-2} \cdot d^{-1}$
dws	dry/wet soil conversion factor	kg dry soil$\cdot kg^{-1}$ wet soil
e	field	unitless
E	evasive flux of CO_2 to the atmosphere from lake	$mol \cdot m^{-2}$ lake surface$\cdot a^{-1}$
$(E^i)_A$	man's intake rate of nuclide i via inhalation of contaminated air	$Bq \cdot a^{-1}$
$(E^i)_S$	man's intake rate of nuclide i via ingestion of contaminated soil	$Bq \cdot a^{-1}$
$(E^I)_T$	man's total annual intake of ^{129}I	$Bq \cdot a^{-1}$

continued...

Symbol	Symbol Description	Unit
$(E^i)_W$	man's intake rate of nuclide i via drinking water	$Bq \cdot a^{-1}$
E^{IS}	man's total intake of stable iodine	kg $^{127}I \cdot a^{-1}$ or μg $^{127}I \cdot d^{-1}$
E_D	amount of water evaporated from surface pools	m $water \cdot a^{-1}$
$(E^i_j)_{IRP}$	man's intake rate of nuclide i from ingrowth following root uptake of nuclide i - 1	$Bq \cdot a^{-1}$
$(E^i_j)_{LP}$	man's intake rate of nuclide i via ingestion of food type j = TE PLANT contaminated by leaf deposition	$Bq \cdot a^{-1}$
$(E^i_j)_{LPA}$	man's intake rate of nuclide i via food types j = TE MEAT, TE MILK and TE BIRD contaminated by leaf deposition of plants	$Bq \cdot a^{-1}$
$(E^i_j)_{RP}$	man's intake rate of nuclide i via food type j = TE PLANT contaminated by root uptake	$Bq \cdot a^{-1}$
$(E^i_j)_{RPA}$	man's intake rate of nuclide i via food types j = TE MEAT, TE MILK and TE BIRD contaminated by root uptake of plants	$Bq \cdot a^{-1}$
$(E^i_j)_{SA}$	man's intake rate of nuclide i via food types j = TE MEAT, TE MILK and TE BIRD contaminated by soil ingestion	$Bq \cdot a^{-1}$
$(E^i_j)_{WA}$	man's intake rate of nuclide i via ingestion of food types j = TE MEAT, TE MILK and TE BIRD contaminated by drinking water	$Bq \cdot a^{-1}$
$(E^i_j)_{WF}$	man's intake rate of nuclide i via food type j = FW FISH	$Bq \cdot a^{-1}$

continued...

Symbol	Symbol Description	Unit
E_o	evaporation occurring from open water bodies	m water$\cdot$a^{-1}
EF	atmospheric pathway in which nuclides are suspended through energy fires	unitless
EMFRAC^i	fraction of nuclide i released in a fire	unitless
$(\mathrm{EMFRAC}^i)_{AF}$	fraction of nuclide i released in an agricultural fire	unitless
$(\mathrm{EMFRAC}^i)_{EF}$	fraction of nuclide i released in an energy fire	unitless
$(\mathrm{EMFRAC}^i)_{LF}$	fraction of nuclide i released in a land-clearing fire	unitless
En	man's total energy need	kJ$\cdot$d^{-1} or kJ$\cdot$a^{-1}
EP	convertible energy content of peat	MJ$\cdot$kg^{-1} dry peat
ET	evapotranspiration	m water$\cdot$a^{-1}
EW	convertible energy content of wood	MJ$\cdot$kg^{-1} wet biomass
ewc	man's water/energy conversion ratio	m^3 water$\cdot$kJ^{-1} or L water$\cdot$kJ^{-1}
f	flushing rate of lake	a^{-1}
F_j	water flow between soil layers $j + 1$ and j	m^3 water$\cdot$m^{-2} soil
F^i_j	terrestrial animal transfer coefficient for nuclide i and food types j = TE MEAT, TE MILK and TE BIRD	d$\cdot$L^{-1} or d$\cdot$kg^{-1} wet biomass
f_1	frequency of forest or land-clearing fires	s^{-1} or a^{-1}

continued...

Symbol	Symbol Description	Unit
F_r^i	fractional rate of removal of nuclide i from soil by root uptake	a^{-1}
FA	floor area of residential buildings	m^2
Fec	fat fuel value	$kJ \cdot g^{-1}$ fat
fd	fluid density	$kg \cdot m^{-3}$
ff	frequency of agricultural fires	s^{-1} or a^{-1}
Fl^i	mass of nuclide i transferred per unit time from one compartment to another	$mol \cdot a^{-1}$
Fmw	fat metabolic water yield	m^3 water$\cdot g^{-1}$ fat
Fo	fat STP oxygen combustion value	m^3 $O_2 \cdot g^{-1}$ fat or L $O_2 \cdot g^{-1}$ fat
fs	percentage of simulations used to calculate SENSYV factors	unitless
FUELUS	amount of energy required to heat a family home	$MJ \cdot s^{-1}$ or $MJ \cdot a^{-1}$
Fw_{ds}	volumetric flow of water out of the compacted sediment layer	m^3 water$\cdot a^{-1}$
Fw_{ob}	volumetric flow of water out of the overburden	m^3 water$\cdot a^{-1}$
Fw_r	flux of water out of the root zone	m^3 water$\cdot m^{-2}$ soil$\cdot d^{-1}$
FY	forest yield or tree mass consumed in a forest fire	kg wet biomass$\cdot m^{-2}$ land

continued...

Symbol	Symbol Description	Unit
FY_j	yield of animal food types j = TE MEAT, TE MILK and TE BIRD	kg wet biomass·a^{-1}
Fym_j	fat content of food type j	g fat·kg^{-1} wet biomass
g	gravitational acceleration constant	m·s^{-2}
gb	mass/activity conversion factor for ^{129}I	kg iodine·Bq^{-1}
gc	mass/activity conversion factor for ^{14}C	kg carbon·Bq^{-1}
GM	geometric mean	unitless
GM_L	geometric mean dose predicted in the lower percentage, fs, of simulations in a given case or set of simulations	Sv·a^{-1}
GM_U	geometric mean dose predicted in the upper percentage, fs, of simulations in a given case or set of simulations	Sv·a^{-1}
GSD	geometric standard deviation	unitless
h	hydraulic head	m
H_{50}	50-year committed effective dose equivalent	Sv
H_c	consequence-frequency histogram based on SYVAC3 simulations in which parameter values remain constant in time	unitless
H_E	effective dose equivalent	Sv

continued...

Symbol	Symbol Description	Unit
h_G, l_G, m_G, q_G, r_G, s_G	regression constants used to calculate the time to steady-state root-zone soil concentration in the groundwater case	unitless
H_T	dose equivalent	Sv
H_v	consequence-frequency histogram based on SYVAC3 simulations in which parameter values vary in time	unitless
Hs	soil ingestion rate from hands	kg dry soil$\cdot$a^{-1} or mg dry soil$\cdot$a^{-1}
i	nuclide	unitless
I	rate at which a nuclide is introduced into a system	mol$\cdot$a^{-1}
I_A	man's inhalation rate	m^3 air$\cdot$a^{-1}
I_N^i	nuclide inventory per unit area of the soil root zone	mol$\cdot$m^{-2} soil
I_o	an instantaneous input of radioactivity to the human body	Bq
I_R	ratio of ^{129}I to total (^{129}I plus stable) iodine ingested by man	unitless
I_R^G	ratio of ^{129}I to total (^{129}I plus stable) iodine in groundwater	unitless
I_W	amount of irrigation water applied	m^3 water$\cdot$d^{-1}, m water$\cdot$a^{-1} or m^3 water$\cdot$m^{-2} soil$\cdot$a^{-1}

continued...

Symbol	Symbol Description	Unit
Ic	dissolved inorganic carbon inventory in the lake	mol·m^{-2} lake surface
IGS	indoor atmospheric pathway in which gases are suspended from soil	unitless
IGW	indoor atmospheric pathway in which gases are suspended from water	unitless
INDRN	indoor radon transfer coefficient	(mol ^{222}Rn·m^{-3} air)/ (mol ^{226}Ra·kg^{-1} dry soil)
INFILT	building infiltration rate	s^{-1} or h^{-1}
Inv^i	mass of nuclide i in a source compartment	mol
IRP	food-chain pathway involving transfer of a daughter nuclide (i - 1), with a half-life between 1 d and 20 a, from soil to plant to man	unitless
j	food type or environmental compartment	unitless
k	pathway	unitless
K	hydraulic conductivity	m water·s^{-1}
K_1^1	hydraulic conductivity of soil layer 1 at time step 1	m water·s^{-1}
K_2^1	hydraulic conductivity of soil layer 2 at time step 1	m water·s^{-1}
K_p	permeability of a porous medium	m^2

continued...

Symbol	Symbol Description	Unit
Kd^i	soil solid/liquid partition coefficient for nuclide i	m^3 water·kg^{-1} dry soil or L water·kg^{-1} dry soil
Kd^i_{ds}	compacted sediment solid/liquid partition coefficient for nuclide i	m^3 water·kg^{-1} dry sediment
KK	building wake entrainment parameter	unitless
LD0, LD1, etc.	designators for low-dipping fracture zones at the Whiteshell Research Area	unitless
LF	atmospheric pathway in which nuclides are suspended via forest and land-clearing fires	unitless
LP	food-chain pathway involving nuclide transfer from air to plants to man (leaf pathway)	unitless
LPA	food-chain pathway involving nuclide transfer from air to plants to terrestrial animals to man	unitless
LW	probability of well water use or lake/well-water switch	unitless
M	nuclide mass	mol
m	number of nuclides	unitless
M^i	mass of nuclide i in a given volume or layer	mol
M^i_{air}	mass of nuclide i at time t in the atmosphere compartment	mol
M^i_{anim}	mass of nuclide i at time t in the terrestrial animals	mol

continued..

Symbol	Symbol Description	Unit
M^i_{bmat}	mass of nuclide i at time t in the building materials	mol
M^i_{fish}	mass of nuclide i at time t in the fish inhabiting the lake	mol
M^i_I	mass of nuclide i added to the root zone with irrigation water	mol
M^i_{in}	mass of nuclide i entering the biosphere up to time t, adjusted for radioactive decay and ingrowth for chain nuclides	mol
M^i_{lake}	mass of nuclide i at time t in the lake water	mol
M^i_{man}	mass of nuclide i at time t in the bodies of the members of the critical group	mol
M^i_{out}	mass of nuclide i flowing out of the biosphere up to time t	mol
M^i_p	mass of nuclide i at time t in the biosphere	mol
M^i_{pla}	mass of nuclide i at time t in the plants	mol
$(M^i_{pla,e})$	mass of nuclide i at time t in plant crops in field e from root uptake	mol
M^i_{sed}	mass of nuclide i at time t in the mixed sediments of the lake	mol
$\hat{M}^i_{sed}$	mass of nuclide i at time t in the sediments accumulated over a period, t_{sed}	mol

continued...

Symbol	Symbol Description	Unit
M^{i}_{soil}	mass of nuclide i at time t in the soil compartment	mol
M^{i}_{T}	total mass of nuclide i in a given volume or layer	mol
M^{i}_{well}	mass of nuclide i in the well water	mol
MC^{H}	average hydrogen concentration in man's body	g hydrogen·kg^{-1} soft tissue
Mw^{i}	molecular weight of nuclide i	kg·mol^{-1}
n	number of food types	unitless
N	product of all other modifying factors used to define the dose equivalent	unitless
N_{A}	Avogadro's number	6.02×10^{23} atoms·mol^{-1}
n_{ID}, q_{ID}, r_{ID}, s_{ID}, t_{ID}, u_{ID}, v_{ID}, w_{ID}, x_{ID}, y_{ID}	regression constants used to calculate the time to steady-state root-zone soil concentration in the irrigation/deposition case	unitless
Na_{j}	number of animals of food types j = TE MEAT, TE MILK and TE BIRD raised by the critical group	unitless
Nph	number of people per household	p or unitless
oac	man's air/oxygen conversion factor	unitless
Ob	man's building occupancy factor	unitless

continued...

Symbol	Symbol Description	Unit
Oe	man's water occupancy factor	unitless
Og	man's ground occupancy factor	unitless
ov	man's oxygen utilization factor	unitless
p	gauge pressure	$kg \cdot m^{-1} \cdot s^{-2}$
p_{ds}	porosity of compacted sediments	unitless
P	precipitation rate	$m\ water \cdot d^{-1}$ or $m\ water \cdot a^{-1}$
$(P)_{as}$	annual precipitation at a representative site on the Canadian Shield	$m\ water \cdot a^{-1}$
$(P)_{ds}$	daily precipitation at a representative site on the Canadian Shield	$m\ water \cdot d^{-1}$
P_I	amount of precipitation water intercepted by plants	$m\ water \cdot a^{-1}$
P_L	a parameter value two standard deviations below its mean value	unitless
P_P	amount of water that percolates down into soil	$m\ water \cdot a^{-1}$
P_U	a parameter value two standard deviations above its mean value	unitless
Pe	effective precipitation equal to the difference between precipitation and surface runoff	$m\ water \cdot a^{-1}$ or $m^3\ water \cdot m^{-2}\ soil \cdot a^{-1}$
$(Pe)_a$	annual effective precipitation	$m\ water \cdot a^{-1}$

continued...

Symbol	Symbol Description	Unit
$(Pe)_d$	daily effective precipitation	m water·d^{-1}
Pec	protein fuel value	kJ·g^{-1} protein
PI	probability of irrigation or irrigation switch for garden and forage field	unitless
Pmw	protein metabolic water yield	m^3 water·g^{-1} protein
Po	protein STP oxygen combustion value	m^3 O_2·g^{-1} protein or L O_2·g^{-1} protein
Pr	a parameter	unitless
Ps	mass of soil adhering to a unit mass of TE PLANT, after normal processing and preparation	kg dry soil·kg^{-1} wet biomass or mg dry soil·g^{-1} wet biomass
PS	probability of sediment use or sediment/soil switch	unitless
PT	probability of peat fuel use or peat/wood switch	unitless
PY	peat yield or mass of peat burned in forest or land-clearing fires	kg dry peat·m^{-2} land
Pym_j	protein content of food type j	g protein·kg^{-1} wet biomass
Q	quality factor for radiation type used to define the dose equivalent	unitless
Q^i	flux of nuclide i from a source compartment to the atmosphere	mol·m^{-2}·s^{-1} or mol·s^{-1}

continued...

Symbol	Symbol Description	Unit
Q_{cap}	well capacity	m^3 water$\cdot a^{-1}$
Q_{crt}	critical well demand	m^3 water$\cdot a^{-1}$
$(Q^i)_{AF}$	flux of nuclide i to the atmosphere via agricultural fires	$mol\cdot m^{-2}$ land$\cdot s^{-1}$
$(Q^i)_{AG}$	flux of nuclide i to the atmosphere via gaseous evasion from the lake surface	$mol\cdot m^{-2}$ water$\cdot a^{-1}$
$(Q^i)_{IGW}$	flux of nuclide i released to indoor air via domestic water	$mol\cdot a^{-1}$
$(Q^i)_k$	flux of nuclide i from a source compartment to the atmosphere via pathway k	$mol\cdot m^{-2}\cdot s^{-1}$ or $mol\cdot s^{-1}$
$(Q^i)_{LF}$	flux of nuclide i to the atmosphere via forest or land clearing fires	$mol\cdot m^{-2}$ land$\cdot s^{-1}$
$(Q^i)_{TG}$	flux of nuclide i to the atmosphere via gaseous evasion from soil	$mol\cdot m^{-2}$ soil$\cdot s^{-1}$
q^{Rn}	radon emission rate from soil	(mol $^{222}Rn\cdot m^{-2}$ soil$\cdot s^{-1}$)/ (mol $^{226}Ra\cdot kg^{-1}$ dry soil)
q_z	volume flux density of water in the z direction through the soil	$m\cdot s^{-1}$
Qdw_j	drinking water ingestion rate for animal food types j = TE MEAT, TE MILK and TE BIRD	m^3 water$\cdot d^{-1}$ or m^3 water$\cdot a^{-1}$
Qf_j	feed or forage ingestion rate for animal food types j = TE MEAT, TE MILK and TE BIRD	kg wet biomass$\cdot d^{-1}$ or kg wet biomass$\cdot a^{-1}$

continued...

Symbol	Symbol Description	Unit
Qs_j	soil ingestion rate for animal food types j = TE MEAT, TE MILK and TE BIRD	kg dry soil·d^{-1} or kg dry soil·a^{-1}
r	linear product moment correlation coefficient	unitless
$r_{e,D}$	plant interception fraction for atmospheric deposition for field e	unitless
$r_{e,I}$	plant interception fraction for irrigation deposition for field e	unitless
r_j	plant interception fraction for food type j	unitless
R	runoff	m water·a^{-1}
R_s	surface runoff	m water·a^{-1}
rb	plant interception fraction for wooden building materials	unitless
RE	retardation factor for GEONET segments interfacing with the biosphere	unitless
RE^d	retardation factor for a daughter radionuclide	unitless
RE^p	retardation factor for a precursor radionuclide	unitless
$RELFRAC^i$	fraction of the inventory of nuclide i in domestic water released to indoor air	unitless
$RF(t,t')$	impulse response function, which defines the fraction of nuclide remaining in a volume at time t following a unit impulse input at time t', where $t' \leq t$	unitless

continued.

Symbol	Symbol Description	Unit
RF^i_4	impulse response function for nuclide i for the bottom or fourth soil layer	unitless
RF^i_l	impulse response function for nuclide i for the lake water compartment	unitless
$(RF^i_s(t))_G$	time-dependent impulse response function for nuclide i for the soil root zone in the groundwater contamination case	unitless
$(RF^i_s(t))_{ID}$	time-dependent impulse response function for nuclide i for the soil root zone in the irrigation/deposition case	unitless
RF^i_{sed}	impulse response function for nuclide i for the mixed sediment compartment	unitless
RP	food-chain pathway involving nuclide transfer from soil to plants to man (root pathway)	unitless
RPA	food-chain pathway involving nuclide transfer from soil to plants to terrestrial animals to man	unitless
S	food-chain pathway involving nuclide transfer from man's soil ingestion	unitless
S_{sed}	rate at which suspended material in the lake water is deposited to the mixed sediment	kg dry sediment$\cdot m^{-2} \cdot a^{-1}$
SA	food-chain pathway involving nuclide transfer from soil to terrestrial animals to man	unitless
sbc	inorganic building material/soil conversion factor	unitless

continued...

Symbol	Symbol Description	Unit
Sf	SENSYV factor	unitless
SM	total mass of mixed sediment	kg dry sediment
t	time	s, d, a
t_{irr}	irrigation period	a
t_p	arbitrary point in time	unitless
t_{sed}	time (a) required to accumulate mixed sediments to depth Z_{sed}	m
T	total simulation time	unitless
TC	transfer coefficient that predicts the nuclide concentration in the components of the food chain between a source compartment, and man and other biota	unitless
te_e	time of above-ground exposure for plant crops in field e	d
te_j	time of above-ground exposure for terrestrial food types j during the growing season	d
teb	time of above-ground exposure for wooden building material	d
TG	atmospheric pathway in which gases are suspended from terrestrial sources	unitless
th	holdup time for TE PLANT	d

continued...

Symbol	Symbol Description	Unit
thb	holdup time for inorganic and wooden building materials	d
thb_{ibm}	holdup time for inorganic building material	d
thb_{wbm}	holdup time for wooden building material	d
thdw	holdup time for man's drinking water	d
thf_j	terrestrial animal feed holdup time for food types j = TE MEAT, TE MILK and TE BIRD	d
Thi	iodine content of man's thyroid gland	kg iodine or mg iodine
Thm	mass of man's thyroid gland	kg thyroid or g thyroid
thp	holdup time for FW FISH	d
ths_j	terrestrial animal soil holdup time for food types j = TE MEAT, TE MILK and TE BIRD	d
thw_j	terrestrial animal drinking water holdup time for food types j = TE MEAT, TE MILK and TE BIRD	d
tp	plant environmental halftime	d
TP	atmospheric pathway in which particles are suspended from terrestrial sources	unitless
$(tr^i)_G$	time scale for nuclide i related to the time, $(tss)^i_G$, at which steady-state root-zone soil concentrations are achieved in the groundwater case	a

continued...

Symbol	Symbol Description	Unit
$(tr^{i})_{ID}$	time scale for nuclide i related to the time, $(tss)^{i}_{ID}$, at which steady-state root-zone soil concentrations are achieved in the irrigation/deposition case	a
$(tss)^{i}$	time at which steady-state root-zone soil concentrations are achieved for nuclide i	a
$(tss)^{i}_{G}$	time at which steady-state root-zone soil concentrations are achieved for nuclide i in the groundwater case	a
$(tss)^{i}_{I}$	time at which steady-state root-zone soil concentrations are achieved for nuclide i in the irrigation case	a
$(tss)^{i}_{ID}$	time at which steady-state root-zone soil concentrations are achieved for nuclide i in the irrigation/deposition case	a
U	use factor that describes man's utilization rate of a given environmental component	unitless
U_{j}	man's ingestion rate of food type j	kg wet biomass·a^{-1}
U^{i}_{N}	annual mass of nuclide i taken up per unit soil area by plant crop	mol·m^{-2} soil·a^{-1}
UCAV	ambient or annual average wind speed at a generic Canadian Shield site	m·s^{-1}
Udw	man's ingestion rate of drinking water	m^{3} water·a^{-1}
Us	man's ingestion rate of soil	kg dry soil·a^{-1}

continued...

Symbol	Symbol Description	Unit
Uwc	annual water demand per person	m^3 water·a^{-1}·p^{-1}
UWGHT	wind speed weighting factor	unitless
v	rate of flow of a fluid or gas through a material	m·s^{-1}
V	total volume of shallov soil layer	m^3 soil
V_l	volume of water in the lake	m^3 water
V_{ld}	volume of water from the lake drawn into the bedrock well	m^3 water·a^{-1}
V_I	volume of irrigation water applied to the soil	m^3 water·m^{-2} soil·d^{-1}
V_{TD}	volume of the bottom soil layer subject to terrestrial discharge	m^3 soil
V0, V1, etc.	designators for vertical fracture zones at the Whiteshell Research Area	unitless
Vd	dry deposition velocity	m·s^{-1}, m·d^{-1} or m·a^{-1}
vi	viscosity of a fluid or gas	N·s·m^{-2}
W	food-chain pathway involving nuclide transfer from man's drinking water	unitless
W_D	domestic water demand of the household	m^3 water·a^{-1}
W_F	irrigation demand for the forage field	m^3 water·a^{-1}
W_L	water demand for livestock	m^3 water·a^{-1}

continued...

Symbol	Symbol Description	Unit
W_T	weighting factor for the relative radiation risk of target organs of man used to define the effective dose equivalent	unitless
W_v	irrigation demand for the garden	m^3 water·a^{-1}
W_w	well demand	m^3 water·a^{-1}
WA	food-chain pathway involving nuclide transfer from water to terrestrial animals to man	unitless
wdw	wet/dry wood conversion factor	kg wet biomass·kg^{-1} dry wood
WF	food-chain pathway involving nuclide transfer from water to fish to man	unitless
Wr	washout ratio	unitless
x	downwind distance	m
x_c	a parameter that is correlated with a second, independently-sampled parameter	unitless
x_i	value of an independently sampled parameter	unitless
Y_e	plant yield for field e	kg wet biomass·m^{-2} soil or land
Y_j	yield of food types j = TE PLANT, TE MEAT, TE MILK and TE BIRD	kg wet biomass·m^{-2} soil or kg wet biomass·m^{-2} soil·a^{-1}
Yb	yield for trees used to make wooden building materials	kg wet biomass·m^{-2} land
Ycf_j	energy weighting factor for food type j	unitless

continued...

Symbol	Symbol Description	Unit
$Ycfs_j$	energy fraction for food type j	unitless
Ywc_j	water content of food type j	m^3 water·kg^{-1} wet biomass
z	elevation of a space coordinate measured vertically upward from the ground surface	m
z_o	surface roughness height	m
Z	nominal depth of a sediment or soil layer	m
Z_4	depth of the bottom or fourth soil layer	m
Z_j	depth of soil layer j	m
Z_l	mean depth of the lake	m
Z_r	depth of the soil root zone	m
Z_s	soil or peat depth	m
Z_{sed}	depth of the mixed sediment layer	m
α^i	rate constant for the net rate of transfer of nuclide i from lake water to mixed sediment	a^{-1}
β	fractional transfer rate due to all the loss mechanisms for a general compartment	a^{-1}
β_l^i	loss rate of nuclide i from lake water taking all the loss mechanisms into account	a^{-1}
$(\beta_s^i)_G$	derived quantity for the groundwater case defined in Equation (6.49)	unitless

continued...

Symbol	Symbol Description	Unit
$(\beta_s^i)_{ID}$	derived quantity for the irrigation/deposition case defined in Equation (6.52)	unitless
γ^i	derived quantity defined in Equation (4.11)	unitless
Γ	random number from a normal distribution with an arithmetic mean of 0 and standard deviation of 1	unitless
δ	terrestrial fraction of the total area of a discharge zone	unitless
Δt	time interval	d
ϵ	fraction of nuclide mass taken up by plants through their roots that is permanently lost from the soil	unitless
ζ^i	rate constant for cropping losses	a^{-1}
η_1^i	rate constant for the loss of a volatile nuclide i to the atmosphere by gaseous evasion from the lake surface	s^{-1} or a^{-1}
η_s^i	rate constant for the loss of a volatile nuclide i to the atmosphere by gaseous evasion from the soil	s^{-1} or a^{-1}
θ	volumetric water content of the soil	m^3 water$\cdot m^{-3}$ soil
θ_1^1	volumetric water content of soil layer 1 at time step 1	m^3 water$\cdot m^{-3}$ soil

continued...

Symbol	Symbol Description	Unit
θ_2^1	volumetric water content of soil layer 2 at time step 1	m^3 water·m^3 soil
θ_{FC}	volumetric water content of the soil at field capacity	m^3 water·m^{-3} soil
θ_j	volumetric water content of soil layer j	m^3 water·m^{-3} soil
θ_j^0	volumetric water content of soil layer j at time t = 0	m^3 water·m^{-3} soil
κ	conversion factor equal to 10^{-3} $m^3 \cdot L^{-1}$ water	unitless
λ^d	radioactive decay constant of a daughter radionuclide	a^{-1}
λ^i	radioactive decay constant of nuclide i	s^{-1}, d^{-1} or a^{-1}
λ^p	radioactive decay constant of a precursor radionuclide	a^{-1}
λ_E^i	effective removal constant of nuclide i from vegetation	d^{-1}
λg	effective removal constant of nuclide from soil, used in the preclosure biosphere model, PREAC	a
μ	arithmetic mean	unitless
μ_{10}	arithmetic mean of a $\log_{10}$-transformed distribution	unitless
μ_c	arithmetic mean of a correlated parameter	unitless

continued...

Symbol	Symbol Description	Unit
μ_e	arithmetic mean of a $\log_e$-transformed distribution	unitless
μ_i	arithmetic mean of an independently sampled parameter	unitless
ξ	empirical quantity related to the applicability of Equation (6.31)	unitless
π	pi = 3.1416	unitless
ρ_{ds}	bulk density of the compacted sediments	kg dry sediment$\cdot$m^{-3} sediment
ρ_s	bulk density of soil or peat	kg dry soil$\cdot$m^{-3} soil or kg dry peat$\cdot$m^{-3} peat
ρ_{sed}	bulk density of mixed sediments	kg dry sediment$\cdot$m^{-3} sediment
σ	standard deviation of a distribution	unitless
σ_{10}	standard deviation of a $\log_{10}$-transformed distribution	unitless
σ_c	standard deviation of a correlated parameter	unitless
σ_e	standard deviation of a $\log_e$-transformed distribution	unitless
σ_i	standard deviation of an independent parameter	unitless
τ	time scale of fluctuations in a parameter	unitless
τ_s	time interval of SCEMR1 output	a

continued...

Symbol	Symbol Description	Unit
υ	small nuclide concentration increment used to determine when steady-state soil concentrations are achieved	mol·kg^{-1} dry soil
ϕ^i	rate constant for the transfer of nuclide i from one compartment to another	a^{-1}
χ^d	mass flow rate out of the geosphere for a daughter radionuclide	mol·a^{-1}
χ^i	mass flow rate of nuclide i from compacted sediments to the lake	mol·a^{-1}
χ^p	mass flow rate out of the geosphere for a precursor radionuclide	mol·a^{-1}
χ^i_{ds}	mass flow rate of nuclide i out of the compacted sediment layer	mol·a^{-1}
χ^i_{ob}	mass flow rate of nuclide i out of the overburden	mol·a^{-1}
χ^i_w	mass flow rate of nuclide i out of the geosphere into the well	mol·a^{-1}
ψ	total potential of soil water	m water
ψ^1_1	total water potential of soil layer 1 at time step 1	m
ψ^0_2	total water potential of soil layer 2 at $t = 0$	m
ψ^1_2	total water potential of soil layer 2 at time step 1	m

continued...

Symbol	Symbol Description	Unit
ψ_g	gravitational potential of the soil water	m water
ψ^0_{gj}	gravitational potential of soil layer j at t = 0	m water
ψ^1_{g1}	gravitational potential of soil layer 1 at time step 1	m water
ψ^0_j	total water potential for soil layer j at t = 0	m water
ψ_m	matric potential of soil water	m water
ψ^1_{m1}	matric potential of soil layer 1 at time step 1	m water
ψ^0_{mj}	matric potential of soil layer j at t = 0	m water

APPENDIX C

FACTORS AND ISSUES CONSIDERED IN DEVELOPING BIOTRAC

CONTENTS

C.1 INTRODUCTION

An important first step in assessing the potential postclosure impacts of a nuclear fuel waste facility on the environment and humans is the identification of the scenarios to be evaluated (Section 1.5.1). A scenario is defined as a combination of factors (features, events or processes) that could affect the ability of the disposal facility to immobilize and isolate the nuclear fuel waste (Goodwin et al., in preparation). For the biosphere, features might include sedimentation in water bodies and the composition of the human diet. Events could include earth-moving projects, and forest or grass fires. Processes could be biogas production and meteoric precipitation. All these factors could influence the transport of nuclides released from the geosphere to the biosphere, and the resulting impacts on the environment and humans.

Several studies have been conducted on scenario analysis for the Canadian Nuclear Fuel Waste Management Program (NFWMP). Merrett and Gillespie (1983) described disruptive events that are important and may be initiated by humans, by vault-related processes, or by natural phenomena. Their analysis defined events that are likely unimportant and events that may require further consideration. Heinrich (1984) documented the results of a workshop dealing with long-term stability of the geosphere. One major conclusion of the workshop was that the effect of potential perturbations should be examined in terms of their influence on the mechanical and thermal stability of the vault. Davis (1986) described time-dependent processes that could affect the biosphere. His analysis included a recommended approach to modelling those processes that are expected to be most important. This approach was largely adopted in developing BIOTRAC (Section 1.5). Finally, Goodwin et al. (in preparation) carried out an extensive scenario analysis to select scenarios for the postclosure assessment of the disposal concept used by Canada.

C.2 NEED FOR SCENARIO ANALYSIS

Scenario analysis is a procedure for identifying and describing all the factors that must be considered in the assessment process. In the postclosure assessment it has two purposes. Firstly, it provides a comprehensive list of factors that could affect the performance of the disposal facility and a systematic procedure for listing the potentially important factors. Secondly, it provides a logical structure within which the importance of each factor can be evaluated, and a framework for grouping factors into scenarios and identifying those scenarios that require quantitative assessment.

C.3 SYSTEMATIC PROCEDURE FOR SCENARIO ANALYSIS

Goodwin et al. (in preparation) described our systematic scenario analysis procedure, which is a refinement of a procedure originally developed at the Sandia National Laboratories (Cranwell et al. 1987). The procedure consists of six steps: the first three steps are designed to generate a comprehensive list of factors that could influence the performance of a disposal facility; the last three steps identify important scenarios that must be evaluated quantitatively. The six steps are:

1. List factors. A factor is any feature, event or process that could influence the behaviour of any component of the disposal facility and its surroundings. The objective of this step is to generate, via a series of brainstorming sessions, a comprehensive list of factors without constraints on the types of factors that are suggested. At this stage other related studies may be reviewed as a source of factors.

2. Classify the factors. Confidence is developed that the list of factors is comprehensive by organizing and ordering them in various ways. The objective is to identify factors missed in the first step and not to devise an all-purpose classification. Related factors may be combined.

3. Screen the factors. All the factors are critically reviewed in detail and classified as to how they should be treated in the remaining steps. Factors may be deemed sufficiently important to require quantitative evaluation, or sufficiently unimportant that they require only qualitative evaluation. Only the first set of factors is passed on to step 4. In all cases, the reasons for the recommended treatment must be documented.

4. Construct scenarios. The factors requiring quantitative evaluation are combined systematically to yield a comprehensive set of candidate scenarios. A central group of scenarios may be constructed first to make this step more tractable (Section 1.5.1). The central group of scenarios contains as many of the important factors as possible. As a general rule, these factors are expected to always be important, to occur frequently, or to apply to a significant degree over the time scale of the assessment. A factor may be excluded from the central group of scenarios for several reasons; for example, it may be important only rarely or under unusual conditions, or its presence may be incompatible with the presence of another factor. The residual factors that do not appear in the central group of scenarios are grouped in combinations to construct alternative scenarios. Each alternative scenario contains a unique combination of one or more of the residual factors, plus all the compatible factors from the central group of scenarios. Alternative scenarios may differ from each other and from the central group of scenarios by one or more factors.

5. Screen the scenarios. The candidate scenarios are examined in detail to select those that must be considered for quantitative assessment. By definition the central group of scenarios must be assessed. An alternative scenario may be eliminated if it can be combined with another scenario or if a bounding analysis shows that it cannot have a significant effect on meeting regulatory criteria.

6. Define the scenarios. Each scenario that has been retained for quantitative assessment is defined in detail. Any special issues are clarified and resolved. Finally, a probability of occurrence is assigned to each scenario and the reasons for the selected values are documented.

C.4 APPLICATION OF PROCEDURES

The scenario analysis for the postclosure assessment was performed by a group of experts having diverse expertise and experiences. Lists were compiled of the factors that could influence the vault, the geosphere and the biosphere. Altogether, a total of over 1000 factors were initially identified. This number was then greatly reduced because many of the factors were related and could be combined. Of the final list of about 280 factors, about half were deemed to require quantitative assessment through the central group of scenarios and alternative scenarios, and the remainder were selected for qualitative treatment. It was eventually found that all but one of the factors designated for quantitative treatment could be combined into the central group of scenarios for quantitative analysis in the postclosure assessment (Section 1.5.1). This left only one alternative scenario concerned with open or unsealed boreholes. The effects of continental glaciation on the biosphere and on nuclide transport are treated outside the scenario analysis, as discussed in Chapter 12.

The biosphere factors considered in the scenario analysis are listed alphabetically in Table C-1. The table indicates whether a factor is part of the central group of scenarios, has been treated qualitatively or given alternative treatment, and whether it is documented in this report, a submodel report or another report (Figure 1-7) . Many of the factors are also discussed in various other supporting documents. All the factors considered in the scenario analysis are more fully discussed by Goodwin et al. (in preparation).

TABLE C-1

LIST OF BIOSPHERE FACTORS CONSIDERED IN SCENARIO DEVELOPMENT

	Factor	Treatment*	Reference**
1	Acid rain	C	SW
2	Alkali flats	Q	S
3	Animal grooming and fighting	C	F
4	Animal soil ingestion	C	F
5	Animal diets	C	F
6	Artificial lake mixing	C	W
7	Ashes and sewage sludge fertilizers	C	F
8	Bacteria and microbes in soil	C	FS
9	Bioconcentration	C	BF
10	Biogas production	Q	A
11	Biological evolution	Q	BF
12	Biotoxicity	C	BFT
13	Bioturbation of soil and sediment	C	SW
14	Building materials	C	BF
15	Burrowing animals	C	FS
16	Capillary rise in soil	C	BS
17	Carcasses	C	F
18	Carcinogenic contaminants	C	BFPT
19	Charcoal production	C	AB
20	Chemical precipitation	C	BSW
21	Chemical toxicity	C	BFP
22	Climate	C	ABFSW
23	Climate change	AC	BT
24	Collisions, explosions and impacts	Q	X
25	Colloids	C	SW
26	Convection, turbulence and diffusion (atmospheric)	C	ABT AB
27	Correlation	C	ABFSW
28	Critical group - agricultural labour	C	ABF
29	Critical group - clothing and home furnishings	Q	BF
30	Critical group - evolution	Q	BF
31	Critical group - house location	C	BF
32	Critical group - individuality	C	BF
33	Critical group - leisure pursuits	C	BF
34	Critical group - pets	Q	F
35	Crop fertilizers and soil conditioners	C	F
36	Crop storage	C	BF
37	Cure for cancer	Q	X
38	Deposition (wet and dry)	C	BAFS

continued...

TABLE C-1 (continued)

Factor	Treatment*	Reference**
39 Dermal sorption - nuclides other than tritium	Q	BF
40 Dermal sorption - tritium	C	BF
41 Dispersion	C	ABFSW
42 Dust storms and desertification	C	ABS
43 Earthmoving projects	Q	S
44 Earthquakes	Q	X
45 Erosion - lateral transport	Q	ABS
46 Erosion - wind	Q	ABS
47 Fires - agricultural	C	AB
48 Fires - forest and grass	C	AB
49 Fish farming	Q	BF
50 Flipping of earth's magnetic poles	Q	X
51 Flooding	Q	B
52 Flushing of water bodies	C	BW
53 Food preparation	C	BF
54 Game ranching	C	BF
55 Gas leakage into basements	C	BA
56 Glaciation	A	BT
57 Greenhouse food production	C	BF
58 Greenhouse effect	Q	AB
59 Groundshine (ground exposure)	C	BF
60 Heat storage in lakes or underground	Q	X
61 Herbicides, pesticides and fungicides	C	F
62 Household dust and fumes	C	BA
63 Houseplants	Q	X
64 Human diet	C	BF
65 Human soil ingestion	C	BF
66 Hydroponics	Q	F
67 Industrial water use	Q	X
68 Intake of drugs	Q	X
69 Intrusion - deliberate	Q	BT
70 Intrusion - inadvertent	A	I
71 Ion exchange in soil	C	BS
72 Irrigation	C	BSF
73 Lake infilling	C	BS
74 Mutagenic contaminants	C	BFP
75 Outdoor spraying of water	C	ABFS
76 Ozone layer failure	Q	X
77 Peat and leaf litter harvesting	C	S
78 Plant roots	C	BFS
79 Precipitation (meteoric)	C	BAS
80 Radioactive decay	C	ABFSW
81 Radiotoxic contaminants	C	BFPT
82 Radon emission	C	ABF

continued...

TABLE C-1 (concluded)

	Factor	Treatment*	Reference**
83	River-course meander	C	B
84	Runoff	C	BSW
85	Saltation	C	BA
86	Scavengers and predators	C	BF
87	Seasons	C	ABFSW
88	Sediment resuspension in water bodies	C	BW
89	Sedimentation in water bodies	C	BW
90	Sensitization to radiation	C	BF
91	Showers and humidifiers	C	ABF
92	Smoking	Q	F
93	Soil	C	BFS
94	Soil depth	C	BS
95	Soil leaching	C	BS
96	Soil porewater pH	C	S
97	Soil sorption	C	BS
98	Soil type	C	BS
99	Space heating	C	AB
100	Surface water bodies	C	BW
101	Surface water pH	C	W
102	Suspension in air	C	ABFS
103	Technological advances in food production	Q	BF
104	Teratogenic contaminants	C	BFPT
105	Terrestrial surface	C	ABFS
106	Toxicity of mined rock	Q	X
107	Tree sap	C	F
108	Uncertainties	C	ABFSWP
109	Urbanization on the discharge site	Q	X
110	Water leaking into basements	Q	F
111	Water management projects	Q	W
112	Water source	C	BFSW
113	Wetlands	C	ABFSW
114	Wind	C	AB

* A - alternative treatment
C - central group of scenarios
Q - qualitative treatment
** A - atmospheric submodel (Amiro 1992)
B - biosphere model (this EIS primary reference)
F - food-chain and dose submodel (Zach and Sheppard 1992)
I - intrusion analysis (Wuschke 1992)
P - postclosure assessment (Goodwin et al. 1994)
S - soil submodel (Sheppard 1992)
W - surface-water submodel (Bird et al. 1992)
T - topical reports
X - scenario analysis report

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* Unrestricted, unpublished report available from SDDO, AECL Research, Chalk River, Ontario, K0J 1J0.

APPENDIX D

SAMPLE BIOTRAC CALCULATION

CONTENTS

D.1 INTRODUCTION

We present here step-by-step results from a typical BIOTRAC simulation to illustrate how the model works (Section 9.2) and to put the various exposure pathways into perspective. We take ^{99}Tc as our example radionuclide, for a number of reasons. It is a potentially important nuclide (Section 10.3.2.1), and is simple to deal with because it has no precursor. It has no special attributes, as is the case for ^{14}C and ^{129}I (Section 2.5) and so is modelled in the same way as the vast majority of nuclides in the vault inventory (Table 1-1). The radioactive decay constant for ^{99}Tc is $\lambda^{Tc} = 3.25 \times 10^{-6}\ a^{-1}$.

The results were obtained from a simulation of the model in which normally distributed parameters were assigned their mean or median values, and log-normally distributed parameters their geometric mean (GM) values. We assume that the nuclide flows and concentrations needed to drive BIOTRAC are constant in time. This allows the lake water, sediment and soil concentrations to be calculated using the simple analytical expressions derived in Sections 5.3.3 and 6.3.6. The analytical solutions are much easier to follow than the full solutions. Moreover, the assumption of a constant source term is not unreasonable, since the time required for the biosphere to reach steady state is generally much shorter than the time scale of variation in the nuclide flow out of the geosphere (Figure 10-1, Goodwin et al. 1994). The inputs used to drive BIOTRAC in our example simulation correspond to the values at an arbitrary point in the time series of flows and concentrations predicted in a realistic simulation of the vault and geosphere models. We will use these inputs to calculate steady-state environmental concentrations and doses to man.

D.2 CHARACTERISTICS AND BEHAVIOUR OF THE CRITICAL GROUP

We consider a three-member household (Nph = 3) to constitute the critical group (Section 9.1.1.1). Water supplies for the household come from a bedrock well (Section 9.1.2), and heating fuel from a woodlot (Section 7.5.2.7). The soil in the area is sandy (Section 6.5.1.1), and not conditioned with lake sediments (Section 6.3.7.3). The household garden is watered, but the forage field is not irrigated (Section 6.5.5.2).

The first step in the calculation is to use Equations (8.42) to (8.45) to estimate the food and water ingestion rates, and the inhalation rate, of an individual household member. The parameter values used in this calculation are the mean values discussed in Sections 8.5.6 and 8.5.12, and are summarized in Table D-1. The calculated rates are given in Table D-2.

The ingestion rates, Qf_j, for the three terrestrial animal food types are used in Equation (9.1) to calculate the number of each animal type needed by the household. With $FY_j = 4.6 \times 10^3$ L milk$\cdot a^{-1}$, 1.45×10^2 kg wet biomass$\cdot a^{-1}$ and 2.03 kg wet biomass$\cdot a^{-1}$ for j = TE MILK, TE MEAT and TE BIRD respectively (Section 9.1.1.2), we find that the household requires one dairy cow, three beef cattle and 79 chickens to meet its dietary requirements for a year. The size of the forage field needed to raise

TABLE D-1

PARAMETER VALUES USED TO CALCULATE FOOD AND WATER INGESTION, AND INHALATION RATES OF MAN

Parameter*	Units*	Food Type j				
		TE PLANT	TE MILK	TE MEAT	TE BIRD	FW FISH
Ycf_j	unitless	0.32	0.36	0.26	0.05	0.01
$Ycfs_j$	unitless	0.32	0.36	0.26	0.05	0.01
Cym_j	$g \cdot kg^{-1}$	168.8	32.0	5.4	3.2	0.0
Fym_j	$g \cdot kg^{-1}$	25.6	191.1	203.4	43.9	62.6
Pym_j	$g \cdot kg^{-1}$	49.4	113.9	170.0	197.8	177.6
Ywc_j	$m^3 \cdot kg^{-1}$	7.44×10^{-4}	6.44×10^{-4}	6.11×10^{-4}	7.45×10^{-4}	7.51×10^{-4}

En = 5.33×10^{6} $kJ \cdot a^{-1}$
ewc = 2.5×10^{-7} $m^3 \cdot kJ^{-1}$
oac = 4.78 (unitless)
ov = 6.25 (unitless)

Co = 8.2×10^{-4} $m^3 \cdot g^{-1}$
Fo = 2.03×10^{-3} $m^3 \cdot g^{-1}$
Po = 9.7×10^{-4} $m^3 \cdot g^{-1}$

Cec = 16.3 $kJ \cdot g^{-1}$
Fec = 37.7 $kJ \cdot g^{-1}$
Pec = 16.7 $kJ \cdot g^{-1}$

Cmw = 6.00×10^{-7} $m^3 \cdot g^{-1}$
Fmw = 1.07×10^{-6} $m^3 \cdot g^{-1}$
Pmw = 4.20×10^{-7} $m^3 \cdot g^{-1}$

* For parameter descriptions and full units see list of symbols in Appendix B.

TABLE D-2

MAN'S FOOD AND WATER INGESTION, AND INHALATION RATES CALCULATED USING THE PARAMETER VALUES LISTED IN TABLE D-1

Rate	Units	Value
Food Ingestion, U_j		
TE PLANT	kg wet biomass·a^{-1}	375.7
TE MILK	L milk·a^{-1}	199.4
TE MEAT	kg wet biomass·a^{-1}	130.9
TE BIRD	kg wet biomass·a^{-1}	53.2
FW FISH	kg wet biomass·a^{-1}	10.0
Water Ingestion, Udw	m^3 water·a^{-1}	0.641
Inhalation, I_A	m^3 air·a^{-1}	8617

these animals is calculated from Equation (9.4). Using the parameter values listed in Table D-3 (Section 8.5.5.1 and 8.5.8.1), we find that an area of 2.74×10^4 m^2 is required for the dairy cow, 5.47×10^4 m^2 for the beef cattle, and 9.61×10^3 m^2 for the poultry. The total area of the forage field, A_F, is therefore 9.17×10^4 m^2. Similarly, the size of the garden, A_V, is found from Equation (9.3) to be 1.41×10^3 m^2, assuming that the yield, Y_j, of TE PLANT is 0.8 kg wet biomass·m^{-2} soil. The wood required to heat the household can be grown sustainably on an area of $50 \cdot A_W$, where A_W is given by Equation (9.5). With FUELUS = 1.1×10^5 MJ·a^{-1}

TABLE D-3

PARAMETER VALUES USED TO CALCULATE THE AREA OF THE FORAGE FIELD

Parameter*	Units*	Animal Food Type j		
		TE MILK	TE MEAT	TE BIRD
Qf_j	kg·a^{-1}	2.19×10^4	1.82×10^4	146
Y_j	kg·m^{-2}	0.8	1.0	1.2

* For parameter descriptions and the full units see list of symbols in Appendix B.

(Section 7.5.2.1), $EW = 5.5$ MJ·kg^{-1} wet biomass (Section 7.5.2.2), and $Yb = 10.5$ kg wet biomass·m^{-2} land (Section 8.5.8.2), we determined that the sustainable area of the woodlot must be 9.52×10^4 m^2.

We next calculate the household demand for water. We assume that each person uses 130 m^3 water annually (Section 9.1.1.4) and so the domestic demand, W_D, is 390 m^3 water·a^{-1} (Equation (9.7)). The water demand for livestock, W_L, is found from Equation (9.8). We take $Qdw_j = 21.9$, 14.6 and 0.146 m^3 water·a^{-1} for TE MILK, TE MEAT and TE BIRD respectively (Section 8.5.5.2), and find that the livestock demand is 77.2 m^3 water·a^{-1}. The amount of water, I_W, required annually to irrigate one square metre of soil is calculated from Equation (6.1). We set $Z_s = 1.5$ m (Section 6.5.1.2) and calculate $Pe = 0.47$ m water·a^{-1} from Equation (6.54), assuming $P = 0.78$ m water·a^{-1} and $R = 0.31$ m water·a^{-1} (Section 9.1.3). Using the regression constants for sand listed in Table 6.1, we find $I_W = 0.616$ m water·a^{-1}. The total irrigation demand for a garden 1.41×10^3 m^2 in area is therefore 869 m^3 water·a^{-1}. Adding the domestic, livestock and irrigation demands, we calculated that the household uses a total of 1335 m^3 water a^{-1}. All the water needs of the household can be met by the well (Section 9.1.2). The total demand, W_W, is used to calculate well-water concentrations in Equation (4.18).

D.3 SOURCE TERMS

At the time of interest in our example simulation, the vault and geosphere models predict that the ^{99}Tc plume has entered the biosphere through the Boggy Creek south discharge zone (Figure 4.4). Apart from the bedrock well, this is the only active discharge zone, since the plume has not yet had time to reach the Boggy Creek north or Pinawa Channel discharge zones. The total area, A_{Dz}, of the south zone is predicted to be 2.85×10^5 m^2, and the annual volume of groundwater passing through it, Fw_{ds}, 2.24×10^3 m^3 water·a^{-1}. The flow rate of ^{99}Tc out of the overburden, χ_{ob}^{Tc}, is 7.73×10^{-12} mol·a^{-1}; the flow rate out of the compacted sediment layer, χ_{ds}^{Tc}, is 7.62×10^{-12} mol·a^{-1}; and the flow rate into the well, χ_{w}^{Tc}, is 7.17×10^{-12} mol·a^{-1}. The ^{99}Tc concentration in well water, C_{ww}^{Tc}, is predicted to be 5.39×10^{-15} mol·m^{-3} water (Equation 4.18).

Given these inputs from GEONET (Section 4.1), the interface models described in Section 4.4 can be used to calculate ^{99}Tc concentrations in compacted sediments and in the pore water of the bottom or fourth soil layer. Setting $Kd_{ds}^{Tc} = 10^{-3}$ m^3 water·kg^{-1} dry sediment, $P_{ds} = 0.457$ and $\rho_{ds} = 125$ kg·m^{-3} sediment (Chan et al. 1993), Equations (4.3) and (4.7) predict that the compacted sediment concentration, C_{ds}^{Tc}, equals 1.58×10^{-17} mol·kg^{-1} dry sediment. Since we assume that the flow rate out of the geosphere is constant, the pore-water concentration of the bottom soil layer, C_{pw}^{Tc}, can be evaluated using the simple relationship given in Equation (4.17). With $A_{Dz} = 2.85 \times 10^5$ m^2, $Z_4 = 0.2$ m (Section 6.1), $\theta = 0.12$ m^3 water·m^{-3} soil (Section 6.5.1.7), $\rho_s = 1500$ kg dry soil·m^{-3} soil (Section 6.5.1.3) and $Kd^{Tc} = 10^{-4}$ m^3 water·kg^{-1} dry soil (Section 6.5.3), we find $C_{pw}^{Tc} = 3.45 \times 10^{-15}$ mol·m^{-3} water for the bottom soil layer.

D.4 WATER AND SEDIMENT CONCENTRATIONS

Since the nuclide flow to the lake is assumed to be constant, we can use Equations (5.19) and (5.22) to calculate ^{99}Tc concentrations in lake water and mixed sediments respectively. The total nuclide flow to the lake is given by the sum of the flows to the well and to the south discharge zone, $\chi^{Tc} = 7.17 \times 10^{-12} + 7.62 \times 10^{-12} = 1.48 \times 10^{-11}$ mol·a^{-1}. The remaining parameters in Equations (5.19) and (5.22) have been assigned values equal to the means of their respective distributions (Section 5.5), as summarized in Table D-4. Note that the total loss rate from the water column, β_l^{Tc}, is given by Equation (5.16). These parameter values lead to a steady-state water concentration, C_l^{Tc}, of 4.43×10^{-19} mol·m^{-3} water, and a mixed-sediment concentration, C_{sed}^{Tc}, of 2.11×10^{-17} mol·kg^{-1} dry sediment. Given a mixed sediment depth, Z_{sed}, of 0.055 m (Section 5.5.6), the depth-weighted average ^{99}Tc concentration in the top 30 cm of the sediment profile, C_{se}^{Tc}, is 1.68×10^{-17} mol·kg^{-1} dry sediment (Equation (5.15)). Because the critical group does not access the sediments in our example simulation, the sediment concentrations are put to no further use in the model.

D.5 SOIL CONCENTRATIONS

Because the nuclide flow out of the geosphere is assumed to be time-independent, the soil concentrations for each contamination pathway can be calculated using the analytical equations derived in Section 6.3.6.

TABLE D-4

PARAMETER VALUES USED TO CALCULATE ^{99}Tc WATER AND MIXED-SEDIMENT CONCENTRATIONS

Parameter*	Units	Value
A_l	m^2	7.0×10^4
Z_l	m	4.6
A_d	m^2	1.06×10^8
R	m·a^{-1}	0.31
S_{sed}	kg·m^{-2}·a^{-1}	0.164
α^{Tc}	a^{-1}	1.7
η_l^{Tc}	a^{-1}	0.0

* For parameter descriptions and the full units see list of symbols in Appendix B.

D.5.1 TERRESTRIAL DISCHARGE

Soil concentrations resulting from the terrestrial discharge of ^{99}Tc are given by Equation (6.48). The pore-water concentrations driving the model in this simulation differ from field to field because the total required field area (Section D.2) exceeds the area of terrestrial discharge, A_{TD}. Only a fraction, $\delta = 0.055$, of the total discharge area is assumed to underlie arable unsaturated soils (Section 4.5.1). Since the area, A_{DZ}, of the south discharge zone is 2.85×10^5 m^2 (Section D.3), $A_{TD} = 1.57 \times 10^4$ m^2 (Equation (4.1)). In the absence of a peat bog, this area is assumed first to underlie the garden, as explained in Section 6.3.7.1. Since the garden, with an area, A_v, of 1.41×10^3 m^2 (Section D.2) can fit entirely into A_{TD}, the pore-water concentration in its bottom soil layer equals 3.45×10^{-15} mol·m^{-3} water (Section D.3). The area remaining from A_{TD} (1.43×10^4 m^2) fits entirely within the forage field. If we assume that the remainder of the field (7.74×10^4 m^2) is not subject to groundwater discharge, the area-weighted pore-water concentration in the lowest layer of the field is 5.38×10^{-16} mol·m^{-3} water. Since the entire terrestrial discharge area has now been used up, the woodlot is not subject to contamination by terrestrial discharge. These results are summarized in Table D-5.

The normalized steady-state concentration, $(Css)_G^{Tc}$, appearing in Equation (6.48) is evaluated using Equation (6.24) and the regression constants for a sand soil (Table 6.2). With Pe = 0.47 m water·a^{-1} (Section 6.5.2.2), $Z_s = 1.5$ m (Section 6.5.1.2) and $Kd^{Tc} = 0.10$ L water·kg^{-1} dry soil (Section 6.5.3), we obtain $(Css)_G^{Tc} = 0.629$ (mol·kg^{-1} dry soil)/(mol·L^{-1} water). Similarly, the time to steady state, $(tss)_G^{Tc}$, is found from Equation (6.25) and the appropriate regression constants from Table 6.3 to equal 1.05×10^4 a. The time scale $(tr^{Tc})_G$ can then be evaluated using Equation (6.31). For this simulation, $\upsilon = 10^{-4}$ mol·kg^{-1} dry soil and $\tau_s = 100$ a (Sheppard 1992), so that $(tr^{Tc})_G = 1.77 \times 10^3$ a.

TABLE D-5

PORE-WATER ^{99}Tc CONCENTRATIONS IN THE BOTTOM OR FOURTH SOIL LAYER AND ROOT-ZONE SOIL CONCENTRATIONS FROM TERRESTRIAL DISCHARGE FOR EACH FIELD

Concentration	Garden	Forage Field	Woodlot
Pore-water concentration in bottom soil layer C_{pw}^{Tc} (mol·m^{-3} water)	3.45×10^{-15}	5.38×10^{-16}	0
Steady-state root-zone soil concentration $(C_s^{Tc})_G$ (mol·kg^{-1} dry soil)	1.57×10^{-18}	2.45×10^{-19}	0

The final parameter in Equation (6.48) requiring evaluation is $(\beta_s^{Tc})_G$, which is defined in Equation (6.49). Values for $(tr^{Tc})_G$ and λ^{Tc} have already been set, and $\eta_s^{Tc} = 0$ because technetium is not volatile. The rate constant for cropping losses, ς^{Tc}, is defined in Equation (6.44). We set $\epsilon = 0.05$ (Section 6.5.5.1), $Bv^{Tc} = 2.4$ (mol·kg^{-1} wet biomass)/(mol·kg^{-1} dry soil) (Section 8.5.1.1), $Z_r = 0.3$ m (Section 6.3.1.2), $\rho_s = 1500$ kg dry soil·m^{-3} soil (Section 6.5.1.3), and $Y_j = 0.8$ kg wet biomass·m^{-2} soil (Section 8.5.8.1). With these values, $\varsigma^{Tc} = 2.13 \times 10^{-4}$ a^{-1} and $(\beta_s^{Tc})_G = 7.81 \times 10^{-4}$ a^{-1}.

Values for all of the parameters in Equation (6.48) have now been obtained. The root-zone soil concentration, $(C_s^{Tc})_G$, from terrestrial discharge to each field is found by evaluating Equation (6.48) using the pore-water concentration appropriate to each field. The results are listed in Table D-5.

D.5.2 DEPOSITION

Soil concentrations, $(C_s^{Tc})_D$, in the various fields resulting from atmospheric deposition are calculated in much the same way, using Equation (6.51). The source concentration in this case, C_D^{Tc}, is obtained from Equation (6.17). Since ^{99}Tc is not volatile, $(C_a^{Tc})_{AG} = 0$ mol·m^{-3} air, and $(C_a^{Tc})_{AP}$ is given by Equation (7.13). With AADL = 2.9×10^{-10} m^3 water·m^{-3} air (Section 7.5.1.2), $C_1^{Tc} = 4.09 \times 10^{-19}$ mol·m^{-3} (Section D.4), $Vd = 1.89 \times 10^5$ m·a^{-1} (Section 7.5.4.1), $P = 0.78$ m water·a^{-1}

TABLE D-6

ROOT-ZONE SOIL ^{99}Tc CONCENTRATIONS FROM ATMOSPHERIC AND IRRIGATION DEPOSITIONS, AND INTERMEDIATE RESULTS

Parameter or Concentration	Units*	Value
C_D^{Tc}	mol·m^{-3}	7.39×10^{-23}
$(Css)_{ID}^{Tc}$	L.kg^{-1}	571
$(tss)_{ID}^{Tc}$	a	971
$(tr^{Tc})_{ID}$	a	165
$(\beta_s^{Tc})_{ID}$	a^{-1}	6.28×10^{-3}
Root-zone soil concentration from deposition, $(C_s^{Tc})_D$	mol·kg^{-1}	4.07×10^{-23}

* For parameter descriptions and the full units see list of symbols in Appendix B.

(Section 9.1.3), Wr = 2.5 x 10^5 (Section 7.5.4.2) and I_W = 0.616 m water·a^{-1} (Section D.2), C_D^{Tc} becomes 7.39 x 10^{-23} mol·m^{-3} water. $(Css)_{ID}^{Tc}$ and $(tss)_{ID}^{Tc}$ are calculated from Equations (6.26) and (6.27) respectively, using regression constants appropriate to sand soil (Tables 6.4 and 6.5). The time scale $(tr^{Tc})_{ID}$ is found from an equation analogous to Equation (6.31) for the deposition case, with υ = 0.10 mol·kg^{-1} dry soil and τ_s = 10 a (Sheppard 1992). $(\beta_s^{Tc})_{ID}$ is evaluated using Equation (6.52). The final root-zone concentration from deposition, and the intermediate calculations used to obtain it, are listed in Table D-6. These results apply to all fields, because all experience the same deposition source.

D.5.3 IRRIGATION

The root-zone soil concentration resulting from irrigation is calculated from Equation (6.53). Since the source of irrigation water is the well, we set C_I^{Tc} equal to 5.39 x 10^{-15} mol·m^{-3} water, the ^{99}Tc concentration in well water (Section D.3). The parameters $(Css)_I^{Tc}$, $(tss)_{ID}^{Tc}$, $(tr^{Tc})_I$ and $(\beta_s^{Tc})_I$ have the same values for irrigation as the corresponding parameters for deposition (Table D-6). Assuming an irrigation period, t_{irr}, of 100 a (Section 6.5.5.3), Equation (6.53) predicts a root-zone soil concentration from irrigation $(C_s^{Tc})_I$ of 1.39 x 10^{-15} mol·kg^{-1} dry soil. This concentration applies to the garden only, because the forage field is not irrigated.

D.5.4 TOTAL CONCENTRATIONS

The total root-zone concentration for each field is found by summing over the pathways by which the field can become contaminated. The garden is subject to terrestrial discharge, atmospheric deposition and irrigation, with irrigation the dominant pathway. For this field, the total root-zone concentration is 1.39 x 10^{-15} mol·kg^{-1} dry soil. Deposition and terrestrial discharge contribute to the concentration in the forage field, with terrestrial discharge dominating. The total root-zone concentration for the forage field is 2.45 x 10^{-19} mol·kg^{-1}. The woodlot is subject to contamination via atmospheric deposition only, and has a total root-zone concentration of 4.07 x 10^{-23} mol·kg^{-1}.

D.6 AIR CONCENTRATIONS

Because ^{99}Tc is not volatile, it can reach the atmosphere via only five pathways: particle suspension from terrestrial sources, TP; particle suspension from aquatic sources, AP; agricultural fires, AF; energy fires, EF; and forest and land-clearing fires, LF. Each fire pathway involves a dispersion factor. For agricultural fires (Equation (7.26)), $(DISP)_T$ is calculated using Equation (7.4) with UWGHT = 1 (Section 7.5.3.2) and A_T = 9.17 x 10^4 m^2, the area of the forage field (Section D.2); this yields $(DISP)_T$ = 16.8 s·m^{-1}. The dispersion factor for forest and land-clearing fires (Equation (7.30)) is calculated in the same way, but with A_T = 9.52 x 10^4 m^2, the area of the woodlot; in this case, we find $(DISP)_T$ = 16.9 s·m^{-1}. For energy fires (Equation (7.27)), we calculate $(DISP)_B$ = 9.1 x 10^{-3} s·m^{-3}, using Equation (7.7), and KK = 2.0 (Section 7.5.3.5), BW = 9.7 m (Section 7.5.3.3), BH = 2.4 m (Section 7.5.3.4) and UCAV = 2.36 m·s^{-1} (Section 7.5.3.1).

Table D-7 summarizes the air concentrations resulting from the five pathways, and the equations and parameter values used to obtain these values. In our example, the air concentration caused by particle suspension from terrestrial sources is much larger than that from any other pathway. The total outdoor, $(C_a^{Tc})_O$, and indoor, $(C_a^{Tc})_I$, air concentrations for ^{99}Tc are therefore the same, 8.20×10^{-23} mol·m^{-3} air.

The rate at which ^{99}Tc is deposited to vegetation is calculated using Equation (7.43). With $Vd = 1.89 \times 10^5$ m·a^{-1} (Section 7.5.4.1), $P = 0.78$ m water·a^{-1} (Section 9.1.3), $Vr = 2.5 \times 10^5$ (Section 7.5.4.2) and the air concentrations given in Table D-7, we find $D_D^{Tc} = 3.15 \times 10^{-17}$ mol·m^{-2} soil·a^{-1}, or 8.63×10^{-20} mol·m^{-2}·d^{-1}.

D.7 DOSES TO MAN

The first step in implementing CALDOS is to convert the environmental concentrations predicted by the interface model and the other three submodels to units of becquerels per unit volume or mass. This was done by multiplying the molar concentrations by a factor 6.26×10^{10} Bq·mol^{-1} (Section 8.3). Values for all of the calculated input parameters required for CALDOS have been defined above, with the exception of the deposition rate, $(D_D^{Tc})_I$, to vegetation with irrigation water. Using Equation (8.9) with C_w^{Tc} equal to the concentration in well water (5.39×10^{-15} mol·m^{-3} water) and $I_w = 0.616$ m water·a^{-1} (Section D.2), we obtain $(D_D^{Tc})_I = 3.32 \times 10^{-15}$ mol·m^{-2} soil·a^{-1}, or 5.69×10^{-7} Bq·m^{-2}·d^{-1}.

Table D-8 summarizes the doses from the various exposure pathways, and the equations and parameter values used to obtain them. Full parameter descriptions and units are given in the list of symbols in Appendix B. In this simulation, the largest dose by far is received through the soil/plant/man pathway. This is the same as in the BIOTRAC sensitivity analysis (Figure 10-4). The plants eaten directly by man become highly contaminated because they are grown on soil irrigated with well water, and because the plant/soil concentration ratio, B_v^{Tc}, is large. The air/plant/man pathway involving irrigation, and the water/man pathway are the next most important pathways, but result in doses more than two orders of magnitude less than the soil/plant/man pathway. External exposure pathways make an insignificant contribution to the total dose for this simulation. This would likely be so for most simulation and most radionuclides.

REFERENCE

Sheppard, M.I. 1992. The soil submodel, SCEMR1, for the assessment of Canada's nuclear fuel waste management concept. Atomic Energy of Canada Limited Report, AECL-9577, COG-91-194.

TABLE D-7

EQUATIONS, PARAMETER VALUES AND CONCENTRATIONS FOR SUSPENSION PATHWAYS CONTRIBUTING TO THE TOTAL ^{99}Tc INDOOR AND OUTDOOR AIR CONCENTRATIONS

Pathway	Governing Equation	Parameter Values		Source of Parameter Values	Air Concentration, (C_a^{Tc})(mol·m^{-3} air)
TP	7.12	ADL	= 5.9 x 10^{-8} kg dry soil·m^{-3} air	7.5.1.1	8.20 x 10^{-23}
		C_s^{Tc}	= 1.39 x 10^{-15} mol·kg^{-1} dry soil (garden soil)	D.5.4	
AP	7.13	AADL	= 2.9 x 10^{-10} m^3 water·m^{-3} air	7.5.1.2	1.19 x 10^{-28}
		C_l^{Tc}	= 4.43 x 10^{-19} mol·m^{-3} water	D.4	
AF	7.24 to 7.26	Bv^{Tc}	= 2.4 (mol·kg^{-1} wet)/(mol·kg^{-1} dry)	8.5.1.1	7.51 x 10^{-26}
		C_s^{Tc}	= 2.45 x 10^{-19} mol·kg^{-1} dry soil (forage field soil)	D.5.4	
		ff	= 3.17 x 10^{-8} s^{-1}	7.3.5.1	
		Y_j	= 1.2 kg wet biomass·m^{-2} soil	8.5.8.1	
		$(EMFRAC)_{AF}$	= 0.2	7.5.2.6	
		$(DISP)_T$	= 16.8 s·m^{-1}	D.6	
EF	7.27	Bv^{Tc}	= 2.4 (mol·kg^{-1} wet)/(mol·kg^{-1} dry)	8.5.1.1	1.13 x 10^{-28}
		C_s^{Tc}	= 4.07 x 10^{-23} mol·kg^{-1} dry soil (woodlot soil)	D.5.4	
		FUELUS	= 3.5 x 10^{-3} MJ·s^{-1}	7.5.2.1	
		$(EMFRAC)_{EF}$	= 0.2	7.5.2.6	
		EW	= 5.5 MJ·kg^{-1} wet biomass	7.5.2.2	
		$(DISP)_B$	= 9.1 x 10^{-3} s·m^{-3}	D.6	
LF	7.29 and 7.30	Bv^{Tc}	= 2.4 (mol·kg^{-1} wet)/(mol·kg^{-1} dry)	8.5.1.1	2.30 x 10^{-30}
		C_s^{Tc}	= 4.07 x 10^{-23} mol·kg^{-1} dry soil (woodlot soil)	D.5.4	
		FY	= 2.2 kg wet biomass·m^{-2} land	7.5.2.4	
		$(EMFRAC)_{LF}$	= 1.0	7.5.2.6	
		$(DISP)_T$	= 16.9 s·m^{-1}	D.6	

TABLE D-8

EQUATIONS, PARAMETER VALUES AND ^{99}Tc DOSES FOR PATHWAYS LEADING TO THE EXPOSURE OF MAN

Pathway	Governing Equation	Parameter Values	Source of Parameter Values	Dose ($Sv \cdot a^{-1}$)
Soil/plant/man	8.2 and 8.4	Bv^{Tc} = 2.4 ($Bq \cdot kg^{-1}$ wet)/($Bq \cdot kg^{-1}$ dry)	8.5.1.1	5.10×10^{-11}
j = TE PLANT		C_s^{Tc} = 8.70×10^{-5} $Bq \cdot kg^{-1}$ dry soil (garden soil)	D.5.4	
		th = 1.0 d	8.5.3.1	
		U_j = 375.7 kg wet biomass$\cdot a^{-1}$	D.2	
		DFe^{Tc} = 6.5×10^{-10} $Sv \cdot Bq^{-1}$	8.5.2.1	
Soil/plant/ animal/man	8.6 and 8.7	Bv^{Tc} = 2.4 ($Bq \cdot kg^{-1}$ wet)/($Bq \cdot kg^{-1}$ dry)	8.5.1.1	
		C_s^{Tc} = 1.53×10^{-8} $Bq \cdot kg^{-1}$ dry soil (forage field soil)	D.5.4	
		DFe^{Tc} = 6.5×10^{-10} $Sv \cdot Bq^{-1}$	8.5.2.1	
j = TE MILK		F_j^{Tc} = 9.9×10^{-4} $d \cdot L^{-1}$ milk	8.5.1.2	2.83×10^{-16}
		Qf_j = 60 kg wet biomass$\cdot d^{-1}$	8.5.5.1	
		thf_j = 1 d	8.5.3.1	
		U_j = 199.4 kg milk$\cdot a^{-1}$	D.2	
j = TE MEAT		F_j^{Tc} = 8.5×10^{-3} $d \cdot kg^{-1}$ wet biomass	8.5.1.2	1.33×10^{-15}
		Qf_j = 50 kg wet biomass$\cdot d^{-1}$	8.5.5.1	
		thf_j = 5 d	8.5.3.1	
		U_j = 130.9 kg wet biomass$\cdot a^{-1}$	D.2	

continued...

TABLE D-8 (continued)

Pathway	Governing Equation	Parameter Values	Source of Parameter Values	Dose ($Sv \cdot a^{-1}$)
j = TE BIRD		F_j^{Tc} = 1.9 d·kg^{-1} wet biomass	8.5.1.2	9.67 x 10^{-16}
		Qf_j = 0.4 kg wet biomass·d^{-1}	8.5.5.1	
		thf_j = 1 d	8.5.3.1	
		U_j = 53.2 kg wet biomass·a^{-1}	D.2	
Air/plant/man	8.8, 8.10 and	Y_j = 0.80 kg wet biomass·m^{-2} soil	8.5.8.1	
j = TE PLANT	8.11	th = 1.0 d	8.5.3.1	
		tp = 12 d	8.5.3.3	
		te_j = 100 d	8.5.3.2	
		U_j = 375.7 kg wet biomass·a^{-1}	D.2	
		DFe^{Tc} = 6.5 x 10^{-10} Sv·Bq^{-1}	8.5.2.1	
Irrigation		D_b^{Tc} = 5.69 x 10^{-7} Bq·m^{-2} soil·d^{-1}	D.7	1.50 x 10^{-13}
		r_j = 0.05	8.5.7.1	
Deposition		D_b^{Tc} = 5.40 x 10^{-9} Bq·m^{-2} soil·d^{-1}	D.6	2.85 x 10^{-14}
		r_j = 1.0	8.5.7.1	
Air/plant	8.12 and 8.13	D_b^{Tc} = 5.40 x 10^{-9} Bq·m^{-2} soil·d^{-1}	D.6	
animal/man		tp = 12 d	8.5.3.3	
Deposition		DFe^{Tc} = 6.5 x 10^{-10} Sv·Bq^{-1}	8.5.2.1	
j = TE MILK		F_j^{Tc} = 9.9 x 10^{-4} d·L^{-1} milk	8.5.1.2	8.50 x 10^{-16}
		r_j = 1.0	8.5.7.1	

continued...

TABLE D-8 (continued)

Pathway	Governing Equation	Parameter Values	Source of Parameter Values	Dose ($Sv{\cdot}a^{-1}$)
		Y_j = 0.8 kg wet biomass·m^{-2} soil	8.5.8.1	
		Qf_j = 60 kg wet biomass·d^{-1}	8.5.5.1	
		thf_j = 1 d	8.5.3.1	
		te_j = 50 d	8.5.3.2	
		U_j = 199.4 kg milk·a^{-1}	D.2	
j = TE MEAT		F_j^{Tc} = 8.5 x 10^{-3} d·kg^{-1} wet biomass	8.5.1.2	3.19 x 10^{-15}
		r_j = 1.0	8.5.7.1	
		Y_j = 1.0 kg wet biomass·m^{-2} soil	8.5.8.1	
		Qf_j = 50 kg wet biomass·d^{-1}	8.5.5.1	
		thf_j = 5 d	8.5.3.1	
		te_j = 50 d	8.5.3.2	
		U_j = 130.9 kg wet biomass·a^{-1}	D.2	
j = TE BIRD		F_j^{Tc} = 1.9 d·kg^{-1} wet biomass	8.5.1.2	2.04 x 10^{-15}
		r_j = 1.0	8.5.7.1	
		Y_j = 1.2 kg wet biomass·m^{-2} soil	8.5.8.1	
		Qf_j = 0.4 kg wet biomass·d^{-1}	8.5.5.1	
		thf_j = 1 d	8.5.3.1	
		te_j = 100 d	8.5.3.2	
		U_j = 53.2 kg wet biomass·a^{-1}	D.2	
Water/animal/ man	8.14 and 8.15	C_w^{Tc} = 3.37 x 10^{-4} Bq·m^{-3} water (well)	D.3	
		DFe^{Tc} = 6.5 x 10^{-10} Sv·Bq^{-1}	8.5.2.1	

continued...

TABLE D-8 (continued)

Pathway	Governing Equation	Parameter Values	Source of Parameter Values	Dose ($Sv \cdot a^{-1}$)
j = TE MILK		F_j^{Tc} = 9.9 x 10^{-4} $d \cdot L^{-1}$ milk	8.5.1.2	2.60 x 10^{-15}
		Qdw_j = 0.060 m^3 $water \cdot d^{-1}$	8.5.5.2	
		thw_j = 0.0 d	8.5.3.1	
		U_j = 199.4 kg $milk \cdot a^{-1}$	D.2	
j = TE MEAT		F_j^{Tc} = 8.5 x 10^{-3} $d \cdot kg^{-1}$ wet biomass	8.5.1.2	9.76 x 10^{-15}
		Qdw_j = 0.040 m^3 $water \cdot d^{-1}$	8.5.5.2	
		thw_j = 4.0 d	8.5.3.1	
		U_j = 130.9 kg wet $biomass \cdot a^{-1}$	D.2	
j = TE BIRD		F_j^{Tc} = 1.9 $d \cdot kg^{-1}$ wet biomass	8.5.1.2	8.87 x 10^{-15}
		Qdw_j = 4.0 x 10^{-4} m^3 $water \cdot d^{-1}$	8.5.5.2	
		thw_j = 0.0 d	8.5.3.1	
		U_j = 53.2 kg wet $biomass \cdot a^{-1}$	D.2	
Soil/animal/man	8.16 and 8.17	C_s^{Tc} = 1.53 x 10^{-8} $Bq \cdot kg^{-1}$ dry soil (forage field)	D.5.4	
		DFe^{Tc} = 6.5 x 10^{-10} $Sv \cdot Bq^{-1}$	8.5.2.1	
j = TE MILK		F_j^{Tc} = 9.9 x 10^{-4} $d \cdot L^{-1}$ milk	8.5.1.2	1.97 x 10^{-18}
		Qs_j = 1.0 kg wet $biomass \cdot d^{-1}$	8.5.5.3	
		ths_j = 0.0 d	8.5.3.1	
		U_j = 199.4 kg $milk \cdot a^{-1}$	D.2	

continued...

TABLE D-8 (continued)

Pathway	Governing Equation	Parameter Values	Source of Parameter Values	Dose ($Sv \cdot a^{-1}$)
j = TE MEAT		F_j^{Tc} = 8.5×10^{-3} $d \cdot kg^{-1}$ wet biomass	8.5.1.2	8.90×10^{-18}
		Qs_j = 0.8 kg wet biomass$\cdot d^{-1}$	8.5.5.3	
		ths_j = 4.0 d	8.5.3.1	
		U_j = 130.9 kg wet biomass$\cdot a^{-1}$	D.2	
j = TE BIRD		F_j^{Tc} = 1.9 $d \cdot kg^{-1}$ wet biomass	8.5.1.2	6.05×10^{-18}
		Qs_j = 6×10^{-3} kg wet biomass$\cdot d^{-1}$	8.5.5.3	
		ths_j = 0.0 d	8.5.3.1	
		U_j = 53.2 kg wet biomass$\cdot a^{-1}$	D.2	
Water/fish/man j = FW FISH	8.18 and 8.19	C_l^{Tc} = 2.56×10^{-8} $Bq \cdot m^{-3}$ water	D.4	2.50×10^{-18}
		B_j^{Tc} = 0.015 m^3 water$\cdot kg^{-1}$ wet fish	8.5.1.3	
		thp = 0.5 d	8.5.3.1	
		U_j = 10.0 kg wet biomass$\cdot a^{-1}$	D.2	
		DFe^{Tc} = 6.5×10^{-10} $Sv \cdot Bq^{-1}$	8.5.2.1	
Water/man	8.20 and 8.21	C_w^{Tc} = 3.37×10^{-4} $Bq \cdot m^{-3}$ water (well)	D.3	1.41×10^{-13}
		thdw = 0.0 d	8.5.3.1	
		Udw = 0.641 $m^3 \cdot$water$\cdot a^{-1}$	D.2	
		DFe^{Tc} = 6.5×10^{-10} $Sv \cdot Bq^{-1}$	8.5.2.1	
Soil/man j = TE PLANT	8.22 to 8.24	C_s^{Tc} = 8.70×10^{-5} $Bq \cdot kg^{-1}$ dry soil (garden)	D.5.4	1.29×10^{-14}
		Hs = 0.04 kg dry soil$\cdot a^{-1}$	8.5.6.8	

continued...

TABLE D-8 (continued)

Pathway	Governing Equation	Parameter Values		Source of Parameter Values	Dose (Sv·a^{-1})
		Ps	= 5 x 10^{-4} kg soil·kg^{-1} biomass	8.5.6.9	
		U_j	= 375.7 kg wet biomass·a^{-1}	D.2	
		DFe^{Tc}	= 6.5 x 10^{-10} Sv·Bq^{-1}	8.5.2.1	
Air/man	8.25 and 8.26	$(C_a^{Tc})_O$	= 5.13 x 10^{-12} Bq·m^{-3} air	D.6	1.19 x 10^{-16}
		$(C_a^{Tc})_I$	= 5.13 x 10^{-12} Bq·m^{-3} air	D.6	
		Og	= 0.2	8.5.4.2	
		Ob	= 0.8	8.5.4.3	
		I_A	= 8.62 x 10^{3} m^{3} air·a^{-1}	D.2	
		DF^{Tc}	= 2.7 x 10^{-9} Sv·Bq^{-1}	8.5.2.1	
Air immersion	8.35	$(C_a^{Tc})_O$	= 5.13 x 10^{-12} Bq·m^{-3} air	D.6	4.11 x 10^{-21}
		$(C_a^{Tc})_I$	= 5.13 x 10^{-12} Bq·m^{-3} air	D.6	
		Og	= 0.2	8.5.4.2	
		Ob	= 0.8	8.5.4.3	
		DFa^{Tc}	= 8.0 x 10^{-10} (Sv·a^{-1})/(Bq·m^{-3})	8.5.2.2	
Water immersion	8.36	C_w^{Tc}	= 3.37 x 10^{-4} Bq·m^{-3} water (well)	D.3	5.80 x 10^{-18}
		Oe	= 0.02	8.5.4.1	
		DFh^{Tc}	= 8.6 x 10^{-13} (Sv·a^{-1})/(Bq·m^{-3})	8.5.2.2	
Ground exposure	8.37	C_s^{Tc}	= 8.70 x 10^{-5} Bq·kg^{-1} dry soil (garden)	D.5.4	4.30 x 10^{-16}
		dws	= 0.95 kg dry soil/kg^{-1} wet soil	8.5.12.2	

continued...

TABLE D-8 (concluded)

Pathway	Governing Equation	Parameter Values	Source of Parameter Values	Dose ($Sv \cdot a^{-1}$)
		$Og = 0.2$	8.5.4.2	
		$DFg^{Tc} = 2.6 \times 10^{-11}\ (Sv \cdot a^{-1})/(Bq \cdot kg^{-1})$	8.5.2.2	
Exposure to building materials	8.38 to 8.40	$DFb^{Tc} = 0.0\ (Sv \cdot a^{-11})/(Bq \cdot kg^{-1}\ day)$*	8.5.2.2	0.0
Total ingestion dose				5.14×10^{-11}
Total internal dose				5.14×10^{-11}
Total external dose				4.36×10^{-16}
Total dose				5.14×10^{-11}

Note: For parameter descriptions and full units see list of symbols in Appendix B.

* With a DFb^{Tc} value of 0.0, the dose is also zero.

APPENDIX E

GLOSSARY

alpha particle: The nucleus of a helium atom, consisting of two protons and two neutrons. It has a charge equal to two electrons but with the opposite (positive) sign. Alpha particles are commonly emitted from heavy radionuclides such as ^{239}Pu when they decay. Alpha particles transfer their energy in a very short distance and are readily shielded by a piece of paper or the dead layer of human skin. See also "alpha radiation".

alpha radiation: The emission of alpha particles from the nucleus of an unstable atom (radionuclide). Since alpha radiation cannot penetrate the outer layer of skin, it is not normally a radiation hazard to humans and other biota, unless it is located inside the body. See also "alpha particle".

annual dose: See "dose equivalent", effective dose equivalent", "committed effective dose equivalent" and "radiological dose".

Atomic Energy Control Board (AECB): The Canadian federal regulatory agency which has jurisdiction over nuclear fuel wastes. Established in 1946, the organization's mandate is to ensure that the use of nuclear energy in Canada does not pose undue risks to health, safety, security and the environment. Through its licensing and inspection systems, the AECB provides control and supervision of the development, application and use of atomic energy in Canada.

Atomic Energy of Canada Limited (AECL): A Canadian crown corporation created in 1952 to develop nuclear technology for peaceful uses.

backfill: In a disposal vault, the material used to refill excavated portions in disposal rooms, shafts and tunnels after the containers with the waste and buffer have been emplaced. In the CNFWMP, the backfills being considered are a mixture of glacial lake clay and crushed granite from the vault excavation, and a mixture of sodium bentonite clay and silica sand.

barrier: A feature of a disposal system which delays or prevents radionuclides from escaping from the disposal vault and migrating into the biosphere. A natural barrier is a feature of the geosphere in which the disposal vault is located. An engineered barrier is a feature made by or altered by man and includes the wasteform and its container, casks for transportation and disposal of the waste, and any sealing materials used.

BEIR: Committee on Biological Effects of Ionizing Radiation Research of the National Research Council. This committee has prepared a series of reports to advise the U.S. government on the health consequences of radiation exposure. These reports have had a strong influence on radiation protection standards.

becquerel: The SI unit of radioactivity for measuring the rate of decay of a radioactive substance. It is equivalent to the disintegration of one radioactive nucleus per second.

beta particle: A free electron or positron emitted by many radionuclides (e.g., ^{14}C and ^{129}I) during radioactive decay. The emission of an electron indicates the transformation of a neutron to a proton, and the emission of a positron indicates the transformation of a proton into a neutron. Beta particles can penetrate biological tissue to a depth of 1 to 2 cm. They may pose both an internal and external hazard to humans and other biota. See also "beta radiation".

beta radiation: The emission of electrons or positrons from the nucleus of an unstable atom (radionuclide). See also "beta particle".

biosphere: Usually defined as the portion of the earth inhabited by living organisms. In the CNFWMP, it has a more specific meaning. In aquatic areas the biosphere/geosphere interface occurs between the deep compacted and the shallow mixed sediments, and in terrestrial areas the interface is formed by the watertable. Thus, the biosphere includes mixed sediments, surface waters, soils, and the lower parts of the atmosphere. Even though the overburden and the geosphere may contain microorganisms, these regions are considered parts of the geosphere. See also "geosphere".

buffer: See "reference buffer material".

Canadian Nuclear Fuel Waste Management Program (CNFWMP): A program of research and development on radioactive waste management established in a 1978 Joint Statement by the Federal Government and the Government of Ontario. The aim is to develop and assess the concept of disposing of nuclear fuel waste in the plutonic rock of the Canadian Shield. AECL is responsible for verifying the safety of this disposal concept. Ontario Hydro is responsible for developing and demonstrating nuclear fuel waste storage technology, and for transportation of these wastes from reactor sites. A second Joint Statement in 1981 imposed the restriction that the concept must be assessed, reviewed and accepted before a site could be accepted.

Canadian Shield: An extensive area of Precambrian rocks exposed over large parts of central and eastern Canada. It lies approximately to the east of a line passing through Great Bear Lake, Great Slave Lake, Lake Athabasca and Lake Winnipeg, and to the north of the continuation of this line through Lake Superior, Lake Huron and the St. Lawrence River. It is composed of metamorphic and igneous rocks. Orogenic events have occurred over different parts of the Shield at various times but some parts have been free of such activity for about 2.5 billion years. Almost the entire Shield has been stable for the last 900 million years. See also "pluton".

CANDU Owners Group (COG): A group formed in 1984 by the Canadian CANDU-owning utilities and Atomic Energy of Canada Limited. Its purpose is to provide a framework that will promote closer cooperation among the utilities owning and operating CANDU stations in matters relating to plant operation and maintenance, and to foster cooperative development programs leading to improved plant performance.

CANDU: CANadian Deuterium Uranium, the name of the Canadian-designed reactor which uses natural uranium fuel and is moderated by heavy water. CANDU is a registered trademark of Atomic Energy of Canada Limited.

Chalk River Laboratories (CRL): A laboratory owned and operated by AECL Research, located at Chalk River, Ontario. Previously named Chalk River Nuclear Laboratories (CRNL).

closure: In the CNFWMP, a stage in the evolution of a disposal facility that follows decommissioning. Closure includes the shutdown and removal of monitoring systems whose continued existence could affect the long-term safety of the disposal vault, and the sealing of boreholes. Completion of this stage forms the event that defines the end of the preclosure phase and the start of the postclosure phase.

collective dose: The total dose for a human population, usually based on the committed effective dose equivalent. The collective dose can be calculated by multiplying the average dose for the exposed population by the number of people in the population. The units used are person-Sv. See also "radiological dose", "effective dose equivalent" and "committed effective dose equivalent".

committed effective dose equivalent: The summation over time of either the dose equivalent rate or the effective dose equivalent rate over some specified time, usually 50 a. That is the dose a person would receive over a lifetime, measured in $Sv \cdot a^{-1}$, after an internal intake of of a radionuclide. For external exposure there is no intake of radionuclides and, hence, no commitment. However, in such cases, the effective dose equivalent rate in $Sv \cdot a^{-1}$ can be added to the committed effective dose equivalent from internal exposure to assess the combined exposures. See also "radiological dose", "dose equivalent", "effective dose equivalent" and "risk factor".

concept assessment: The prediction and evaluation of the consequences for humans and other biota from the geological disposal concept for Canada's nuclear fuel wastes, documented in an environmental impact statement for submission to the Federal Environmental Assessment Review Panel. Concept assessment consists of the preclosure assessment and the postclosure assessment.

container: See "reference container".

containment: In the NFWMP, the retention of radioactive wastes in such a way that they are effectively prevented from being dispersed into the biosphere, or are released only at a regulated and acceptable rate. Containment also refers to the structures used to effect such retention.

disposal: A permanent method of long-term management of radioactive wastes in which there is no intention of retrieval and which, ideally, uses techniques and designs that do not rely for their success on long-term institutional control beyond a reasonable period of time.

disposal container: See "reference container".

disposal facility: A disposal vault and the supporting buildings and equipment to receive the waste and package it in durable containers; shafts and equipment to transfer the containers from the surface to the vault; equipment to handle the containers in the vault; and the materials and equipment to excavate the vault, emplace the disposal containers and to fill and seal the vault, tunnels and shafts.

disposal vault: An underground structure excavated in rock including horizontal access tunnels and disposal rooms where containers of nuclear fuel waste would be placed for disposal. Note that the vault does not include shafts or service areas.

dose: See "radiological dose".

dose equivalent: The strict definition of radiological dose for humans is the energy absorbed per unit mass of tissue exposed to ionizing radiation, measured in gray (Gy). The dose equivalent, measured in sievert (Sv), is the product of the dose and a radiation quality factor, Q. This quality factor is a function of how a certain type of radiation deposits its energy within the body. Radiations with high quality factors deposit a lot of energy in a short distance, whereas those with lower factors deposit less energy over the same distance. For example, alpha radiation has a quality factor of 20, whereas beta and gamma radiations have a value of 1. The dose equivalent accounts for the fact that different types of radiations react differently within the body. See also "radiological dose", "effective dose equivalent", "committed effective dose equivalent", "alpha radiation", "beta radiation" and "gamma radiation".

effective dose equivalent: The summation of the products of the dose equivalent that a particular human tissue or organ has received and the corresponding organ weighting factor, W_T. This summation usually considers the entire body. The organ weighting factors are determined by the relative radiosensitivity of each organ. The effective dose equivalent is used to accurately determine the detrimental effect of a particular dose to the body, accounting for the fact that some organs are more sensitive to the effects of radiation exposure than others. See also "radiological dose", "dose equivalent", and "committed effective dose equivalent".

fuel recycling: The reprocessing of used nuclear reactor fuel and the reuse as new fuel of the uranium and plutonium thus recovered.

gamma radiation: The emission of photons (gamma rays), which carry energy but no charge, by an unstable atom (radionuclide). Gamma radiation is the most highly penetrating radiation. It can pass through the human body, but is stopped a few metres of water or concrete. See also "gamma ray".

gamma ray: High-energy, highly-penetrating photons of short wave length commonly emitted from the nucleus of a radioactive atom (radionuclide) during radioactive decay as a result of a transition from one of its excited energy levels to a lower level.

geological disposal: All approaches to the long-term management of nuclear fuel wastes that depend upon placing the wastes underground in a selected host medium to isolate the wastes from humans and other biota.

geosphere: The solid outer portion of the earth's crust. In the CNFWMP concept for the geological disposal of nuclear fuel waste, the geosphere, consisting of rock, overburden, compacted sediment and associated groundwater flow systems, is one of the major barriers surrounding the disposal vault. See also "biosphere".

gray (Gy): The SI unit of absorbed dose for ionizing radiation, equal to 1 joule of radiation energy absorbed in 1 kilogram of the material of interest. See also "radiological dose".

International Atomic Energy Agency (IAEA): The organization established in 1957 by the United Nations as the international body responsible for on-site nuclear reactor inspections and safeguards measures that assist the member states of this agency to demonstrate that no nuclear material is being diverted to non-peaceful purposes from safeguarded nuclear facilities.

International Commission on Radiological Protection (ICRP): An independent non-government expert body founded in 1928. This commission establishes radiation protection standards that are followed by most countries. See also "UNSCEAR and "BEIR".

long-term management: In nuclear waste management, periods of time which exceed the time during which institutional controls can be expected to last.

median-value simulation: A single simulation performed for deterministic analysis with all the probabilistic parameter values at their median values (i.e., the central value on the 50th quantile).

nuclear fuel waste: A solid, highly radioactive material that is either the used nuclear fuel that has been removed from a CANDU nuclear power reactor or a waste form incorporating the highly radioactive waste that would be removed from the fuel if the fuel were to be recycled.

pluton: An intrusive body of igneous rock formed beneath the surface of the earth by consolidation of magma. Plutons, and the similar larger batholiths, are common on the Canadian Shield.

population dose: See "collective dose".

postclosure: The project phase following the closure stage of a disposal facility, after the vault and other underground facilities have been decommissioned and sealed, the monitoring systems whose continued operation could affect long-term disposal vault safety have been sealed, and the surface facilities have been decontaminated and decommissioned. See also "preclosure".

postclosure assessment: Safety analysis of the waste disposal system, starting after the disposal vault has been closed. The objectives are to determine the long-term impacts on the environment and humans of the disposal facility, and to provide estimates of risk that can be compared with regulatory criteria. See also "preclosure assessment" and "concept assessment".

preclosure: The project phase which includes the siting, construction, operation, decommissioning and closure of a disposal facility including the disposal vault, surface facilities and surrounding site. It also includes the final shaft and monitoring borehole sealing. The transportation of used nuclear fuel from nuclear generating stations to the disposal facility is also part of the preclosure phase. See also "postclosure".

preclosure assessment: Safety analysis of the waste disposal system that deals with the potential impacts on the environment and humans during construction, operation, decommissioning and closure of a disposal facility. It includes an assessment of the transportation of used nuclear reactor fuel from nuclear generating stations to the disposal facility. See also "postclosure assessment" and "concept assessment".

radioactive decay: The changing and progressive decrease in the number of unstable atoms (radionuclides) in a substance, due to their spontaneous nuclear disintegration or transformation into different atoms, during which particles and/or photons are emitted. See also "alpha radiation", "beta radiation" and "gamma radiation".

radiological dose: The strict definition of radiological dose is the energy absorbed per unit mass of biological tissue exposed to ionizing radiation, measured in gray (Gy). However, the term radiological dose, or simply dose, is commonly used in different ways. In the CNFWMP we use it also as abbreviations for dose equivalent, effective dose equivalent, committed effective dose equivalent and collective dose in units of Gy, $Gy \cdot a^{-1}$, Sv, $Sv \cdot a^{-1}$ and person-Sv. Furthermore, we use the term for both humans and other biota.

reference buffer material: A sealing material of specified chemical and physical properties which would surround disposal containers in a disposal vault. In the CNFWMP, the reference buffer material is a compacted sand-bentonite mixture.

reference container: In the CNFWMP, an enclosed cylindrical vessel of titanium alloy which would hold 72 bundles of used nuclear reactor fuel. Glass beads would be compacted around the fuel bundles inside the container to support the container walls.

reference disposal system (or reference system): A hypothetical disposal system evaluated during concept assessment. A reference system is a specific (but hypothetical) implementation of the concept for disposal of Canada's nuclear fuel waste.

risk factor: A factor used to convert radiological dose for man to risk. The AECB risk factor of 0.02 serious health effects per sievert is used to quantify the risk that an individual will die from cancer or transmit a serious genetic effect to offsprings. The value is an average over age and sex.

seal: Such things as buffer material, backfill, bulkheads, grout and plugs which, in the CNFWMP, act as barriers in a disposal vault by helping to isolate the waste material and to retard the movement of water. Seals such as buffer materials would also affect the rates of container corrosion, fuel dissolution, and radionuclide migration.

secular equilibrium: An equilibrium reached between a precursor and daughter nuclide in which the daughter nuclide decays at the same rate as it is produced. The daughter nuclide must be much shorter-lived than the precursor. Secular equilibrium is reached after a period equivalent to 6 to 10 daughter half-lives. See also "radioactive decay".

sievert (Sv): The SI unit of dose equivalent. 1 Sv equals 1 joule per kilogram. See also "radioactive decay".

siting: In the CNFWMP, the project stage of selecting a suitable location for a facility, including appropriate assessment and definition of the related design bases, and numerous other factors.

SYVAC: SYstems Variability Analysis Code, a family of computer programs written at AECL's Whiteshell Laboratories to perform probabilistic calculations on the long-term performance of disposal systems. Several generations and versions of SYVAC have been produced. Different generations of computer code are substantially different from one another. Three generations of SYVAC now exist; they are referred to as SYVAC1, SYVAC2 and SYVAC3. Different versions of computer code are only slightly different from one another; each SYVAC generation has several versions. See also "SYVAC3-CC3".

SYVAC3-CC3: SYstems Variability Analysis Code - Generation 3, with models describing the Canadian concept, Generation 3. This computer program belongs to the SYVAC family, and is the name given to the program used in the postclosure assessment. It consists of the SYVAC3 executive code and the third generation of the vault, geosphere and biosphere models.

Technical Advisory Committee (TAC): The Technical Advisory Committee to AECL on the Canadian Nuclear Fuel Waste Management Program is an independent group of distinguished scientists and engineers that regularly examines the technical quality of the work performed in the program. TAC includes a separate Biosciences Subcommittee. Members are nominated by Canadian professional societies. The TAC publishes an annual report, available to the public, commenting on the CNFWMP.

UNSCEAR: United Nations Scientific Committee on the Effects of Atomic Radiation established by the General Assembly in 1955. It reports annually to the General Assembly and submits at irregular intervals comprehensive reports with scientific annexes. These review reports have had a strong influence on radiation protection standards, e.g., by the ICRP.

Used-Fuel Disposal Centre (UFDC): The surface and underground site, workings, structures, processes and systems necessary to receive used nuclear fuel in transportation casks, package it in disposal containers, emplace and seal it in a geological medium and provide all the supporting services and systems to do so in a safe and acceptable manner. In the CNFWMP, it is a conceptual design of a used-fuel disposal facility developed for use in concept assessment. The design was used by AECL to assess the engineering feasibility, costs, safety and potential environmental impact of disposing of used nuclear fuel in the manner described in the EIS documents. The design is based on specifications for all disposal system components and activities.

vault: See "disposal vault".

Whiteshell Laboratories (WL): A laboratory owned and operated by AECL Research, located at Pinawa, Manitoba. Previously named Whiteshell Nuclear Research Establishment (WNRE).

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* Unrestricted, unpublished report available from SDDO, AECL Research, Chalk River Laboratories, Ontario, K0J 1J0.

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