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The Disposal of Canada's Nuclear Fuel Waste: A Study of Postclosure Safety of In-Room Emplacement of Used CANDU Fuel in Copper Containers in Permeable Plutonic Rock Volume 4: Biosphere Model

Le stockage permanent des déchets de combustible nucléaire du Canada : Étude de la sûreté post-fermeture de la mise en place en chambre du combustible CANDU irradié renfermé dans des conteneurs en cuivre enfouis dans la roche plutonique perméable Volume 4 : Modèle de biosphère

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THE DISPOSAL OF CANADA'S NUCLEAR FUEL WASTE: A STUDY OF POSTCLOSURE SAFETY OF IN-ROOM EMPLACEMENT OF USED CANDU FUEL IN COPPER CONTAINERS IN PERMEABLE PLUTONIC ROCK VOLUME 4: BIOSPHERE MODEL

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ABSTRACT

AECL (Atomic Energy of Canada Limited) has developed a disposal concept for Canada's nuclear fuel waste, which calls for a vault deep in plutonic rock of the Canadian Shield. The concept has been fully documented in an environmental impact statement (EIS) for review by a panel under the Canadian Environmental Assessment Agency. The EIS includes the results of the EIS postclosure assessment case study to address the long term safety of the disposal concept. To more fully demonstrate the flexibility of the disposal concept and our assessment methodology, we are now carrying out another postclosure assessment study, which involves different assumptions and engineering options than those used in the EIS. In response to these changes, we have updated the BIOTRAC (BIOsphere Transport and Assessment Code) model developed for the EIS postclosure assessment case study.

The main changes made to the BIOTRAC model are the inclusion of ^{36}Cl, ^{137}Cs, ^{239}Np and ^{243}Am; animals inhalation pathway; International Commission on Radiological Protection 60/61 human internal dose conversion factors; all the postclosure assessment nuclides in the dose calculations for non-human biota; and groundwater dose limits for ^{14}C, ^{36}Cl and ^{129}I for non-human biota to parallel these limits for humans. We have also reviewed and changed several parameter values, including evasion rates of gaseous nuclides from soil and release fractions of various nuclides from domestic water, and indicated changes that affect the geosphere/biosphere interface model. These changes make the BIOTRAC model more flexible.

As a result of all of these changes, the BIOTRAC model has been significantly expanded and improved, although the changes do not greatly affect model predictions. The modified model for the present study is called BIOTRAC2 (BIOTRAC - Version 2). The full documentation of the BIOTRAC2 model includes the report by Davis et al. (1993a) and this report.

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LE STOCKAGE PERMANENT DES DÉCHETS DE COMBUSTIBLE NUCLÉAIRE DU CANADA : ÉTUDE DE LA SÛRETÉ POST-FERMETURE DE LA MISE EN PLACE EN CHAMBRE DU COMBUSTIBLE CANDU IRRADIÉ RENFERMÉ DANS DES CONTENEURS EN CUIVRE ENFOUIS DANS LA ROCHE PLUTONIQUE PERMÉABLE
VOLUME 4 : MODÈLE DE BIOSPHÈRE

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RÉSUMÉ

EACL (Énergie atomique du Canada limitée) a mis au point un concept de stockage permanent des déchets de combustible nucléaire du Canada dans une installation de stockage profonde creusée dans la roche plutonique du Bouclier canadien. Le concept a été décrit entièrement dans une Étude d'impact sur l'environnement (EIE) examinée par une commission coiffée par l'Agence canadienne d'évaluation environnementale. L'EIE comprend les résultats de l'étude de cas de l'évaluation post-fermeture qui a porté sur la sûreté à long terme du concept de stockage permanent. Afin de démontrer plus clairement la souplesse du concept de stockage permanent et notre méthodologie d'évaluation, nous procédons actuellement à une autre étude d'évaluation post-fermeture qui porte sur des hypothèses et des options techniques différentes de celles qui ont été utilisées dans l'EIE. Face à ces changements, nous avons mis à jour le modèle BIOTRAC (code BIOTRAC pour le calcul du transport dans la biosphère et des conséquences) mis au point pour l'étude de cas de l'évaluation post-fermeture.

Les principaux changements apportés au modèle BIOTRAC comprennent l'inclusion du ^{36}Cl, du ^{137}Cs, du ^{239}Np et de l'^{243}Am, les voies d'inhalation par les animaux; les facteurs de conversion de dose interne 60/61 de la Commission internationale de radioprotection chez les humains, tous les nucléides utilisés dans l'évaluation post-fermeture pour les calculs de dose relatifs au biote non humain et limites de dose dans les eaux souterraines dans le cas du ^{14}C, du ^{36}Cl et de l'^{150}I pour le biote non humain afin d'établir un parallèle avec ces limites dans le cas des êtres humains. Nous avons examiné et modifié plusieurs paramètres, y compris les taux d'échappement de nucléides gazeux du sol et les fractions de rejet de divers nucléides des eaux domestiques, et indiqué les modifications qui touchent le modèle d'interface entre la géosphère et la biosphère. Ces modifications assouplissent le modèle BIOTRAC.

À la suite de toutes ces modifications, le modèle BIOTRAC a été considérablement étendu et amélioré, même si les modifications ne touchent pas pour la peine les prévisions du modèle. Le modèle modifié pour l'étude en cours est appelé BIOTRAC2 (BIOTRAC - Version 2). La documentation complète du modèle BIOTRAC2 comprend le rapport de Davis et collaborateurs (1993a) et le présent rapport.

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1996

AECL-11494-4
COG-95-552-4

CONTENTS

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CONTENTS (continued)

concluded...

CONTENTS (concluded)

LIST OF TABLES

LIST OF FIGURES

PREFACE

The concept for disposal of Canada's nuclear fuel waste involves isolating the waste in corrosion-resistant containers emplaced and sealed within a vault at a depth of 500 to 1000 m in plutonic rock of the Canadian Shield. The technical feasibility and social aspects of the concept, and its impact on the environment and human health, are presented in an Environmental Impact Statement (EIS) (AECL 1994a), a summary of the EIS (AECL 1994b) and a set of nine primary references (Davis et al. 1993, Davison et al. 1994a, b; Goodwin et al. 1994, Greber et al. 1994, Grondin et al. 1994, Johnson et al. 1994a, b; Simmons and Baumgartner 1994).

The disposal concept permits a choice of methods, materials, site locations and designs (AECL 1994, Johnson et al. 1994a, Simmons and Baumgartner 1994). This preface puts into perspective the following three studies which illustrate the long-term safety of different implementations of the concept:

- the postclosure assessment case study of a reference disposal system presented in the EIS (AECL 1994a, b; Goodwin et al. 1994);

- a study to illustrate how to identify a favourable vault location that would ensure long groundwater travel times from the vault to the accessible environment (Stevenson et al. 1995, 1996, Ophori et al. 1995, 1996); and

- the present study that illustrates (i) the flexibility for designing engineered barriers to accommodate a permeable host-rock condition in which advection is the rate-determining contaminant transport process (Baumgartner et al. 1996), and (ii) the flexibility of the modelling methodology to simulate the long-term performance of different design options and site characteristics (Goodwin et al. 1996, Johnson et al. 1996, Stanchell et al. 1996, Wikjord et al. 1996, Zach et al. 1996a (this report)).

THE POSTCLOSURE ASSESSMENT CASE STUDY PRESENTED IN THE EIS

The EIS (AECL 1994a, b) and four of the primary references (Davis et al. 1993a, Davison et al. 1994b, Goodwin et al. 1994, Johnson et al. 1994b) describe a case study of the long-term (i.e., postclosure) performance of a hypothetical implementation of the concept, referred to as the *reference disposal system*.

The reference system illustrates what a disposal system, including the vault, geosphere and biosphere, might be like. Although it is hypothetical, it is based on information derived from extensive laboratory, field and engineering investigations. Many of the assumptions made about the long-term performance of the reference system are conservative; that is, they would tend to overestimate adverse effects. The technology specified is either available or judged to be readily achievable. The reference disposal system includes one possible choice among the options for such things as the waste form, the disposal container, the buffer and backfill, the shaft seals and bulkheads, the location and depth of the vault, and the orientation and layout of the vault with respect to the geological features of the site. The components and designs chosen for the engineered barriers and the site conditions represented in the reference system are not being recommended, but rather, they illustrate a technically feasible way of implementing the disposal concept. In an actual implementation of the concept, the engineered system would be adapted to the lithostructural, hydrogeological, geochemical, geothermal, geomechanical, and geomicrobiological conditions of the host rock formation, and the expected evolution of those conditions over thousands of years.

The *reference vault* (Johnson et al. 1994b) of the EIS postclosure assessment case study includes used-fuel bundles from CANDU® reactors, encapsulated in thin-walled Grade-2 titanium alloy containers packed with particulate for mechanical support, emplaced in boreholes in the floor of rooms, and surrounded by a sand-bentonite mixture. The rooms are filled with a lower backfill of crushed granite and lake clay and an upper backfill of sand and bentonite, and the entrances are sealed with concrete bulkheads. The plan area and the design capacity of the vault were initially set at 4.0 km^2 and 10.1 million fuel bundles (191 000 Mg U) respectively. The fuel inventory is roughly equivalent to the waste that would accrue in 100 a at the current production rates in Canada. The plan area was subsequently reduced to 3.2 km^2 and the inventory to 8.5 million bundles (162 000 Mg U), as a result of design constraints to ensure a large margin of safety in the case study. The borehole-emplacement geometry was modelled as layered planar elements (slabs) representing the waste form, buffer, backfill and host rock.

The *reference geosphere* (Davison et al. 1994b) consists of the host rock formation, its groundwater flow system, the materials used to seal the shafts and exploration boreholes, and a water well. The geological characteristics of the reference geosphere are derived from data from AECL's Whiteshell Research Area (WRA), located near Lac du Bonnet, Manitoba. This area includes a substantive portion of the Lac du Bonnet Batholith, a large granitic rock body several kilometres deep with an exposed surface measuring over 60 km long and 20 km across at its widest part. The granitic body was intruded over 2.5 billion years ago into the rocks existing at the time. The batholith, the surrounding rocks, and the interfaces between them have been the subject of field investigations for more than 15 a. Most of the information about the rock mass, such as the location and orientation of fractures and fracture zones, is based on field studies of the WRA, including detailed investigations that were conducted to locate and construct an Underground Research Laboratory (URL) to a depth of 440 m. For geological structures outside the areas where detailed borehole information was available, inferences have been made on the basis of nearby boreholes; geological mapping; and satellite, airborne and ground-based geophysical surveys. The hypothetical vault for the reference system was located at a depth of 500 m within the rock mass investigated at the URL to ensure that the maximum amount of available subsurface data was used to construct the geosphere model.

In the postclosure assessment of the reference system, we assumed that a large, low-dipping, fracture zone — designated LD1 — was located close to the vault horizon. Although field evidence from the URL revealed that this fracture zone did not extend beyond a depth of about 400 m, we conservatively assumed that it continued to much greater depths and connected with other vertical fracture zones. In this situation, LD1 became a pathway for rapid groundwater flow from the depth of the hypothetical vault to the accessible environment. We constrained all waste disposal rooms to be located beneath LD1 (i.e., to the footwall side of the fracture) and imposed a waste exclusion distance of 50 m within the low-permeability, sparsely fractured rock domain between this fracture zone and the nearest waste disposal room of the vault. To accommodate the waste exclusion distance, we chose to restrict the waste capacity of the vault relative to the capacity specified in a conceptual engineering study (Simmons and Baumgartner 1994). These design constraints, together with the hydrogeological properties of the rock beneath LD1, ensured that (i) contaminants passed through the backfill, a large reservoir which reacts strongly with most of the contaminants, and (ii) diffusion was the dominant transport process from the waste disposal rooms through the lower rock domain to the fracture zone.

CANDU® is a registered trademark of Atomic Energy of Canada Limited (AECL).

The *reference biosphere* (Davis et al. 1993a) consists of the surface and near-surface environment, including the water, soil, air, people, and other organisms, as encountered on the Canadian Shield as a whole. However, the parts of the biosphere that interface with the geosphere are specific to the WRA. In all other respects, the biosphere is assumed to be typical of the Canadian Shield, consisting of rocky outcrops; bottom lands with pockets of soil, bogs, and lakes; and uplands with meadows, bush, and forests. No major changes in the topography of the region are likely to occur during the 10 000 a following closure of a disposal facility. Changes in climate, surface water flow patterns, soils, and vegetation types are expected to be within the range of variation currently observed across the Shield; such variations are included in the distributions of values of model parameters specified for the EIS case study.

The long-term safety analyses of this system of engineered and geological barriers indicated that the maximum estimated mean dose rate to an individual in the critical group during the first 10 000 a is about 100 million times smaller than dose rate from natural background radiation. The corresponding risk is about a million times smaller than the radiological risk criterion specified by the Atomic Energy Control Board in Regulatory Document R-104 (AECB 1987).

A STUDY TO IDENTIFY A FAVOURABLE VAULT LOCATION

In an actual implementation, it would be advantageous to locate the disposal vault in a hydraulically favourable setting within the large-scale groundwater flow system of a siting area. Recently, we completed a separate study to illustrate how such a location could be found within the WRA. The conceptual hydrogeological model of the WRA was revised using information from a program of regional geologic mapping, geophysical surveys and borehole drilling and testing (Stevenson et al. 1995, 1996). Large-scale groundwater flow modelling was then performed using a three-dimensional, finite-element hydrogeological code; and groundwater travel times, flow pathways and discharge locations were determined with a particle tracking code (Ophori et al. 1995, 1996).

This study has indicated that diffusion is the rate-determining transport process and diffusive transport times greater than 10^5 a could likely be achieved by selecting a vault location at 750 m depth about 5 km northeast of the URL. Advective travel times are about two orders of magnitude longer than the diffusive transport time. Since the groundwater flow and particle-tracking analyses indicated that such a favourable location would likely ensure a margin of safety even greater than that calculated for the EIS case study, a full systems analysis was not carried out. Instead, we directed our efforts to the present study in which we evaluate the long-term effects of a hypothetical geological setting with a permeable host-rock condition.

THE PRESENT STUDY

A wide range of design options is possible within the general definition of the disposal concept (AECL 1994a, b; Johnson et al. 1994a, Simmons and Baumgartner 1994). In the present study, we illustrate the potential of designing the engineered barriers and the vault to increase the robustness of the long-term safety case, or to compensate for hydrogeological conditions that could result in a less effective geosphere barrier than the one we specified for the EIS case study. In addition, we illustrate the flexibility of the modelling approach to integrate new features, processes and data representing different design options and site characteristics into a full systems assessment. To achieve these ends, we have undertaken an analysis of the feasibility and safety of emplacing long-lasting copper containers within vault rooms (as opposed to deposition in boreholes in the floor of rooms) in a hypothetical volume of permeable plutonic rock where advective travel times from the vault to the biosphere are very short relative to those in the EIS case study.

Although we have not encountered such conditions at disposal-vault depths in our investigations at various research areas on the Shield, performance assessments done for the Swedish and Finnish nuclear waste disposal programs have considered these conditions in the crystalline rocks of the Fennoscandian Shield. We are not suggesting that such rock conditions might constitute favourable, desirable, or even acceptable conditions for an eventual disposal site on the Canadian Shield. Rather, the study is intended to illustrate the effectiveness of the in-room emplacement method and copper containers in inhibiting the release of contaminants from the vault.

The vault model for the present study simulates dissolution of used CANDU fuel in a geochemical environment, which evolves from an initial oxidative condition, caused by residual air and radiolysis, to an eventual steady-state anoxic condition. The model simulating the performance of copper containers is based on pinhole manufacturing defects and indefinite lifetime (i.e., no corrosion-induced failures). The in-room emplacement geometry is modelled as a line source representing the waste form, point sources representing pinholes in the defected containers randomly located in the vault, and concentric cylinders representing the buffer, backfill and excavation disturbed zone.

The geosphere model for the present study is more speculative than the one used for the EIS case study because it does not represent conditions we have encountered at any of our geologic research areas. We assume that the vault depth, the geometry of the geosphere model, and the arrangement of major fracture zones and rock mass domains surrounding the disposal vault are identical to those of the EIS case study. However, we assume much higher permeability and lower porosity conditions in the rock domain adjacent to the vault than the conditions observed at the URL and used in the EIS case study. As a result, the lower rock domain is not a diffusion-dominated barrier and the low-dipping fracture zone, LD1, is not the dominant advection pathway to the surface. The effects of geothermal gradient, vault heat and a water supply well on the groundwater flow field have been simulated and the implications on the long-term redox conditions in the vault have been assessed. The groundwater travel times from the disposal vault to the surface are up to 10 000 times shorter in this present geosphere model than in the model used for the EIS case study.

For this study, there is no advantage to constraining the location of the disposal rooms relative to LD1 as was done in the EIS case study. Thus the waste disposal rooms are located both below and above LD1 (i.e., on both the footwall and hangingwall sides of the fracture). The 50-m waste exclusion distance is retained but is relatively insignificant because advection is the dominant transport process in the permeable lower rock domain. Thermal restrictions and shielding requirements of the in-room emplacement option result in a reduction in the density of waste containers of roughly 50% relative to the borehole emplacement option of the EIS case study.

The biosphere model for the present study includes a number of changes, notably inclusion of additional radionuclides with shorter half-lives, inhalation pathways for animals, the most recent internal dose conversion factors of the International Commission on Radiological Protection (ICRP 1991a, b), geosphere dose limits for non-human biota, and updated values of model input parameters. Moreover, the part of the model representing the biosphere/geosphere interface was improved to account more fully for terrestrial discharge of radionuclides.

COMPARISON OF THE EIS CASE STUDY AND THE PRESENT STUDY

The key features of the EIS postclosure assessment case study and the present study are summarized as follows:

	EIS CASE STUDY	PRESENT STUDY
DESIGN CONSIDERATIONS		
Emplacement option	borehole	in-room
Vault area/depth	3.2 km^2/500 m	3.4 km^2/500 m
Fuel inventory: number of bundles	8.5 million	4.3 million
mass of uranium	162 000 Mg	82 000 Mg
Fuel Burnup	685 GJ/kg U	720 GJ/kg U
Fuel Cooling time	10 a	10 a
Number of bundles per container	72	72
Number of waste containers	118 700	60 100
Room locations	footwall of LD1	footwall and hangingwall of LD1
VAULT MODEL		
Vault model geometry	layered slabs	nested cylinders
Fuel dissolution model	thermodynamic	kinetic
Container shell material	Grade-2 Ti	high purity Cu
Container corrosion mechanisms	localized crevice and delayed hydride cracking	general corrosion and pitting
Fraction of containers failed instantly	10^{-3} to 10^{-4} (complete failure)	10^{-3} to 10^{-4} (pinhole failure)
Fraction of containers failed by 10^4 a	1.0	10^{-3} to 10^{-4}
Effective buffer thickness	0.25 m	1.48 m
Effective backfill thickness	1.4 m	0.76 m
Excavation disturbed zone	evaluated outside system model	evaluated explicitly within system model
GEOSPHERE MODEL		
Conceptual model of fracture zones and rock domains	URL area of WRA	URL area of WRA*
Permeability of rock domain surrounding vault	10^{-19} m^2	10^{-17} m^2
Effective transport porosity of rock domain surrounding vault	3×10^{-3}	10^{-3} to 10^{-5}
Minimum contaminant transport times from vault to biosphere	tens of thousands of years	tens of years
Rate-determining transport process	diffusion	advection
Maximum well depth	200 m	100 m

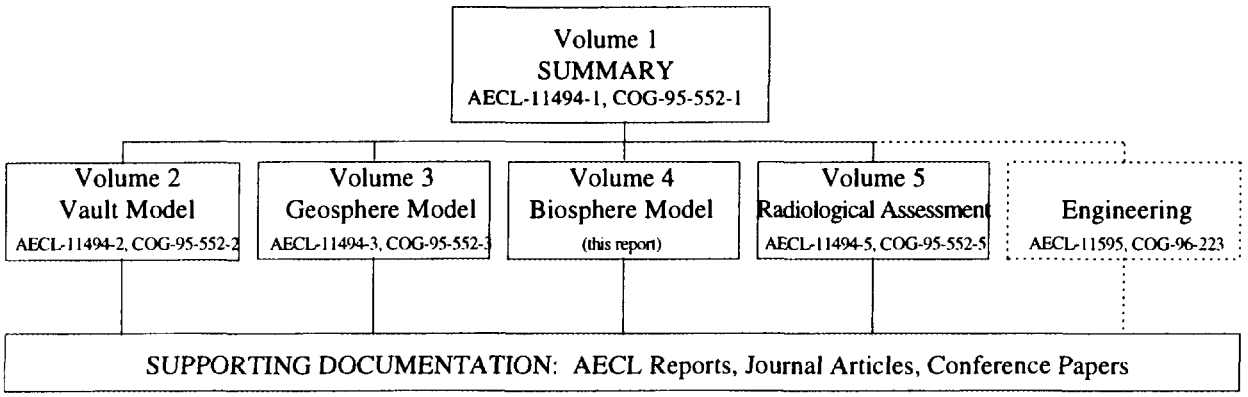
	EIS CASE STUDY	PRESENT STUDY
BIOSPHERE MODEL	BIOTRAC - typical of the Canadian Shield	BIOTRAC2 - modifications to improve the model and update the parameters
SYSTEMS ANALYSIS		
Computer Code	third generation code (SYVAC3-CC3-ML3)	prototype (PR4) of fourth generation code (SYVAC3-CC4)
Maximum estimated dose rate to a member of critical group up to 10^4 a	about 10^{-11} Sv/a	about 10^{-6} Sv/a
Time at which estimated dose rate reaches peak	> 10^5 a	about 10^4 a
Key radionuclides contributing to estimated dose rate up to 10^4 a	^{129}I ^{36}Cl ^{14}C	^{129}I, ^{36}Cl ^{14}C, ^{79}Se ^{90}Sr, ^{90}Y, ^{99}Tc
Principal safety feature	low permeability rock domain surrounding vault	long-lasting containers

* The conceptual model used for this present study does not represent a combination of conditions that we have encountered at any of our geologic research areas on the Shield. It has the same geometric arrangement of fracture zones and rock domains as was used in the EIS case study; however, the permeability of the rock domain surrounding the vault has been assumed to be 10^{-17} m^2. This permeability is 100 times greater than the value specified for the EIS case study, which was based on actual measurements within the lower rock zone at the URL.

The EIS case study, the study to identify a favourable vault location, and the present study illustrate the flexibility of AECL's disposal concept in taking advantage of the retention, delay, dispersion, dilution and radioactive decay of contaminants in a system of natural barriers provided by the geosphere and the hydrosphere and of engineered barriers such as the waste form, container, buffer and backfill. In an actual implementation, the engineered system would be designed for the geological conditions encountered at the host site.

HIERARCHY AND SCOPE OF DOCUMENTS FOR THE PRESENT STUDY

This study, presented in five main volumes and a number of supporting documents, is organized as follows:



Volume 1, Summary (Wikjord et al. 1996), provides an overview of this study and summarizes the design considerations and safety of in-room emplacement of CANDU used-fuel in long-lasting copper containers in permeable plutonic rock.

Volume 2, Vault Model (Johnson et al. 1996), describes and justifies the assumptions, model and data used to analyze the long-term behaviour of the engineered system (the near-field), including the waste form (used CANDU fuel), container shell (deoxidized, low-phosphorous copper), buffer (precompacted bentonite clay and silica sand), backfill (glacial lake clay and crushed rock), and excavation disturbed zone.

Volume 3, Geosphere Model (Stanchell et al. 1996), describes and justifies the assumptions, model and data used to analyze the transport of contaminants through permeable plutonic rock of the Canadian Shield, including the effects of a pumping well. The geological characteristics assumed in this study are not based on an integrated data set for any particular field research area.

Volume 4, Biosphere Model (Zach et al. 1996a (this report)), describes and justifies the assumptions, model and data used to analyze the movement of contaminants through the near-surface and surface environments and to estimate radiological impacts on humans and other biota.

Volume 5, Radiological Assessment (Goodwin et al. 1996), provides an estimate of long-term radiological effects of the hypothetical disposal system on human health and the natural environment, including an analysis of how uncertainties of the assumed site and design features affect system performance.

A separate engineering study (Baumgartner et al. 1996), shown by the dotted lines, is closely linked to this 5-volume series. It describes the conceptual design, technical feasibility, thermal and mechanical analyses, and project lifecycle for implementing an engineered system based on the in-room emplacement of copper containers. It is applicable to a broader range of geosphere conditions than assumed in the present study.

1. INTRODUCTION

1.1 BACKGROUND

AECL (Atomic Energy of Canada Limited) has developed a concept for disposal of Canada's nuclear fuel waste (AECL 1994a). The concept calls for the enclosure of used fuel or immobilized fuel reprocessing waste in metal containers placed in a vault 500 to 1000 m deep in plutonic rock of the Canadian Shield. The waste form of used fuel is a highly insoluble ceramic, contained in used fuel bundles, and that of reprocessing waste is a corrosion-resistant glass. The containers are surrounded by a clay-based buffer material and the vault with its disposal rooms are thoroughly sealed to retard the movement of nuclides from the waste into the geosphere. The rock surrounding the vault protects the vault physically and further retards the movement of nuclides to the surface environment. Thus, the concept involves a series of engineered and natural barriers to reduce the movement of nuclides and to ensure the long-term safety of the environment and humans. The disposal concept and its safety have been fully documented in an environmental impact statement (EIS) for scientific and for public reviews by a panel under the Canadian Environmental Assessment Agency (CEAA) (AECL 1994a). These reviews focus on the disposal concept only and not on the actual selection of a disposal site. Site selection can only start after the concept has been approved (Joint Statement 1981).

The postclosure environmental and safety assessment in the EIS is based on a case study involving the Whiteshell Research Area (WRA) in south-eastern Manitoba (Goodwin et al. 1994). The hypothetical vault has an area of 3.2 km^2, is 500 m below the rock surface and contains 8.5 million used fuel bundles (Preface). The WRA is not considered as a disposal site, it is simply used to demonstrate the safety of the disposal concept and the assessment methodology for it. This methodology includes a systems model composed of a vault model, a geosphere model and a biosphere model called BIOTRAC (BIOsphere Transport and Assessment Code). Together, these three models are designed to quantitatively predict the movement of nuclides from the vault to the surface environment for at least 10 000 years, as directed by the Atomic Energy Control Board (AECB 1987). The systems model addresses the main postclosure scenario - corrosion of containers, dissolution of waste, transport of nuclides in groundwater to the surface environment, and prediction of doses to humans (annual committed effective dose equivalent) and other biota (annual absorbed dose) for comparison with regulatory and other criteria. Doses are predicted for members of a critical group of humans, including aboriginal peoples and northerners (Zach et al. 1996b), and for four non-human generic target organisms. A code has been implemented for the systems model in SYVAC3 (Systems Variability Analysis Code - Generation 3) for Monte Carlo simulation to estimate the variability and uncertainty in the predicted doses (Goodwin et al. 1994).

The EIS postclosure assessment case study involves a series of assumptions regarding the vault location and layout, and a number of engineering options concerning the containers and their mode of emplacement in the vault (Preface, AECL 1994a, Goodwin et al.1994). In order to more fully demonstrate the flexibility and safety of the disposal concept, another postclosure assessment study was initiated (Goodwin et al. 1996). It is also based on the

WRA and a similar vault size and location as in the EIS postclosure assessment case study (Preface, Davison et al. 1994b). However, the general vault layout is changed to include disposal rooms on both sides of a fracture zone, LD1, that projects down to near the vault depth of 500 m (Goodwin et al. 1994). The containers are made of 2.54-cm thick copper rather than of 6.35-mm thick titanium, and they are directly emplaced in the disposal rooms rather than in holes drilled in the floor of such rooms (Johnson et al. 1994b, 1996). Copper is extremely corrosion resistant under anoxic conditions and so manufacturing defects in the form of pinholes would be the main concern in nuclide release rather than corrosion. On average, one in 5000 containers is assumed to have a pinhole and some nuclides are assumed to escape from defective containers immediately upon vault closure (Johnson et al. 1996). The vault capacity is reduced to 4.3 million used fuel bundles or about 60 000 containers. These changes necessitate modifications in the detailed vault layout, and in the clay-based buffer and sealing materials (Baumgartner et al. 1996, Johnson et al. 1996).

The permeability of the rock in the WRA at vault depth is very low and this greatly retards the movement of nuclides to the biosphere (Davison et al. 1994b). This is reflected in the EIS postclosure assessment case study. For the present study, we have increased this permeability to reflect conditions that might be encountered elsewhere (Preface, Stanchell et al. 1996). This in turn increases groundwater velocity and reduces the transit time of nuclides through the geosphere to the surface environment so that there needs to be more reliance on the engineered barriers for the containment of nuclides.

All these changes have required substantial modifications in the vault model and the geosphere model (Stanchell et al. 1996, Johnson et al. 1996). This is also true for the BIOTRAC model.

The BIOTRAC model, which has been documented in detail by Davis et al. (1993a) and by Zach et al. (1994), is a generic model that reflects conditions over much of the Shield (Section 1.3). This is especially so for five of its six submodels, but not for the geosphere/biosphere interface submodel (Figure 1-1). This submodel needs to have site-specific aspects in order to smoothly interface with the geosphere model which is based on the WRA. The interface submodel is affected by the exact vault size and location, which, together with geological, hydrological and environmental settings, determine the locations where groundwater with dissolved nuclides from the vault might reach the surface environment. Because the vault size and location for the present study and the EIS postclosure assessment case study are the same, the interface submodel does not need to be modified. However, there are some changes to the geosphere model that affect the interface submodel and that make it more flexible. Changes to the BIOTRAC model are related to model improvements, and to the availability of more or better data for defining parameter values and probability density functions (PDFs). Furthermore, there is a need to expand the BIOTRAC model by including ^{36}Cl, a nuclide of relatively high importance in nuclear fuel waste disposal (Johnson et al. 1995). We have also added ^{137}Cs, ^{239}Np and ^{243}Am to the suite of postclosure assessment nuclides.

1.2 REPORT OBJECTIVE AND OUTLINE

The objective of this report is to document all the changes made to the BIOTRAC model for the present study. We have not documented the entire model again but only the changes made to the BIOTRAC model, as used for the EIS postclosure assessment case study and as documented by Davis et al. (1993a). Thus, this report and the report by Davis et al. (1993a) together document the BIOTRAC model for the present study. To facilitate this, the two reports have a similar structure. The revised version of the BIOTRAC model introduced in this report will be referred to as BIOTRAC2, as discussed in Section 8.

In Section 1.3, we provide a brief general overview that applies to both the BIOTRAC and BIOTRAC2 models to put the changes presented in Chapters 2 to 7 into a broader perspective. Chapter 8 is concerned with the effects of the changes on model predictions, and Chapter 9 provides the summary and conclusions. To further assist the reader, we provide in Appendix A a list of all the acronyms, names and abbreviations, and in Appendix B a list of all the symbols used in the report. Furthermore, Appendix C includes corrections of a few errors made in the documentation of the BIOTRAC model (Davis et al.1993a). Assessment results for the present study, which are based on the BIOTRAC2 model, are presented by Goodwin et al. (1996).

1.3 OVERVIEW OF THE BIOTRAC MODEL

This overview of the BIOTRAC model is very general so that it applies to both the BIOTRAC and the BIOTRAC2 models. The overview is designed to put the changes presented in Chapters 2 to 7 into a broader perspective.

The BIOTRAC model was specifically developed to evaluate the postclosure environmental and health impacts of the concept of disposal of Canada's nuclear fuel waste (Davis et al. 1993a, AECL 1994a). The model is largely generic for much of the Canadian Shield because site selection can only proceed once the disposal concept has been approved (Joint Statement 1981). The model is applicable for up to 10 000 years into the future, the quantitative assessment period specified by the AECB (1987). This period is assumed to be free from continental glaciation.

The BIOTRAC model and its numerous parameter values are based on a large amount of literature data (Davis et al. 1993a). 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 through the biosphere and their effects on various biota (Zach 1985a, Zach et al. 1987, Davis et al. 1993a). Most of this research has focussed on model structure, parameter values and model validation, as related to the Shield.

This research has involved interactions with universities and nuclear fuel waste disposal programs in many other countries. In areas where knowledge is still limited, the BIOTRAC

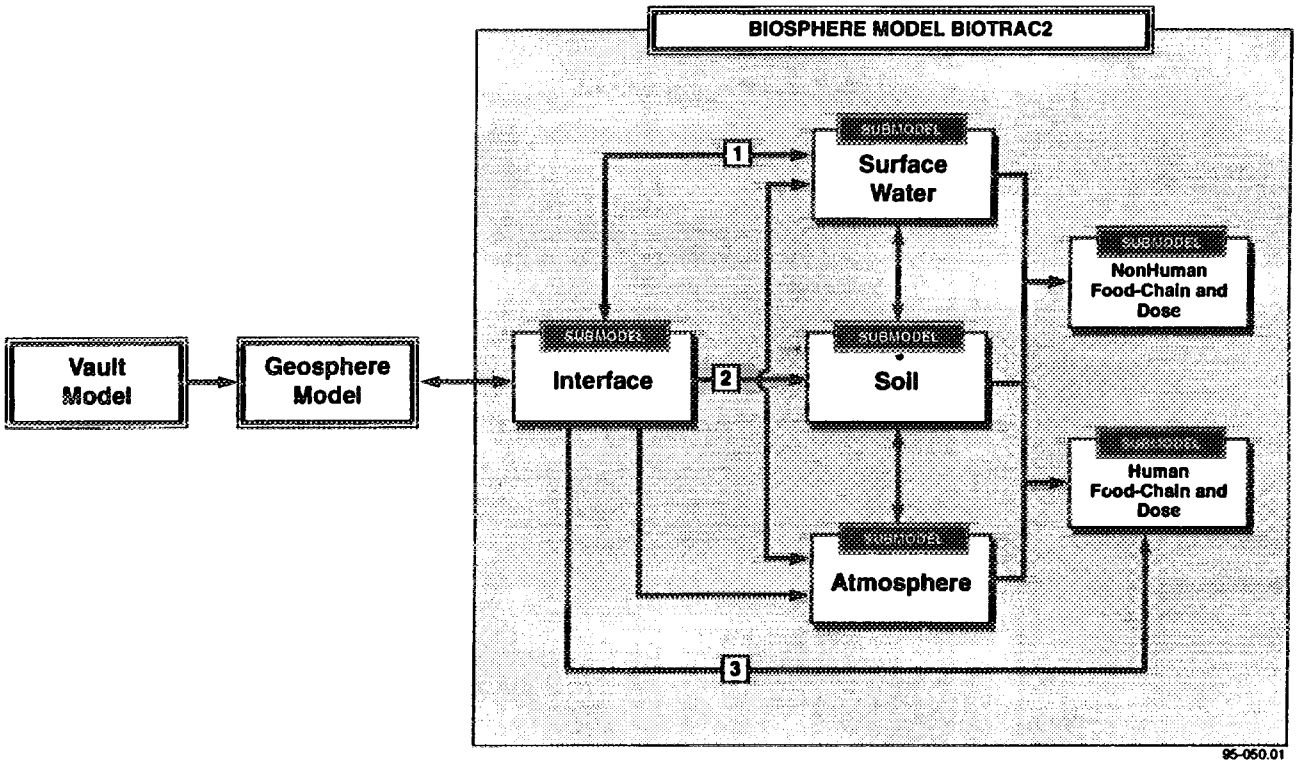


Figure 1-1. Diagram of the Three Main Assessment Models (Vault, Geosphere and Biosphere) and of the Main Nuclide Transfers Among the Six Submodels of the Biosphere Model (Interface, Surface Water, Soil, Atmosphere, Human Food-Chain and Dose, and Non-Human Food-Chain and Dose). Discharges from the geosphere to the biosphere are: (1) aquatic, (2) terrestrial and (3) bedrock well.

model makes conservative assumptions that tend to overpredict consequences (environmental concentrations and doses) to ensure safety (Davis et al. 1993a).

The important biosphere processes and pathways were identified through a rigorous scenario analysis procedure for inclusion in the BIOTRAC model. These processes and pathways are modelled probabilistically using a systems variability approach (Goodwin et al. 1994, 1996). The values of many model parameters are represented by PDFs to allow for the spatial variability that occurs across the Shield. Environmental fluctuations or changes over time are also primarily accounted for through the PDFs. At the beginning of each simulation, a set of parameter values is selected from these PDFs so that many different biospheres can be considered that might occur at a given site on the Shield in the future.

The BIOTRAC model is driven by the nuclides released from the geosphere (Figure 1-1). The nuclides are traced through the surface environment to predict various environmental concentrations, and radiological doses for humans (annual committed effective dose equivalent) and other biota (annual absorbed dose). These quantities can be compared with regulatory and other criteria to assess the safety of the disposal concept. Humans are represented in the BIOTRAC model by the critical group and by International Commission on Radiological Protection reference man (ICRP 1975, Davis et al. 1993a, Zach et al. 1994). The critical group is located where nuclides are predicted to discharge from the geosphere into the biosphere and where dilution is at a minimum. Furthermore, the critical group is totally self-sufficient and dependent for all of its needs on the local, potentially contaminated environment. Although this lifestyle is very unlikely, it ensures that consequences are not underpredicted. Non-human biota are represented by several generic target organisms - a terrestrial plant, a mammal, a bird and a fish (Amiro and Zach 1993). They share the environment with the critical group and so are also exposed in the same conservative way.

The six submodels of the BIOTRAC model are fully integrated with each other and with the geosphere model (Figure 1-1).

The geosphere/biosphere interface submodel is the only submodel with site-specific attributes so that it smoothly interfaces with the geosphere model, which is based on the WRA (Chapter 2, Stanchell et al. 1996). In essence, this means that some of the parameter values and PDFs are restricted to conform to the WRA. The submodel considers several discharge zones at which nuclides are predicted to enter the biosphere. Each of these discharge zones has an aquatic and a terrestrial portion. The submodel also includes a bedrock well that can be used as a source of relatively undiluted water by the critical group.

The surface water submodel is time-dependent and includes a typical Shield lake that can become contaminated by nuclides released through the compacted sediment from the geosphere (Bird et al. 1992, 1993). Once in the water, nuclides may be deposited to the mixed sediment, or lost through lake flushing, gaseous evasion and radioactive decay. They may also be transferred to the land through irrigation.

The time-dependent soil submodel is derived from a very detailed mechanistic model, which is represented in the BIOTRAC model by a simpler regression model (Sheppard 1992, Davis et al. 1993b). Several soil types are considered. The soil profile can become contaminated through terrestrial discharge to the bottom of the soil profile and through irrigation on top with contaminated water. Nuclides can move through the soil profile through capillary rise and leaching, and they may be lost through drainage, gaseous evasion and radioactive decay. Nuclide concentrations are predicted for several fields, including a garden and a forage field for animals.

The atmosphere submodel assumes steady-state conditions and considers nuclide suspension from the lake and soil as well as from burning of biomass (Amiro and Davis 1991, Amiro 1992a). The model also allows for atmospheric dispersion which can result in the loss of

nuclides. Suspended nuclides can be deposited to a variety of underlying surfaces. The submodel considers both indoor and outdoor air.

The human food-chain and dose submodel also assumes steady-state conditions and it includes all the important internal and external pathways that might lead to radiation exposure of the critical group, based on the predicted nuclide concentrations in the surface water, soil and air (Zach and Sheppard 1991, 1992). For most of the nuclides, transfer is handled through transport models. However, alternative specific-activity models are also used. They take into account the special attributes of radionuclides such as ^{3}H, ^{14}C, ^{36}Cl and ^{129}I. For the last three of these nuclides, specific-activity models based on nuclide concentrations in groundwater discharging to the biosphere are used to establish upper dose limits.

The non-human food-chain and dose submodel closely reflects the model for humans. However, the submodel is concerned with predicting doses for the four generic target organisms rather that for humans (Amiro and Zach 1993). The non-human biota rely only on the lake water and the forage field for their survival.

Some of the BIOTRAC submodels have been validated experimentally to ensure they provide realistic predictions. However, model validation over thousands of years is clearly impossible. Therefore, a variety of other methods were used to establish model credibility, including international code comparison studies and peer review of many of the supporting data (Davis et al. 1993a). Furthermore, conservative assumptions were used whenever needed to ensure consequences are not underpredicted.

2. GEOSPHERE/BIOSPHERE INTERFACE SUBMODEL

2.1 MODEL CHANGES

Because the vault size and location in the WRA, and the general vault layout remain essentially the same as in the EIS postclosure assessment case study, the interface model remains largely unchanged (Davis et al. 1993a). Below we briefly discuss minor changes to this model associated with changes in the number, size and location of the discharge zones, where nuclides released from the geosphere are predicted to enter the biosphere. We also present details of a change in the GEONET geosphere model (Stanchell et al. 1996) that directly affects the BIOTRAC model. The GEONET model consists of a simplified network of segments and nodes that describe the flowpaths of nuclides released from the disposal vault through the geosphere to the biosphere (Figure 1-1).

2.1.1 Discharge Zones

In the EIS postclosure assessment case study, the GEONET model included three discharge points to the biosphere, other than the bedrock well (Davison et al. 1994b). These points

corresponded to three discharge zones: Boggy Creek south, Boggy Creek north and Pinawa Channel. Because of the geohydrological changes related to higher rock permeability (Chapter 1), the GEONET model for the present study includes four discharge points (Stanchell et al. 1996). They correspond to the Boggy Creek south, Boggy Creek north, Pinawa Channel south and Pinawa Channel north discharge zones (Figure 2-1). Each of the four discharge zones has a terrestrial portion, δ (unitless), and an aquatic portion, $1 - \delta$, as determined in the BIOTRAC model. However, the size of the actual discharge zones is determined by particle tracking through the geosphere (Stanchell et al. 1996).

In the EIS postclosure assessment case study, the Boggy Creek south discharge zone was by far the most important because it is directly linked to the fracture zone LD1 that extends down to near the vault. This linkage also exists in the present study, but it is far less important in terms of nuclide discharge to the biosphere because of the increased rock permeability. Thus, all the discharge zones are predicted to be active in nuclide discharge during the 10 000 year quantitative assessment period (AECB 1987) and not just the Boggy Creek south discharge zone.

In aquatic discharge, nuclides pass into the lake through overburden and a layer of compacted sediment; in terrestrial discharge, they pass through the overburden to the bottom of the unsaturated soil zone at the water table. In addition to the four discharge zones, nuclides can also discharge through a well. If the well is deep enough to penetrate to bedrock, nuclide concentrations in well water are determined jointly by the geosphere model and by the BIOTRAC model, otherwise they are based on the lake water concentrations determined by the BIOTRAC model (Davis et al. 1993a).

2.1.2 Geosphere/Biosphere Interface

Discharge points in the GEONET model used for the EIS postclosure assessment case study did not include a separate node for terrestrial discharge to the soil; only nodes for aquatic discharge to the lake were included (Davison et al. 1994b). Although we modelled terrestrial discharge, it was awkward to include the flow of secular equilibrium progeny nuclides to the biosphere. These progeny nuclides are not modelled separately in the GEONET model because their radioactive half-lives are relatively short compared to the geosphere transit times. The flow of these progeny nuclides is initiated at the geosphere/biosphere interface, assuming secular equilibrium with the precursor nuclides so that all the nuclides with a half-life of one day or more are explicitly included in the biosphere (Davis et al. 1993a). With the introduction of separate terrestrial nodes, this can now be readily accomplished in the same way as for the aquatic nodes.

For the present study, we have introduced a separate terrestrial node for each of the four discharge zones to complement the existing aquatic nodes. Thus, each discharge zone has two nodes - one terrestrial and one aquatic. The values for δ, and those for the overburden and compacted sediment characteristics related to nuclide sorption, remain the same as for the EIS postclosure assessment case study (Davis et al. 1993a, Davison et al. 1994b). We also

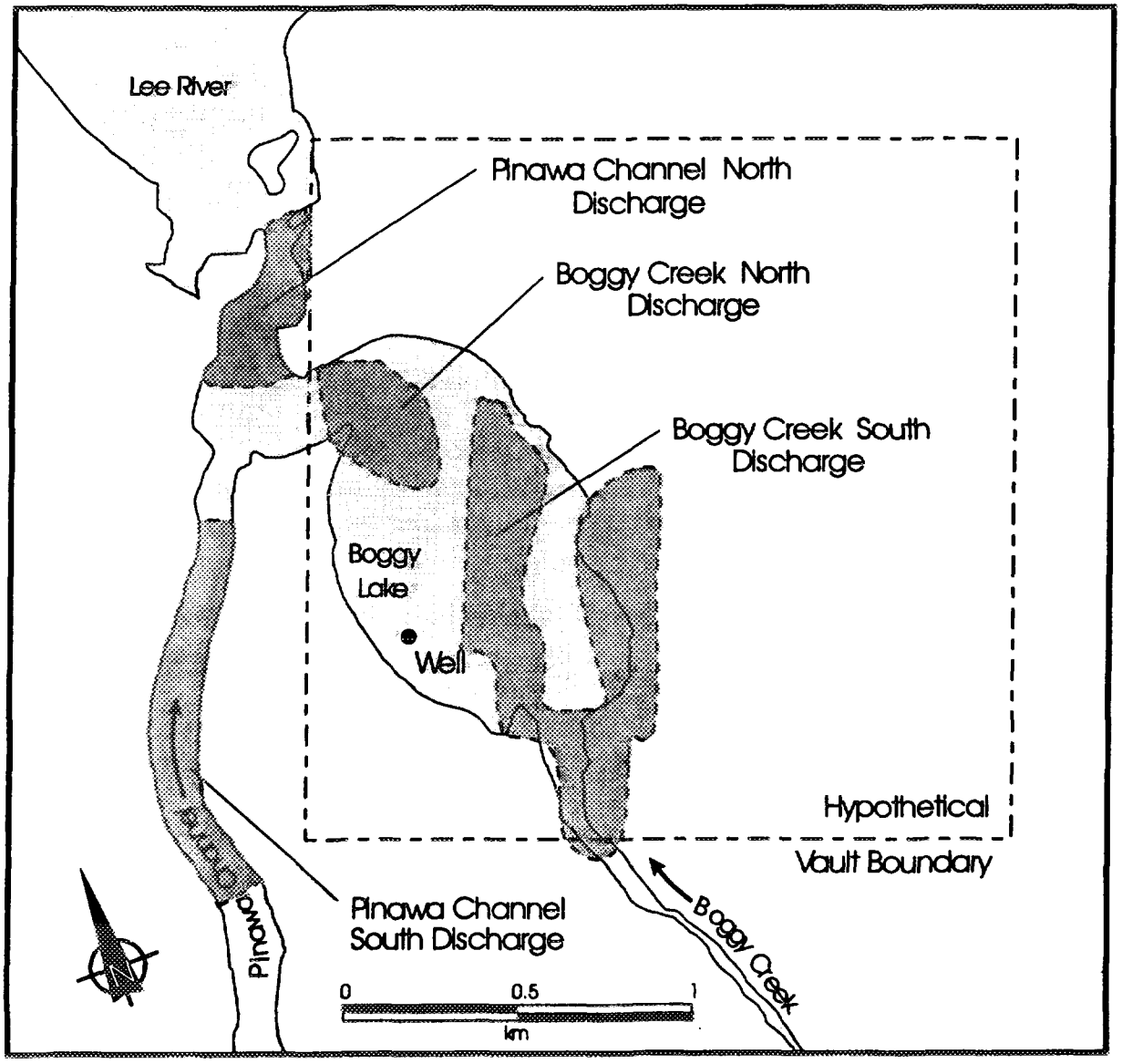


Figure 2-1. The Four Zones (Shaded Areas) where Groundwater Contaminated with Nuclides from the Hypothetical Vault Below is Predicted to Discharge to the Biosphere at Boggy Lake in the Whiteshell Research Area. The location of the bedrock well reflects its conservative placement in the potentially most contaminated groundwater associated with the fracture zone LD1.

retain the feature of directing all the geosphere nuclide discharge to the lake even though some of it is duplicated in discharge to the soil to implicitly account for subsequent nuclide movement through runoff from the soil to the lake (Davis et al. 1993a). This creates some nuclide mass but it greatly simplifies modelling.

Although not used in the present study, the introduction of separate terrestrial nodes greatly increases the flexibility of the geosphere/biosphere interface model. The number of discharge zones can now be varied because it is no longer fixed at three, δ can be varied because it can have different values for each of the discharge zones, and overburden and compacted sediment characteristics can also be varied because sorption characteristics can have different values for the various nodes. This flexibility is important for realistically interfacing the GEONET and BIOTRAC models in other studies or assessment situations.

2.2 PARAMETER VALUE CHANGES

There are no parameter value changes, but we have added a new PDF to quantify sorption of ^{36}Cl in compacted sediment as groundwater discharges from the geosphere into the biosphere. We now fully include ^{36}Cl in the BIOTRAC model.

2.2.1 Compacted Sediment Partition Coefficient, Kd^i_{ds} (m^3 water·kg^{-1} dry sediment)

This element-specific parameter is the ratio of the nuclide concentration in compacted sediment solids to the concentration in the corresponding pore water. Thus, Kd^i_{ds} describes the sorption of nuclides in compacted sediment, which is considered in the geosphere model (Davison et al. 1994b). However, Kd^i_{ds} values are also used in the BIOTRAC model to determine nuclide concentrations for sediment used as soil. Kd^i_{ds} values for all the nuclides have been established by Bird et al. (1992) as part of the surface water model. These values are the same as those established for nuclide sorption in organic soil (Sheppard 1992, Davis et al. 1993a), but no value for assessing ^{36}Cl has been established thus far. Low Kd^i_{ds} values are conservative for the lake water pathways because they correspond to low sorption and increased nuclide mobility. However, high Kd^i_{ds} values are conservative when lake sediment is used as or becomes soil. With "conservative" we mean that predicted nuclide concentrations, and especially predicted doses for humans and other biota are overestimated rather than underestimated.

The PDF for the organic soil partition coefficient for chlorine, Kd^{Cl}, is documented in Section 4.2.1. This PDF reflects aerobic soil conditions. Following previously established procedures (Bird et al. 1992), we also use this PDF for compacted sediment. Thus, Kd^{Cl}_{ds} is lognormally distributed with a geometric mean (GM) of 1.13×10^{-2} m^3 water·kg^{-1} dry sediment and with a geometric standard deviation (GSD) of 10.0. This PDF applies to both aerobic and anaerobic sediments because too few data are available to establish a separate PDF for anaerobic conditions, and because the movement of chlorine is not very sensitive to oxic conditions. The specified PDF is not subject to any truncation or correlation.

3. SURFACE WATER SUBMODEL

3.1 MODEL CHANGES

The model remains unchanged as described by Bird et al. (1992, 1993) and Davis et al. (1993a), and as summarized in Section 1.3.

3.2 PARAMETER VALUE CHANGES

We have added a new PDF concerned with the transfer of ^{36}Cl from lake water to mixed sediment, and updated the same PDF for ^{129}I in the light of new data.

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

This element-specific parameter defines the fraction of a nuclide in the water column that is transferred to the sediment per year. The parameter and its derivation are discussed in detail by Bird et al. (1992) and by Davis et al. (1993a) for all the nuclides, except for chlorine. High α^i values lead to high nuclide concentrations in sediment, but low α^i values lead to high lake water concentrations. Low α^i values are conservative for the lake water pathways because they correspond to high doses for humans and for other biota. High α^i values are conservative when lake sediment is used as or becomes soil.

Several studies have published data for Shield lakes on chlorine transfer that can be used to calculate α^{Cl} values, using the mass balance Equation (15) of Bird et al. (1992). As shown in Table 3-1, the calculated α^{Cl} values are quite variable. Many of them are negative, which indicates a net transfer of chlorine from sediment to the water column. Since the BIOTRAC model assumes all the ^{36}Cl discharged from the geosphere enters the water column of the lake (Davis et al. 1993a), we have set these negative values to 0.001 a^{-1}. This procedure of Bird et al. (1992) is conservative for the lake sediment pathways because it allows for the potential build-up of ^{36}Cl in the mixed sediment, which can be transformed to soil for supporting a variety of terrestrial food chains in the BIOTRAC model. The α^i values for all the nuclides in the BIOTRAC model are lognormally distributed. On the basis of the 13 values in Table 3-1, we have established a lognormal PDF for chlorine with a GM of 0.005 a^{-1} and a GSD of 12. As in the case of all the other nuclides, this PDF is not subject to any truncation or correlation.

The α^i PDF for ^{129}I in the BIOTRAC model was based on a single observed value, and assigned a GM value of 2.3 a^{-1} and our default GSD of 6.9 (Bird et al. 1992). We have now completed a study on the behaviour of iodine in two Shield lakes of differing trophic states (Bird et al. 1995a), and so can revise the PDF for ^{129}I. The revised PDF remains lognormal as before, but with a GM of 1.6 a^{-1} and a GSD of 2.2. This PDF is consistent with the data from Kolehmainen et al. (1969) for a Finnish lake and those from an enclosure study in Perch

Lake, Ontario (Milton et al. 1992). It is also consistent with the GM values calculated by Equation (21) of Bird et al. (1992), and with Kd values measured in recent laboratory studies (Bird et al. 1995b, Stephenson and Motycka 1995, Bird and Schwartz 1996).

TABLE 3-1

CHLORINE SEDIMENT TRANSFER RATES, α^{Cl}, FOR SHIELD LAKES CALCULATED FROM LITERATURE DATA

α^{Cl} (a^{-1})	Lake	Data Source
-0.350	Batchwana	Jeffries et al. (1988)
-1.021	Batchwana	"
0.396	Wishart	"
0.723	Wishart	"
-1.260	Little Turkey	"
-0.887	Little Turkey	"
0.002	Turkey	"
-0.012	Turkey	"
0.067	ELA 239*	Schindler et al. (1976)
0.058	ELA 239*	"
-0.473	Panther	Galloway et al. (1983)
-1.951	Sagamore	"
-0.108	Woods	"

* Experimental Lakes Area, Ontario.

Note: Negative α^{Cl} value are set to 0.001 a^{-1} for establishing the PDF for the BIOTRAC model, as explained in Section 3.2.1.

4. SOIL SUBMODEL

4.1 MODEL CHANGES

The model remains unchanged as documented by Sheppard (1992) and Davis et al. (1993a, 1993b), and as summarized in Section 1.3.

4.2 PARAMETER VALUE CHANGES

The inclusion of ^{36}Cl in the BIOTRAC model necessitates establishment of an additional PDF concerned with soil sorption (Section 4.2.1). We have also re-examined the loss of gaseous nuclides from soil and new experimental data have allowed revision of the PDFs for ^{14}C and ^{129}I (Section 4.2.2).

4.2.1 Soil Partition Coefficient, Kd^i (m^3 water·kg^{-1} dry soil)

This coefficient represents the ratio of the nuclide concentration on soil solids to the concentration in the corresponding pore water (Sheppard and Thibault 1990). Thus, it quantifies the sorption of nuclides in soil. The BIOTRAC model requires Kd^i values for four soil types - sand, loam, clay and organic. A complete set of values has been established for all the nuclides except for chlorine (Sheppard 1992, Davis et al. 1993a). Generally speaking, low Kd^i values are conservative because only nuclides dissolved in water can disperse and become available for root uptake by plants and so enter the food chain.

The available data indicate that Kd^i values for many nuclides are lognormally distributed with a GSD of 10.0 (Sheppard 1992). We assume this is also true for chlorine. In order to establish GM values for the four soil types, we used the regression equation developed by Baes et al. (1984) for loam and by Sheppard and Thibault (1990) for the other three soil types. These equations predict Kd^i from the plant/soil concentration ratio, Bv^i, which for chlorine has a value of 18.0 Bq·kg^{-1} wet biomass per Bq·kg^{-1} dry soil (Section 6.2.1.1). They also use a correlation coefficient, r, of -0.7 between Kd^i and Bv^i. With these regressions and parameter values, we calculated a GM value for sand of 8.0×10^{-4}, for loam of 2.5×10^{-4}, for clay of 4.4×10^{-3}, and for organic of 1.1×10^{-2} m^3 water·kg^{-1} dry soil. In units of L water·kg^{-1} dry soil, the corresponding values for use in some of the equations in the soil model are 8.0×10^{-1}, 2.5×10^{-1}, 4.4×10^{0} and 1.1×10^{1}. None of the PDFs for Kd^i are subject to truncation. Furthermore, Kd^i and Bv^i are always correlated at -0.7 in the BIOTRAC model because sorbed nuclides are unavailable for uptake by plants (Davis et al. 1993a).

4.2.2 Gaseous Evasion Rate from Soil, η_s^i (a^{-1} or s^{-1})

This evasion rate describes the fraction of a gaseous nuclide in the root-zone of the soil that is lost to the atmosphere per year. Low η_s^i values are conservative because they lead to high soil concentrations and doses to humans and to other biota through food-chain transfer. We have previously defined η_s^i values for ^{14}C, ^{129}I and ^{79}Se, nuclides considered as gases in the soil model (Davis et al. 1993a). All the other nuclides treated in the soil model, including ^{36}Cl, are considered non-gaseous. We have re-examined the gaseous soil evasion rates for ^{14}C and ^{129}I in the light of new data. In the soil model, these rates are expressed in a^{-1}.

On the basis of the results of recent studies of soil degassing, we have confirmed that the PDF for ^{14}C is lognormal, but with a GM of 13.6 a^{-1} and a GSD of 3.3 (Sheppard et al. 1994a). The corresponding previous values were 8.8 a^{-1} and 10.0. Thus, the GM has been slightly increased and the GSD has been greatly decreased. This decrease represents a reduction in uncertainty. Sheppard et al. (1994a) also show that soil normally contains elevated levels of carbon dioxide (CO_2), mainly from plant root and microbial respiration. The loss of CO_2 is controlled primarily by gaseous diffusion. Soil pH and porosity are of major importance in degassing, and soil temperature, moisture and organic matter content are of lesser importance. There is seasonal variation because soil freezing substantially decreases degassing. Lower winter values are included in the revised PDF. In terms of increased GM, the revised PDF decreases ^{14}C soil concentrations and increases the corresponding air concentrations calculated by the BIOTRAC model. However, these changes are not clear cut because of the confounding influence of the reduced GSD.

Results from volatilization studies also confirm that the PDF for iodine is lognormal, but with a GM of 2.1 x 10^{-2} a^{-1} and a GSD of 3.0 (Sheppard et al. 1994b). Thus, both the GM and the GSD have been reduced from the previous values of 3.2 x 10^{-2} a^{-1} and 10.0 respectively. Again, the substantial reduction in the GSD represents a reduction in uncertainty. The revised PDF includes volatile molecular iodine, other inorganic iodine species, and also volatile organic methyliodide. Compared to the old PDF, the revised PDF decreases ^{129}I soil concentrations and increases air concentrations as far as the increased GM is concerned. However, the reduced GSD also influences these trends.

In the atmosphere submodel (Chapter 5), gaseous evasion rates are used to calculate air concentrations, but these rates appear there in s^{-1}, rather than in a^{-1}. In these units, the GM value is for ^{14}C is 4.3 x 10^{-7} s^{-1} and the GSD value remains at 3.3. For ^{129}I, the GM value is 6.7 x 10^{-10} s^{-1} and the GSD value remains unchanged at 3.0. The PDFs remain lognormal for both nuclides.

5. ATMOSPHERE SUBMODEL

5.1 MODEL CHANGES

The model remains unchanged as documented by Amiro and Davis (1991), Amiro (1992a) and Davis et al. (1993a), and as summarized in Section 1.3.

5.2 PARAMETER VALUE CHANGES

We have re-examined the release of nuclides from domestic water to indoor air and re-defined several related PDFs to account for all the diverse nuclides considered in the BIOTRAC model (Section 5.2.1). In this, we have added ^{36}Cl. We have also corrected a previous conversion error (Section 5.2.2) and truncated a PDF for ^{129}I (Section 5.2.3).

5.2.1 Release Fraction, $RELFRAC^i$
(unitless)

This parameter defines the fraction of a nuclide in the domestic water supply of the critical group that is released to the indoor air. Depending on the exact simulation, this supply is derived from either the lake or the well. Previously, $RELFRAC^i$ had a value of 1.0 for ^{14}C and ^{129}I, and a value of zero for all the other nuclides (Amiro 1992a, Davis et al. 1993a). In conjunction with the inclusion of ^{36}Cl in the BIOTRAC model, we have re-examined this parameter to establish more realistic values for ^{222}Rn (radon), ^{14}C and for all the other nuclides, including ^{36}Cl and ^{129}I. High $RELFRAC^i$ values are conservative because they correspond to high doses for humans. Note that $RELFRAC^i$ is not correlated with any of the other BIOTRAC model parameters and none of the PDFs established below is subject to truncation.

Gesell and Pritchard (1980) found that 52% of the radon entering a house with the domestic water was liberated to the air. This included contributions from all sources such as showers, washing, toilets and drinking. This percentage is a good average value, but a range of values is possible because Gesell and Pritchard (1980) measured a 90% release for laundry and only a 30% release for toilets. Therefore, we have adopted a triangular PDF in the BIOTRAC model for radon with a lower limit of 0.30, an upper limit of 0.90, and a peak at 0.52 (Table 5-1).

Carbon-14 could most easily be released as CO_2, but the amount of free CO_2 in water depends on the pH. Stephenson and Motycka (1994) measured the release of ^{14}C from stirred flasks with river water at neutral pH over a period of 7 to 8 d. They observed that between 25 to 100% of ^{14}C was released during this period. Household water residence times are likely much less than 7 to 8 d and so their values would represent conservative estimates for households. However, some water uses such as showers could release substantial amounts of ^{14}C over shorter periods. Based on these data and considerations, we have established a uniform PDF for ^{14}C ranging from 0.25 to 1.0 (Table 5-1).

All the remaining nuclides included in the BIOTRAC model (Davis et al. 1993a) would not be readily released from water to indoor air unless present in a chemically volatile form, but ^{36}Cl and ^{129}I deserve further considerations. At the pH and Eh of typical domestic water, chlorine and iodine would not be present as volatile Cl_2 or I_2. Similarly, volatile methylated forms of these two nuclides are rare in most domestic water sources. Stephenson and Motycka (1994) estimated the release of iodine from stirred flasks was about 0.5% of the release of ^{14}C. Therefore, gaseous evasion can be expected to be relatively unimportant for all the other nuclides, including ^{36}Cl and ^{129}I.

TABLE 5-1

PROBABILITY DENSITY FUNCTIONS FOR THE RELEASE FRACTION, $RELFRAC^i$, FOR NUCLIDE TRANSFER FROM DOMESTIC WATER TO INDOOR AIR*

Nuclide	Distribution Type	Lower Limit	Upper Limit	Peak
^{222}Rn	Triangular	0.30	0.90	0.52
^{14}C	Uniform	0.25	1.0	-
Remainder	Log-uniform	0.00052	0.052	-

* $RELFRAC^i$ is unitless.

Humidifiers could inject nuclides into the atmosphere. Johnston and Amiro (1993, 1994) found that all of the nuclides tested are released from ultrasonic humidifiers at the same rate as the water. They calculated a maximum $RELFRAC^i$ value based on continuous use of a humidifier by people to increase indoor humidity by 50%. This value is conservative because it overestimates humidifier use in the summer and also likely in the winter. The corresponding water flux rate from this use is 6.8 $m^3 \cdot a^{-1}$ (Johnston and Amiro 1994). This compares to a domestic water use of 130 $m^3 \cdot a^{-1}$ per person (Davis et al. 1993a). Since one person represents the smallest critical group in the BIOTRAC model, the maximum fraction of the domestic water released in a humidifier would be 0.052. This fraction would be lower with larger critical groups because the domestic water use would increase. The 0.052 value corresponds to $RELFRAC^i$ because only humidifiers could release the remaining non-volatile nuclides into the air. However, this value is likely too high because not all the households representing the critical group would use humidifiers and none of them would use humidifiers to the extent considered in our calculations to avoid condensation on cold surfaces in both summer and winter.

Based on these data and considerations, $RELFRAC^i$ has a log-uniform PDF ranging from 0.00052 to 0.052 for all nuclides except for radon and ^{14}C, as indicated above (Table 5-1). This log-uniform PDF represents a 100-fold range, weighted towards lower values. The PDF includes the value of 0.005 for iodine measured by Stephenson and Motycka (1994) near the middle of the range.

5.2.2 Infiltration Rate, INFILT (s^{-1})

Building air infiltration rates are required to estimate the mixing volume of indoor air for nuclides released from domestic water (Davis et al. 1993a). We have previously calculated an INFILT value of 0.0058 s^{-1} (Amiro 1992a), which was then used for the BIOTRAC model. This value is incorrect (Appendix C) because of a unit conversion error. The correct value is 9.7 x 10^{-5} s^{-1}, which we have adopted now for the BIOTRAC model.

5.2.3 Aquatic Iodine Mass-Loading Parameter, AIML (m^3 water·m^{-3} air)

AIML is the ratio of the concentration of gaseous iodine in air above a lake surface to the iodine concentration in the water below. In the BIOTRAC model, it is represented by a lognormal PDF with a GM of 1.3 x 10^{-5} m^3 water·m^{-3} air and a GSD of 6.3 (Davis et al. 1993a). This PDF is retained but we are establishing now an upper truncation limit at 1.0 m^3 water·m^{-3} air. This is a conservative physical limit at which the ^{129}I air concentration is equal to the water concentration.

6. HUMAN FOOD-CHAIN AND DOSE SUBMODEL

6.1 MODEL CHANGES

The human food-chain and dose model included in the BIOTRAC model is very complete (Zach and Sheppard 1991, 1992, Davis et al. 1993a). However, it does not include the inhalation pathway for terrestrial animals. To avoid unnecessary model complexity, this pathway was excluded because human inhalation is a much more important dose contributor for people than animals inhalation (Zach 1985b). To make the BIOTRAC model more comprehensive, we have now included animals inhalation pathway (Section 6.1.1). This pathway might be most important when considering doses to terrestrial animals (Section 7.1.1). We have also introduced a groundwater dose limit for ^{36}Cl (Section 6.1.2) and re-examined the groundwater dose limits for ^{14}C and ^{129}I previously established in the BIOTRAC model (Section 6.1.3). All the exposure pathways included in the updated BIOTRAC model are shown in Figure 6-1, and Figure 6-2 shows all the interfaces of the human food-chain and dose submodel with some of the other submodels.

6.1.1 Animals Inhalation Pathway

Terrestrial animals may become contaminated by inhaling air containing nuclides. The nuclides may become deposited in the respiratory tract, absorbed into the body and transferred to human food products, such as meat, milk and eggs. The human intake of nuclide i through animal inhalation is given by

$$\left(E_j^i\right)_{AA} = \left[\left(C_a^i\right)_o \cdot Qa_j \cdot Fl_j^i \cdot \exp\left(-\lambda^i \cdot tha_j\right)\right] \cdot U_j \qquad (6.1)$$

where $(E_j^i)_{AA}$ is the human intake of nuclide i via food types j = TE MEAT, TE MILK and TE BIRD contaminated by inhalation ($Bq \cdot a^{-1}$),

$(C_a^i)_o$ is the annual average outdoor air concentration of nuclide i ($Bq \cdot m^{-3}$ air),

Qa_j is the terrestrial animal inhalation rate (m^3 air$\cdot d^{-1}$),

Fl_j^i is the terrestrial animal inhalation transfer coefficient ($d \cdot L^{-1}$ or $d \cdot kg^{-1}$ wet biomass),

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

tha_j is the terrestrial animal inhalation holdup time (d), and

U_j is the human ingestion rate of food type j ($L \cdot a^{-1}$ or kg wet biomass$\cdot a^{-1}$).

Equation (6.1) implies that the animals are exposed continuously to outdoor air even though they may be sheltered indoors during parts of the year.

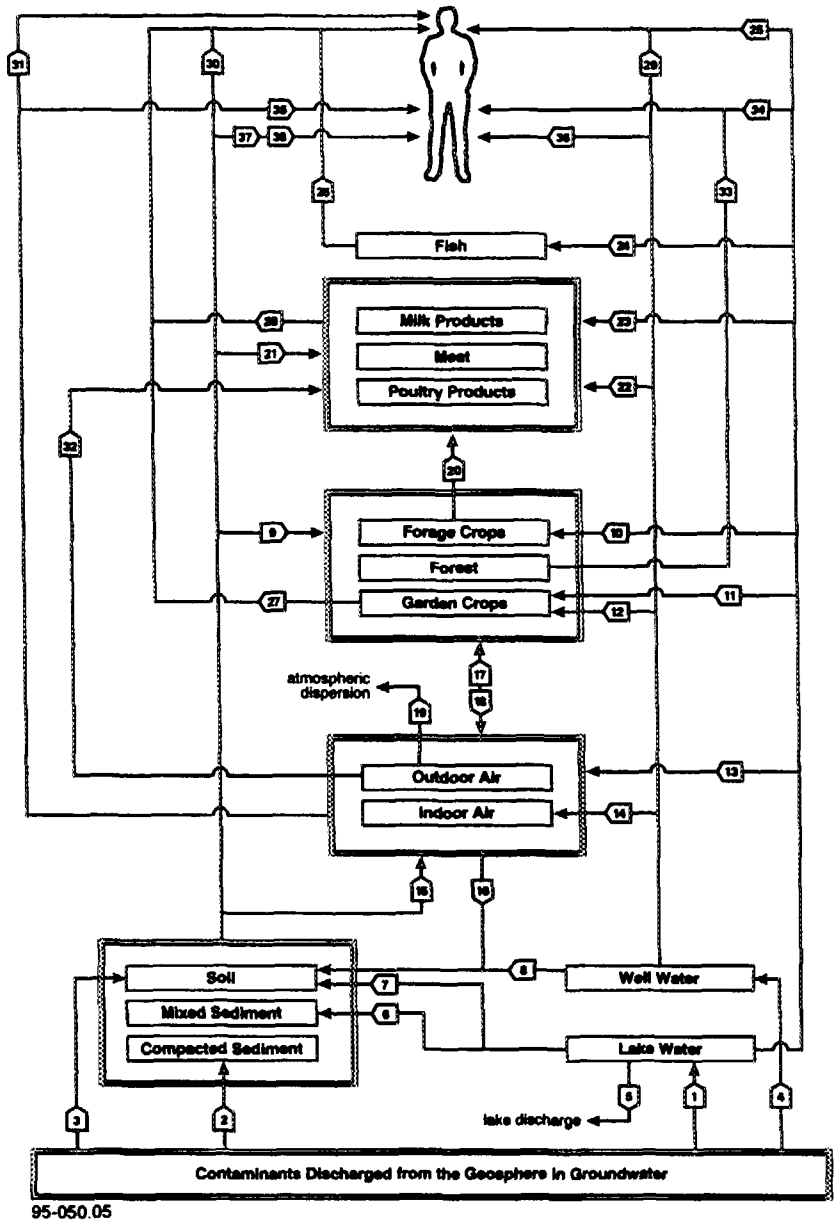
The dose to humans, $(D_j^i)_{AA}$ ($Sv \cdot a^{-1}$), from ingestion of animal products contaminated through inhalation is given by

$$\left(D_j^i\right)_{AA} = \left[\left(E_j^i\right)_{AA} + \left(E_j^{i-1}\right)_{AA}\right] \cdot DFe^i \qquad (6.2)$$

where DFe^i is the human ingestion dose conversion factor (DCF) for nuclide i ($Sv \cdot Bq^{-1}$).

The term $(E_j^{i-1})_{AA}$ accounts for radioactive ingrowth, and is included only when i is a progeny nuclide with a half-life between 1 d and 20 a (Zach and Sheppard 1992). For such ingrowth, the BIOTRAC model conservatively assumes that the progeny nuclide, i, is in secular equilibrium with the precursor nuclide, i-1.

The total dose for animals inhalation pathway is given by summing over the three terrestrial food types, TE MILK, TE MEAT and TE BIRD included in the BIOTRAC model, and over all the relevant nuclides. Note that Equations (6.1 and 6.2) do not apply for ^3H (tritium), ^{39}Ar, ^{81}Kr and ^{85}Kr because these nuclides are treated with special models in BIOTRAC (Davis et al. 1993a). This is also true for radon, another inert gas that is not readily absorbed



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 Transfer from air to milk products, meat and poultry products
33 External exposure of man to wooden building materials
34 External exposure of man to lake water
35 External exposure of man to air
36 External exposure of man to well water
37 External exposure of man to soil/sediments
38 External exposure of man to inorganic building materials

Figure 6-1. Diagram of all the Nuclide Transport and Exposure Pathways for People Treated in the BIOTRAC2 Model.

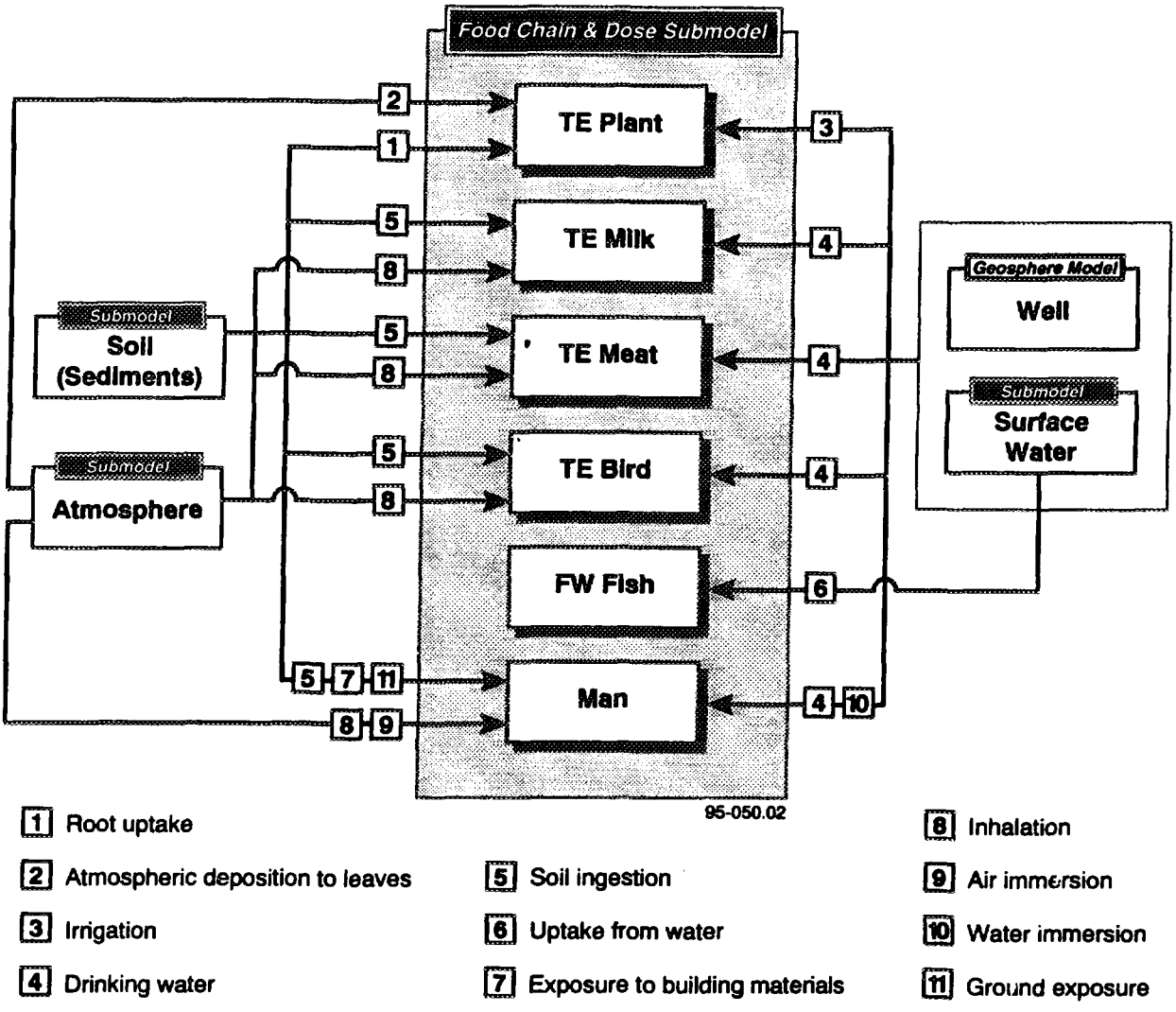


Figure 6-2. Interfaces Between the Human Food-Chain and Dose Submodel and Some of the Other BIOTRAC Submodels.

from the respiratory tract. However, as in the case of inhalation by humans, radon with its four radon progeny nuclides is potentially important for predicting doses for some of the generic target organisms included in the BIOTRAC model, as discussed in Section 7.1.1. The total dose to humans is found by summing the doses over all the pathways, nuclides and, where appropriate, food types (Davis et al. 1993a). This summation now also includes $(D_j^i)_{AA}$.

Two of the parameters of Equations (6.1 and 6.2) are described and assigned values by Zach and Sheppard (1992), as summarized by Davis et al. (1993a). These include λ_i and U_j. The remaining parameters, Fl_j^i, DFe^i, Qa_j and tha_j, are treated in detail in Sections 6.2.1.3, 6.2.2.1 and 6.2.5. Note that $(C_a^i)_o$ is an input term calculated by the atmosphere model (Figure 6-2).

6.1.2 Groundwater Limit to the Human ^{36}Cl Internal Dose

Groundwater discharging into the biosphere contains appreciable amounts of stable chlorine - ^{35}Cl and ^{37}Cl. The ratio of radioactive ^{36}Cl to stable chlorine, Cl_R^G, will be higher in the groundwater than elsewhere in the biosphere, which contains large pools of diluting stable chlorine. The presence of stable chlorine in the groundwater, therefore, can be used to establish an upper limit to the internal dose to humans. This limit corresponds to a theoretical maximum dose. This situation is entirely analogous to that for ^{14}C and ^{129}I discussed by Zach and Sheppard (1992) and Davis et al. (1993a). Chlorine-36 doses calculated with BIOTRAC's nuclide transport model (Davis et al. 1993a) could unrealistically exceed this groundwater limit for some combinations of extreme parameter values because the transport model does not allow for the large amount of stable chlorine that would accompany any ^{36}Cl released from the geosphere into the biosphere (Figure 6-3). Therefore, we have implemented a ^{36}Cl groundwater dose limit in the BIOTRAC model in the same way as was previously done for ^{14}C and ^{129}I (Davis et al. 1993a).

The ratio Cl_R^G (unitless) of ^{36}Cl to the total chlorine in groundwater is calculated from

$$Cl_R^G = \frac{C_{gw}^{Cl} \cdot gd}{C_{gw}^{Cl} \cdot gd + C_{gw}^{sCl}} \qquad (6.3)$$

where C_{gw}^{Cl} is the concentration of ^{36}Cl in groundwater ($Bq \cdot m^{-3}$ water),

gd is the mass/activity conversion factor for ^{36}Cl (kg chlorine$\cdot Bq^{-1}$), and

C_{gw}^{sCl} is the concentration of stable chlorine in groundwater (kg chlorine$\cdot m^{-3}$ water).

If the lake is the source of domestic water for the critical group, C_{gw}^{Cl} is set equal to the concentration in the groundwater discharging to the lake at the most contaminated discharge

zone (see Section 2.1.1). If the water source is the bedrock well, C_{gw}^{Cl} is set equal to the higher of the concentrations in the well water or water discharging to the lake at the most contaminated discharge zone. C_{gw}^{sCl} is treated as a distributed parameter with values based on observed chloride concentrations in near-surface groundwater at the WRA. The upper groundwater limit to the internal ^{36}Cl dose is then given by

$$\left(D^{Cl}\right)_U = DF^{Cl'} \cdot Cl_R^G \cdot B_{Cl} / (gd \cdot Bs) \qquad (6.4)$$

where $(D^{Cl})_U$ is the human maximum total internal dose from ^{36}Cl ($Sv \cdot a^{-1}$),

$DF^{Cl'}$ is the human internal dose conversion factor for ^{36}Cl based on a specific-activity model ($Sv \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ soft tissue),

B_{Cl} is the chlorine content of soft tissue in the human body (kg chlorine), and

Bs is the mass of soft tissue in the human body (kg soft tissue).

We assume that $(D^{Cl})_U$ results from the irradiation of soft tissue in the human body as in the case of the ^{14}C groundwater dose limit (Davis et al. 1993a). Note that ^{36}Cl is not a product of radioactive decay and it does not produce any radioactive progeny nuclides, which simplifies modelling.

In each BIOTRAC model simulation we compare the total internal ^{36}Cl dose predicted by the transport model with the corresponding groundwater dose limit, $(D^{Cl})_U$, and then use the smaller dose in subsequent calculations to determine the total dose to humans (Figure 6-3). This procedure screens out unrealistically high doses predicted by the transport model. The procedure is carried out independently of that for non-human biota (Section 7.1.2) so that the outcomes cannot influence each other.

Bs is documented in Davis et al. (1993a), and the remaining parameters of Equations (6.3 and 6.4), $DF^{Cl'}$, gd, C_{gw}^{sCl}, and B_{Cl}, in Sections 6.2.2.2 and 6.2.4. Note that C_{gw}^{Cl} is an input term calculated by the well model in BIOTRAC (Davis et al. 1993a) or by the geosphere model (Davison et al. 1994b).

6.1.3 Groundwater Limits to the Human ^{14}C and ^{129}I Internal Doses

These dose limits remain essentially unchanged, as documented by Zach and Sheppard (1992) and Davis et al. (1993a). However, there is a revision in the limit for ^{129}I, $(D^I)_U$, because of a change in the internal DCF, DF^I, as discussed in Section 6.2.2.3. This change was prompted by the new recommendations of the ICRP (1991a) and it does not affect the internal DCF for ^{14}C, $DF^{C'}$ (Davis et al. 1993a).

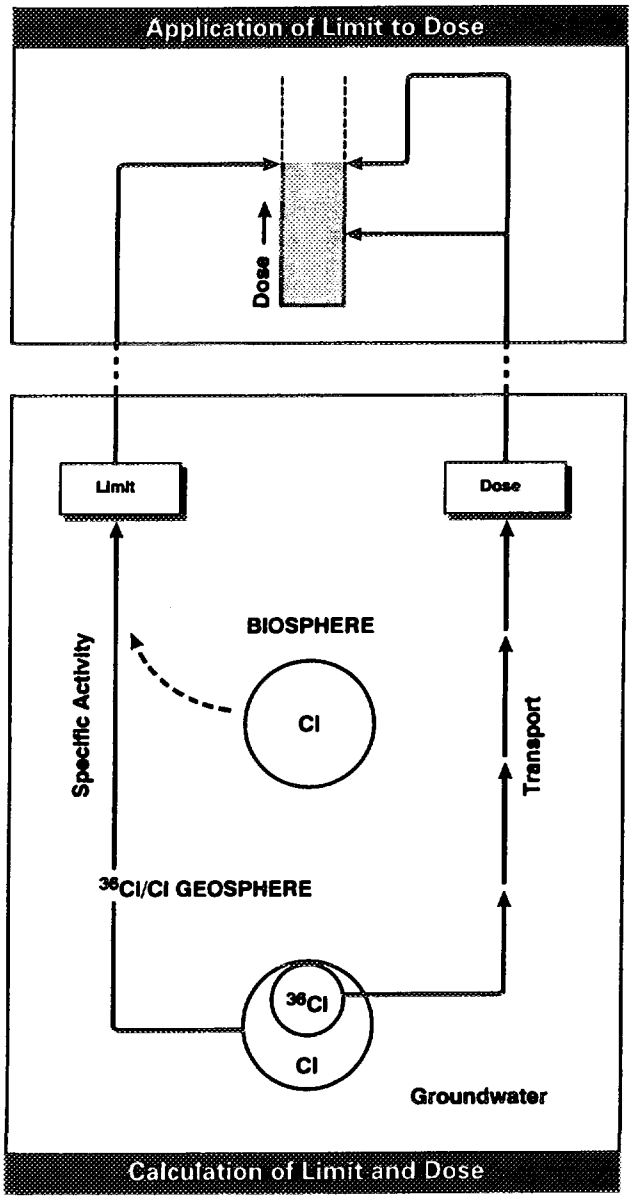


Figure 6-3. Diagram of the Calculation and Application of the Groundwater Dose Limit for ^{36}Cl. The dose calculated with the transport model is compared with the groundwater dose limit. If the dose exceeds the limit, it is set equal to the limit; if it is at or below the limit, it is not capped by the limit. The limit is calculated by assuming the ^{36}Cl/Cl ratio in the human body is the same as in the groundwater discharging to the biosphere. This is conservative because dilution of the ^{36}Cl by the large pool of Cl in the biosphere is ignored. The groundwater dose limits for ^{14}C and ^{129}I are very similar, as are the corresponding limits for humans and non-human biota.

6.2 PARAMETER VALUE CHANGES

We have reviewed the PDFs for all the transfer coefficients to ensure full inclusion of ^{36}Cl (Section 6.2.1). Inclusion of animals inhalation pathway necessitates the definition of a new set of PDFs to account for the transfer of nuclide from the respiratory tract of terrestrial animals to various food products (Section 6.2.1.3). All the internal DCFs have been re-examined in the light of new dosimetric procedures and data published by the ICRP (1991a, 1991b) (Section 6.2.2). We have also added external DCFs for ^{137}Cs, ^{239}Np and ^{243}Am (Section 6.2.3). Furthermore, we have provided parameter values and PDFs for the groundwater dose limit for ^{36}Cl (Section 6.2.4) and for animals inhalation pathway (Section 6.2.5).

6.2.1 Transfer Coefficients

6.2.1.1 Plant/Soil Concentration Ratio, Bv^i ($Bq \cdot kg^{-1}$ wet biomass per $Bq \cdot kg^{-1}$ dry soil)

This element-specific ratio defines the root uptake of nuclides from soil by terrestrial plants. The parameter is discussed in detail by Zach and Sheppard (1992) and by Davis et al. (1993a), who also established Bv^i values for all the nuclides except for chlorine. High Bv^i values are conservative because they correspond to increased nuclide transfer to plants and through the food chain.

The available data indicate that Bv^i values are lognormally distributed with a GSD of 10.0 (Zach and Sheppard 1992). We assume this is also true for ^{36}Cl. The corresponding GM is 18.0 $Bq \cdot kg^{-1}$ wet biomass per $Bq \cdot kg^{-1}$ dry soil. This value is from Baes et al. (1984), the source of most of the GM values for the BIOTRAC model. The PDF for chlorine is not subject to truncation. However, as in the case of all the other nuclides, Bv^{Cl} and Kd^{Cl} are correlated with an r value of -0.7 because plants can only absorb dissolved nuclides (Section 4.2.1).

6.2.1.2 Terrestrial Animal Ingestion Transfer Coefficients, F_j^i ($d \cdot L^{-1}$ or $d \cdot kg^{-1}$ wet biomass)

This element-specific parameter quantifies the transfer of nuclides to the terrestrial animal food types TE MILK, TE MEAT and TE BIRD from feed, water and soil ingestion. This transfer is discussed in detail by Davis et al. (1993a), who also include F_j^i values for all the nuclides except for chlorine. High F_j^i values are conservative because they lead to increased nuclide concentrations in animals.

We follow in this section the procedures and assumptions used by Zach and Sheppard (1992) to establish missing values. Thus, all the F_j^{Cl} values are lognormally distributed with the GM and the GSD values indicated in Table 6-1. The GM value for TE MILK comes from Ng et al. (1977) and the GM value for TE MEAT from Baes et al. (1984). Since the latter

compendium of transfer coefficients is incomplete and does not include poultry, we set the GM value for TE BIRD at 100 times the value for TE MEAT. Zach and Sheppard (1992) demonstrate that this is an appropriate procedure that results in reasonable values. None of the PDFs for F_j^i are subject to truncation or correlation in the BIOTRAC model and this is also true for chlorine.

6.2.1.3 Terrestrial Animal Inhalation Transfer Coefficient, Fl_j^i ($d \cdot L^{-1}$ or $d \cdot kg^{-1}$ wet biomass)

This element-specific coefficient is used to quantify the transfer of inhaled nuclides to TE MILK, TE MEAT and TE BIRD (Equation 6.1). It is also used to calculate doses to the generic target mammal and bird (Equation 7.1) for assessing environmental effects. High Fl_j^i values are conservative because they correspond to high nuclide concentrations in animals, and doses to humans and to other biota.

The uptake and metabolism of elements by animals are very complex processes (Zach and Sheppard 1992) for all modes of intake. Furthermore, inhaled nuclides can be in a variety of physical and chemical forms which adds further uncertainty. Thus, Fl_j^i values can be expected to be very variable in a similar way as the terrestrial animal transfer coefficient, F_j^i, included in the BIOTRAC model (Davis et al. 1993a). F_j^i is concerned with the transfer of ingested nuclides rather than with the transfer of inhaled nuclides. However, unlike for F_j^i, there are few data for establishing Fl_j^i values for use in environmental assessment models. The exception is a compendium of values established by the Canadian Standards Association for assessing normally operating nuclear installations (CSA 1987).

Sheppard (1994) has established a consistent procedure for calculating Fl_j^i values. It makes use of our F_j^i values, and human data on the absorption of nuclides from inhalation and from ingestion. These data are part of the ICRP lung and gastrointestinal-tract models (ICRP 1977). The terrestrial animal inhalation transfer coefficient is given by

$$Fl_j^i = F_j^i \cdot Rlg \qquad (6.5)$$

where Rlg (unitless) is the ratio of the amount of a nuclide absorbed from inhalation to the amount absorbed from ingestion per unit input.

Equation (6.5) implies that the Rlg ratio for humans also applies for mammals in general as well as for birds. This is a reasonable assumption as far as mammals are concerned because the digestive and respiratory systems of all mammals are broadly similar. However, this is not so for birds. In particular, the structure of avian lungs with their extensive air sacs (Gill 1989) is quite different than the structure of mammalian lungs, and thus the assumption may not be appropriate. Equation (6.5), which involves F_j^i, also implies that once absorbed from the lungs, nuclides behave metab olically in the same way as nuclides absorbed from the gut. For example, it assumes that the same amount of a nuclide is transferred to milk per unit nuclide absorbed from the lungs and from the gut. This assumption is reasonable to the

extent that we use separate F_j^i values for TE MILK, TE MEAT and TE BIRD, and so differences between mammals and birds are accounted for.

TABLE 6-1

PROBABILITY DENSITY FUNCTIONS FOR CHLORINE TERRESTRIAL ANIMAL TRANSFER COEFFICIENTS, F_j^i

Food Type	Distribution Type	GM*	GSD
TE MILK	Lognormal	0.017	3.2
TE MEAT	Lognormal	0.08	3.2
TE BIRD	Lognormal	8.0	3.2

* Units are in $d \cdot L^{-1}$ milk or $d \cdot kg^{-1}$ wet biomass.

We are unaware of any data indicating the statistical distribution of Fl_j^i values. However, by analogy with our treatment of F_j^i in the BIOTRAC model, we assume that Fl_j^i values are also lognormally distributed. Thus, Equation (6.5) can be used to establish the GM values of these distributions. Accordingly, we use the GM values for F_j^i listed by Davis et al. (1993a) in Equation (6.5) together with the Rlg values established by Sheppard (1994). The F_j^i values for ^{36}Cl are given in Section 6.2.1.2. The resulting GM Fl_j^i values for use in the BIOTRAC model are listed in Table 6-2.

We are also unaware of any data on the variability of Fl_j^i values, but we suspect that it reflects the variability of our F_j^i values which have a GSD of 3.2 (Zach and Sheppard 1992). In recognition of the fact that there is more uncertainty concerning Fl_j^i than F_j^i, we have adopted a larger GSD of 5.2 for Fl_j^i.

The element-specific lognormal distributions for Fl_j^i for TE MILK, TE MEAT and TE BIRD thus specified for the BIOTRAC model are not subject to any truncation or correlation with any of the other parameters.

6.2.1.4 Aquatic Concentration Ratio, B_j^i (m^3 water$\cdot kg^{-1}$ wet biomass)

This element-specific ratio defines the transfer of nuclides from water to freshwater fish, more specifically, the food type FW FISH included in the BIOTRAC model. Thus, the ratio is

given by the nuclide concentration in the edible portion of fish to the concentration in the water. Values for all the nuclides, except for chlorine, have been established by Zach and Sheppard (1992) for use in the BIOTRAC model (Davis et al. 1993a). High B_j^i values are conservative because they lead to increased nuclide concentration in fish.

In accordance with the procedures outlined by Zach and Sheppard (1992), B_j^{Cl} values are lognormally distributed with a GSD of 12.0. The GM for this distribution is 50.0 L water·kg^{-1} wet biomass, a value taken from Thompson et al. (1972), or 0.05 m^3 water·kg^{-1} wet biomass in the units required for the food-chain and dose model. As is the case of all the other nuclides, the PDF for chlorine is not subject to any truncation or correlation.

6.2.2 Internal Dose Conversion Factors for Humans

6.2.2.1 Ingestion, DFe^i, and Inhalation Dose Conversion Factors, DFi^i (Sv·Bq^{-1})

These DCFs are needed in the BIOTRAC model to predict doses to humans from ingestion and inhalation of nuclides. Previously, we have used DCFs based on ICRP 30 (1979) (Davis et al. 1993a). We are now changing to the more recent ICRP 60/61 (1991a, 1991b) values, which are summarized in Table 6-3. These values differ from the previous ones because of recently made advances in human dosimetry and radiation protection. Even so, the two sets of values are similar, as demonstrated by Zach and Sheppard (1992). High DFe^i and DFi^i values are conservative because they correspond to high human doses.

The DCF values in Table 6-3 were calculated from the annual limits on intake (ALIs) published in ICRP 61 (1991b). These ALIs are based on the principles and procedures established in ICRP 60/61 (1991a, 1991b). Thus, they involve an occupational dose limit of 20 mSv·a^{-1}. Given a choice for a given nuclide in ICRP 61, our DCFs are based on the f_1 value for gut absorption and on the pulmonary clearance class for inhalation that result in the highest ingestion and inhalation DCF values. Because of this conservative procedure, the values in Table 6-3 do not always agree with a similar set of values listed by Zach and Sheppard (1992). The aim of that set was to provide a valid comparison with ICRP 30 values, rather than to provide reasonable but conservative dose estimates as is the case here.

A small number of the values in Table 6-3 are not calculated from the ALIs given in ICRP 61. The values for ^{208}Bi are the same as those in Davis et al. (1993a) because ICRP 61 does not include this nuclide. For tritium and ^{129}I, Table 6-3 includes only a single value which accounts for both ingestion and inhalation because the BIOTRAC model uses specific-activity models to treat these nuclides. The DCF for tritium is documented in Zach and Sheppard (1992) and the DCF for ^{129}I in Section 6.2.2.3. As discussed in Section 6.2.2.3, the ^{129}I DCF value had to be revised in the light of ICRP 60 recommendations because of a change in the organ weighting factor, W_T, for the thyroid gland. No such change applies to our DCF value for tritium and so it remains unchanged.

TABLE 6-2

GEOMETRIC MEAN TERRESTRIAL ANIMAL INHALATION TRANSFER COEFFICIENTS, Fl_j^i, FOR THE FOOD TYPES TE MILK, TE MEAT AND TE BIRD

Element	TE MILK	TE MEAT	TE BIRD
Ac	1.0×10^{-2}	1.3×10^{-2}	1.3×10^{0}
Am	1.2×10^{-4}	1.0×10^{-3}	2.5×10^{0}
Ar	0.0	0.0	0.0
Be	2.6×10^{-5}	2.9×10^{-2}	2.9×10^{0}
Bi	5.5×10^{-3}	4.4×10^{-3}	4.4×10^{-1}
Br	2.0×10^{-2}	2.5×10^{-2}	2.5×10^{0}
C	1.5×10^{-2}	6.4×10^{-2}	6.4×10^{0}
Ca	1.3×10^{-2}	1.9×10^{-3}	5.3×10^{-1}
Cd	1.7×10^{-2}	3.9×10^{-3}	9.2×10^{0}
Cl	1.7×10^{-2}	8.0×10^{-2}	8.0×10^{0}
Cr	6.1×10^{-2}	5.1×10^{-2}	5.1×10^{0}
Cs	7.1×10^{-3}	2.6×10^{-2}	4.4×10^{0}
H*			
Hf	1.3×10^{-3}	2.5×10^{-1}	2.5×10^{1}
I	9.9×10^{-3}	7.0×10^{-3}	2.8×10^{0}
K	7.2×10^{-3}	1.8×10^{-2}	1.8×10^{0}
Kr	0.0	0.0	0.0
Mo	1.5×10^{-3}	7.5×10^{-3}	5.5×10^{-1}
Nb	3.0×10^{-1}	3.8×10^{0}	4.5×10^{-2}
Ni	1.1×10^{-2}	2.2×10^{-2}	2.2×10^{0}
Np	7.5×10^{-5}	8.3×10^{-4}	8.3×10^{-2}
P	1.8×10^{-2}	5.4×10^{-2}	5.4×10^{0}
Pa	7.5×10^{-4}	1.5×10^{-3}	1.5×10^{-1}
Pb	7.8×10^{-4}	1.2×10^{-3}	1.2×10^{-1}
Pd	1.0×10^{0}	4.0×10^{-1}	4.0×10^{1}
Po	1.9×10^{-3}	2.5×10^{-2}	2.5×10^{0}
Pu	1.5×10^{-4}	3.0×10^{-3}	1.1×10^{1}
Ra	5.6×10^{-4}	1.3×10^{-3}	1.3×10^{-1}
Rb	1.2×10^{-2}	1.1×10^{-2}	1.1×10^{0}
Re	1.4×10^{-3}	8.8×10^{-3}	8.8×10^{-1}
Rn	0.0	0.0	0.0

concluded...

TABLE 6-2 (concluded)

Element	TE MILK	TE MEAT	TE BIRD
Sb	6.1×10^{-4}	5.5×10^{-3}	5.5×10^{-1}
Se	4.4×10^{-3}	1.7×10^{-2}	1.0×10^{1}
Si	1.3×10^{-3}	2.0×10^{-3}	2.0×10^{-1}
Sm	9.6×10^{-3}	2.4×10^{0}	2.4×10^{2}
Sn	3.1×10^{-2}	2.1×10^{0}	2.1×10^{2}
Sr	3.1×10^{-3}	1.8×10^{-3}	6.6×10^{-1}
Ta	4.2×10^{-4}	9.0×10^{-2}	9.0×10^{0}
Tc	1.1×10^{-3}	9.4×10^{-3}	2.1×10^{0}
Te	6.0×10^{-4}	4.5×10^{-2}	4.5×10^{0}
Th	3.7×10^{-3}	4.4×10^{-3}	4.4×10^{-1}
U	4.1×10^{-3}	2.2×10^{-3}	1.3×10^{1}
Y	3.0×10^{-2}	4.5×10^{-1}	4.5×10^{1}
Zr	7.5×10^{-3}	5.0×10^{0}	5.0×10^{2}

* No transfer coefficients are required because a specific-activity model is used.
Note: Units are $d \cdot L^{-1}$ for TE MILK, and $d \cdot kg^{-1}$ wet biomass for TE MEAT and TE BIRD.

The DFi^{Rn} value in Table 6-3 is another special case. It was calculated by Richardson (1995), using the most up-to-date dosimetric information for radon and radon progeny nuclides. There is only an inhalation value because ingestion of radon has been shown to be relatively unimportant compared to inhalation. Richardson (1995) established a value of 7.7×10^{-9} $Sv \cdot Bq^{-1}$. He also indicated that radon air concentrations can be modified by using appropriate indoor and outdoor equilibrium factors, F, for radon progeny nuclides relative to radon gas in air. The formulation of the human inhalation pathway in the BIOTRAC model does not include F values (Davis et al. 1993a) and so we need to adjust the DCF value established by Richardson. The indoor and outdoor F values are 0.4 and 0.7 respectively (Richardson 1995). These values, together with the corresponding indoor and outdoor occupancy factors of 0.8 and 0.2 for the critical group (Davis et al. 1993a), yield a weighting factor of 0.46 and a weighted DFi^{Rn} value of 3.5×10^{-9} (Table 6-3). It can be used in Equation (8.25) of Davis et al. (1993a) for calculating human doses from radon inhalation.

TABLE 6-3

ICRP 60/61 HUMAN INGESTION AND INHALATION DOSE CONVERSION FACTOR VALUES, DFe^i and DFi^i

Nuclide	Ingestion DFe^i ($Sv \cdot Bq^{-1}$)	Inhalation DFi^i ($Sv \cdot Bq^{-1}$)
^{225}Ac	4.0×10^{-8}	2.2×10^{-6}
^{227}Ac	2.2×10^{-6}	1.0×10^{-3}
^{241}Am	6.7×10^{-7}	6.7×10^{-5}
^{243}Am	6.7×10^{-7}	6.7×10^{-5}
^{39}Ar	0.0*	0.0
^{10}Be	2.0×10^{-9}	1.0×10^{-7}
^{208}Bi**	1.4×10^{-9}	6.2×10^{-9}
^{210}Bi	2.0×10^{-9}	5.0×10^{-8}
^{210m}Bi	2.0×10^{-8}	2.0×10^{-6}
^{14}C	5.0×10^{-10}	5.0×10^{-10}
^{41}Ca	2.9×10^{-10}	2.9×10^{-10}
^{113m}Cd	2.2×10^{-8}	2.2×10^{-7}
^{36}Cl	1.0×10^{-9}	6.7×10^{-9}
^{135}Cs	2.0×10^{-9}	1.0×10^{-9}
^{137}Cs	2.0×10^{-8}	1.0×10^{-8}
^{3}H	2.9×10^{-8}	***
^{182}Hf	2.0×10^{-9}	5.0×10^{-7}
^{129}I	1.6×10^{-8}	***
^{40}K	5.0×10^{-9}	3.3×10^{-9}
^{81}Kr	0.0	0.0
^{85}Kr	0.0	0.0
^{93}Mo	2.5×10^{-10}	6.7×10^{-9}
^{93m}Nb	2.0×10^{-10}	6.7×10^{-9}
^{94}Nb	2.2×10^{-9}	1.0×10^{-7}
^{59}Ni	6.7×10^{-11}	3.3×10^{-10}
^{63}Ni	2.0×10^{-10}	1.0×10^{-9}
^{237}Np	6.7×10^{-7}	6.7×10^{-5}
^{239}Np	1.0×10^{-9}	6.7×10^{-10}
^{32}P	2.5×10^{-9}	4.0×10^{-9}
^{231}Pa	2.0×10^{-6}	2.0×10^{-4}

continued...

TABLE 6-3 (continued)

Nuclide	Ingestion DFe^i ($Sv \cdot Bq^{-1}$)	Inhalation DFi^i ($Sv \cdot Bq^{-1}$)
^{233}Pa	1.0×10^{-9}	2.9×10^{-9}
^{205}Pb	3.3×10^{-10}	6.7×10^{-10}
^{210}Pb	1.0×10^{-6}	2.0×10^{-6}
^{107}Pd	6.7×10^{-11}	3.3×10^{-9}
^{210}Po	2.2×10^{-7}	2.0×10^{-6}
^{238}Pu	5.0×10^{-7}	6.7×10^{-5}
^{239}Pu	5.0×10^{-7}	6.7×10^{-5}
^{240}Pu	5.0×10^{-7}	6.7×10^{-5}
^{241}Pu	1.0×10^{-8}	1.0×10^{-6}
^{242}Pu	5.0×10^{-7}	6.7×10^{-5}
^{223}Ra	1.0×10^{-7}	2.0×10^{-6}
^{224}Ra	6.7×10^{-8}	1.0×10^{-6}
^{225}Ra	6.7×10^{-8}	2.0×10^{-6}
^{226}Ra	2.2×10^{-7}	2.2×10^{-6}
^{228}Ra	2.9×10^{-7}	1.0×10^{-6}
^{87}Rb	1.0×10^{-9}	1.0×10^{-9}
^{187}Re	4.0×10^{-12}	2.0×10^{-11}
^{222}Rn	0.0	3.5×10^{-9}
^{125}Sb	1.0×10^{-9}	3.3×10^{-9}
^{126}Sb	3.3×10^{-9}	3.3×10^{-9}
^{79}Se	2.0×10^{-9}	2.0×10^{-9}
^{32}Si	1.0×10^{-9}	2.9×10^{-7}
^{126}Sn	6.7×10^{-9}	2.9×10^{-8}
^{90}Sr	3.3×10^{-8}	3.3×10^{-7}
^{182}Ta	2.2×10^{-9}	1.0×10^{-8}
^{99}Tc	6.7×10^{-10}	2.5×10^{-9}
^{125m}Te	1.0×10^{-9}	2.0×10^{-9}
^{227}Th	1.0×10^{-8}	4.0×10^{-6}
^{228}Th	6.7×10^{-8}	1.0×10^{-4}
^{229}Th	5.0×10^{-7}	3.3×10^{-4}
^{230}Th	6.7×10^{-8}	5.0×10^{-5}
^{231}Th	4.0×10^{-10}	2.5×10^{-10}
^{232}Th	4.0×10^{-7}	2.2×10^{-4}

concluded...

TABLE 6-3 (concluded)

Nuclide	Ingestion DFe^i ($Sv \cdot Bq^{-1}$)	Inhalation DFi^i ($Sv \cdot Bq^{-1}$)
^{234}Th	5.0×10^{-9}	1.0×10^{-8}
^{232}U	1.0×10^{-7}	2.0×10^{-4}
^{233}U	2.9×10^{-8}	4.0×10^{-5}
^{234}U	2.9×10^{-8}	3.3×10^{-5}
^{235}U	2.9×10^{-8}	3.3×10^{-5}
^{236}U	2.9×10^{-8}	3.3×10^{-5}
^{238}U	2.5×10^{-8}	3.3×10^{-5}
^{90}Y	4.0×10^{-9}	2.9×10^{-9}
^{93}Zr	2.9×10^{-10}	4.0×10^{-8}

* Values of 0.0 indicate doses are very low and need not be considered because nuclides are not absorbed and deposited in the body.

** Values are from Davis et al. (1993a).

*** Internal dose conversion factors, DF^{H3} and DF^{I}, account for both ingestion and inhalation using a specific-activity model, and are expressed in units of $Sv \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ soft tissue or thyroid gland. The value for ^{3}H is from Davis et al. (1993a).

6.2.2.2 Internal Dose Conversion Factor for ^{36}Cl, $DF^{Cl'}$ ($Sv \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ soft tissue)

This DCF is used to calculate the upper limit, $(D^{Cl})_U$, to the human total internal ^{36}Cl dose, based on the specific activity of ^{36}Cl in groundwater (Section 6.1.2). High $DF^{Cl'}$ are conservative because they lead to high dose limits. $DF^{Cl'}$ differs from the ingestion and inhalation DCFs (Table 6-4) used in the nuclide transport model of BIOTRAC (Davis et al. 1993a) because $DF^{Cl'}$ is calculated from the amount of ^{36}Cl residing in soft issue rather than from a unit intake rate of ^{36}Cl. We determined the $DF^{Cl'}$ value with Equation (A.21) of Zach and Sheppard (1992), adapted for ^{36}Cl rather than for ^{14}C. Using the parameter values for this equation listed in Table 6-4, we obtained a $DF^{Cl'}$ value of 1.4×10^{-6} $Sv \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ soft tissue. We assumed uniform distribution of ^{36}Cl in soft tissue. We also assumed all the emitted radiation (ICRP 1983) is absorbed by soft tissue because ^{36}Cl emits almost exclusively electron (β) radiation, which is not very penetrating.

TABLE 6-4

PARAMETER VALUES FOR CALCULATING THE HUMAN INTERNAL DOSE CONVERSION FACTOR FOR ^{36}Cl, DF^{Cl}

Parameter	Value
Number of disintegrations (Des, disintegration per Bq·a)	3.15×10^{7}
Energy absorbed (Enc, MeV·disintegration^{-1})	2.74×10^{-1}
Energy conversion factor (ecf, Sv·kg·MeV^{-1})	1.60×10^{-13}
Radiation quality factor (Q, unitless)	1.0
Organ/tissue weighting factor (W_T, unitless)	1.0

Note: Parameter values are for Equation (A.21) of Zach and Sheppard (1992).

6.2.2.3 Internal Dose Conversion Factor for ^{129}I, DF^I (Sv·a^{-1} per Bq·kg^{-1} thyroid)

This DCF is used to calculate the upper limit, $(D^I)_U$, to the human internal ^{129}I dose (Section 6.1.3). High DF^I values are conservative because they correspond to high dose limits. The previously calculated value for DF^I is 9.7×10^{-9} Sv·a^{-1} per Bq·kg^{-1} thyroid (Davis et al. 1993a).

This value is based on Equation (A.18) of Zach and Sheppard (1992). It includes a thyroid gland weighting factor, W_T, of 0.03, which was taken from ICRP 30 (1979). This W_T value has recently been increased to 0.05 (ICRP 1991a). Consequently, the DF^I value for the BIOTRAC model has been increased to 1.6×10^{-8} Sv·a^{-1} per Bq·kg^{-1} thyroid. This represents an increase of about 67%, which is directly reflected in predicted doses.

6.2.3 External Dose Conversion Factor for Humans

These DCFs are needed in the BIOTRAC model to predict doses to humans from immersion in contaminated air and water, and exposure to contaminated ground and building materials (Davis et al. 1993a). Unlike the internal DCFs (Section 6.2.2), we are not adopting new values based on ICRP 60/61 (1991a, 1991b) at this time because no such values are currently available. Thus, the values established by Zach and Sheppard (1992) and summarized by Davis et al. (1993a) remain in place. These values are based on ICRP 30 (1979) and were largely calculated by Holford (1989).

However, we have listed the external DCF values for the three new postclosure assessment nuclides, ^{137}Cs, ^{239}Np and ^{243}Am, in Table 6-5.

Application of ICRP 60 with its changed W_T values would result in very minor changes to our external DCF values. Changes would only relate to penetrating photon (γ) radiation, capable of reaching various target organs and tissues inside the human body. Because of limited penetrability, electron (β) radiation is mainly important for the skin. Alpha (α) radiation does not reach living tissues and so does not need to be considered in our external DCFs (Holford 1989). For these reasons and the fact that external exposure of humans is relatively unimportant compared to internal exposure in the EIS postclosure assessment case study (Goodwin et al. 1994), it is reasonable to continue with our previously established external DCF values for now. However, we are in the process of publishing external DCF values based on ICRP 60/61 (1991a, 1991b) (Macdonald 1996).

TABLE 6-5

HUMAN AIR IMMERSION, WATER IMMERSION, GROUND EXPOSURE AND BUILDING EXPOSURE DOSE CONVERSION FACTOR VALUES

Nuclide	Air Immersion DFa^i	Water Immersion DFh^i	Ground Exposure DFg^i	Building Exposure DFb^i
^{243}Am	7.1×10^{-8}	1.4×10^{-10}	3.1×10^{-8}	1.5×10^{-7}
^{137}Cs	2.5×10^{-9}	2.7×10^{-12}	2.0×10^{-10}	1.5×10^{-11}
^{239}Np	2.4×10^{-7}	4.2×10^{-10}	2.4×10^{-7}	4.9×10^{-7}

Note: Units for DFa^i and DFh^i are $Sv \cdot a^{-1}$ per $Bq \cdot m^{-3}$ air or water, for DFg^i $Sv \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet soil, and for DFb^i $Sv \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ dry material.

6.2.4 Parameters for ^{36}Cl Groundwater Dose Limit

Most of the parameters for this dose limit are documented in this section, the remainder are documented in Section 6.2.2.2 and by Davis et al. (1993a). All the parameter values are summarized in Table 6-6.

TABLE 6-6

PARAMETER VALUES FOR CALCULATING THE HUMAN GROUNDWATER DOSE LIMIT FOR ^{36}Cl, $(D^{Cl})_U$

Parameter	Value
Internal dose conversion factor (DF^{Cl}, $Sv \cdot a^{-1}$ per $Bq \cdot kg^{-1}$)	1.38×10^{-6}
Mass radioactivity conversion factor (gd, $kg \cdot Bq^{-1}$)	8.19×10^{-13}
Chlorine content of soft tissue (B_{Cl}, kg)	0.083
Mass of soft tissue (Bs, kg)	63.0*
Stable chlorine concentration in groundwater (C_{gw}^{sCl}, $kg \cdot m^{-3}$)**	

* Documented in Davis et al. (1993a).

** Documented in Section 6.2.4.3.

Note: Parameter values are for equations analogous to Equations (8.32 and 8.33) of Davis et al. (1993a).

6.2.4.1 Chlorine Content of Soft Tissue, B_{Cl} (kg chlorine)

This parameter is required to calculate the ^{36}Cl groundwater dose limit (Section 6.1.2). High B_{Cl} values are conservative because they increase this limit. The soft tissues considered comprise all these tissues of reference man (ICRP 1975), including yellow and red bone marrow. This amounts to 63.0 kg of soft tissue which contains 0.083 kg of chlorine (ICRP 1975).

6.2.4.2 Mass/Activity Conversion Factor for ^{36}Cl, gd ($kg\ chlorine \cdot Bq^{-1}$)

This parameter is required to calculate the ^{36}Cl groundwater dose limit (Section 6.1.2). The parameter value can be calculated from Equation (8.46) of Davis et al. (1993a). With a molecular weight, Mw^{Cl}, of 0.036 $kg \cdot mol^{-1}$, a radioactive decay constant, λ^{Cl}, of 7.30×10^{-14} s^{-1} (ICRP 1983), and Avogadro's number, NA, of 6.02×10^{23} $atoms \cdot mol^{-1}$, gd has a value of 8.2×10^{-13} $kg\ chlorine \cdot Bq^{-1}$.

6.2.4.3 Stable Chlorine Concentration in Groundwater, C_{gw}^{sCl} (kg chlorine·m^{-3} water)

This parameter is required to calculate the ^{36}Cl groundwater dose limit (Section 6.1.2). Low values are conservative because they increase this limit. C_{gw}^{sCl} is based on observed chloride concentration and salinity in rock-matrix pore water and fracture-zone groundwater of plutonic rock in the WRA. Salinity increases with depth and rock-matrix pore water is usually considerably more saline than fracture-zone groundwater. We have previously established $C_{gw}^{s\,i}$ values for carbon and for iodine (Zach and Sheppard 1992, Davis et al. 1993a). Average values for chlorine range from 0.0001 to 0.0005 kg·L^{-1} water or, for use in the BIOTRAC model, from 0.1 to 0.5 kg·m^{-3} water (Gascoyne 1994). Furthermore, values can be assumed to be uniformly distributed. We have adopted this PDF for BIOTRAC. C_{gw}^{sCl} is not correlated with any of the other model parameters.

6.2.5 Parameters for Animals Inhalation Pathway

Two of the parameters for this pathway are documented in this section, the others are documented in Section 6.2.1.3 and 6.2.2.1, and in Davis et al. (1993a).

6.2.5.1 Inhalation Rate of Animals, Qa_j (m^3 air·d^{-1})

This parameter is required in Equation (6.1) for animals inhalation pathway to quantify the transfer of nuclides to the food types TE MILK, TE MEAT and TE BIRD (Davis et al. 1993a). Qa_j is also used in Equation (7.1) to predict doses for the generic target mammal and bird included in the BIOTRAC model. These animals are related to TE MEAT and TE BIRD. High Qa_j values are conservative because they correspond to high nuclide intake rates by the animals, and high doses to humans and to animals.

The animals of concern are basically dairy cows for TE MILK, beef cattle for TE MEAT, and chickens for TE BIRD. However, we have previously defined these animals more broadly (Zach and Sheppard 1992, Davis et al. 1993a). Thus, TE MILK also includes dairy goats and sheep, TE MEAT includes a variety of domestic and wild mammalian meat producers, and TE BIRD includes poultry, upland birds and waterfowl. These broader definitions also allow meaningful dose prediction for the generic target mammal and bird included in the BIOTRAC model.

Qa_j is related to the feed ingestion rate, Qf_j, water ingestion rate, Qdw_j, and soil ingestion rate, Qs_j, of terrestrial animals. As indicated by Davis et al. (1993a), all these ingestion rates are represented in the BIOTRAC model by normal PDFs, truncated three standard deviations (SDs) below the arithmetic mean, and correlated with each other at r = 0.75. This means that Qa_j needs to be treated similarly to achieve a consistent set of intake rates in any given BIOTRAC simulation.

Few data are available to establish the required PDFs for Qa_j, therefore, we use a unique approach presented in detail by Macdonald and Sheppard (1995). Briefly, we use separate allometric equations obtained from the literature to relate inhalation and ingestion rates to the body mass of mammals and birds. The first set of equations relate the inhalation tidal volume and respiratory frequency to body mass. Qa_j is given by the product of the tidal volume and respiratory frequency. In the second set of equations, we relate the feed ingestion rate and body mass to determine the relationship between inhalation rate and feed ingestion rate.

For both of these sets of equations the average body masses for the animals of concern are required. Values for dairy cows, beef cattle and chickens can vary considerably in relation to breed, sex, age, feeding condition and market demand (Acker 1983). We have adopted average values of 600 kg for dairy cows, 500 kg for beef cattle and 3 kg for chickens (Acker 1983, Wittenberg 1992). The corresponding CSA (1987) values are 500, 350 and about 2 kg respectively. Note that high values are conservative because they correspond to high inhalation and nuclide intake rates. More conservatism is introduced because we restrict ourselves to adult animals and so ignore the lower inhalation rates that would occur earlier during development when body mass is lower.

Using these body mass values in the first set of equations for mammals and birds, we obtain inhalation rates of 90 m^3 air·d^{-1} for dairy cows, 80 m^3 air·d^{-1} for beef cattle, and 2 m^3 air·d^{-1} for chickens (Macdonald and Sheppard 1995). These values are in general agreement with the CSA (1987) values, taking differences in body mass into account. However, the CSA value of 280 m^3 air·d^{-1} for dairy cows seems excessively high. Our value for chickens is comparable to observed values from caged birds (Fedde 1976). To be consistent with our previously established PDFs for feed, water and soil ingestion rates (Davis et al. 1993a), we assume inhalation rates have normal distributions with the values calculated above as the arithmetic means. The distributions have SDs of 25% of the mean values, or 23, 20, 0.5 m^3 air·d^{-1} respectively. Furthermore, the distributions are truncated three SDs below the arithmetic mean to avoid unreasonably low values. Table 6-7 summarizes these specifications.

Using our average body mass values in the second set of equations clearly shows inhalation rate and feed ingestion rate are roughly linearly related. Therefore, we assume Qf_j and Qa_j are correlated at $r = 0.75$. This means the rates for feed, water and soil ingestion (Davis et al. 1993a), and now for inhalation are all correlated at the same level in the BIOTRAC model.

6.2.5.2 Animal Inhalation Holdup Time, tha_j (d)

This parameter specifies the average time between animals inhalation of contaminated air and the consumption by humans of the food types TE MILK, TE MEAT and TE BIRD derived from the animals (Equation (6.1)). Low tha_j are conservative because they allow less time for radioactive decay, which can be significant for short-lived nuclides. There are two potential components to tha_j, one for the air and the other for the food types. The component for air can be set to be zero because animals usually breath fresh air without any delays. We have

previously established values for the food types at 0 d for TE MILK, 4 d for TE MEAT and 0 d for TE BIRD (Davis et al. 1993a). These values imply that milk and poultry/eggs are consumed fresh whereas meat is consumed about four days after the animal has been slaughtered. We have adopted these food types specific values for tha_j.

TABLE 6-7

PROBABILITY DENSITY FUNCTIONS FOR ANIMALS

INHALATION RATES, Qa_j

Food Type	Distribution Type	Arithmetic Mean*	SD*
TE MILK	Normal	90	23
TE MEAT	Normal	80	20
TE BIRD	Normal	2	0.5

* Units are m^3 air·d^{-1}.

Note: PDFs truncated three SDs below the arithmetic mean.

7. NON-HUMAN FOOD-CHAIN AND DOSE SUBMODEL

7.1 MODEL CHANGES

One of the main changes in this model is the addition of the inhalation pathway for terrestrial animals, i.e., for the generic target mammal and bird included in the BIOTRAC model (Davis et al. 1993a). This change (Section 7.1.1) closely parallels the addition of animals inhalation pathway to the human food-chain and dose model (Section 6.1.1) because the food types TE MEAT and TE BIRD are similar to the generic target mammal and bird discussed here. We have also expanded the model to include all the nuclides considered for postclosure assessment and not just the most important ones in the EIS postclosure assessment case study. This change necessitated the inclusion of radioactive ingrowth for some of the nuclides for all the generic target organisms, i.e., for the plant, mammal, bird and fish, treated in the model (Section 7.1.1). Note that we are concerned with freshwater fish only (Davis et al. 1993a). We have also implemented ^{14}C, ^{36}Cl and ^{129}I groundwater dose limits for all the generic target organisms (Section 7.1.2), and introduced a specific-activity model for tritium (Section 7.1.3). Furthermore, we have made minor changes in our procedure for calculating external doses for the generic target organisms (Section 7.1.4). All the exposure pathways included in

the updated BIOTRAC model are shown in Figure 7-1, and Figure 7-2 shows all the interfaces of the non-human food-chain and dose submodel with the other submodels.

7.1.1 Internal Exposure of Animals and Radioactive Ingrowth

Animals, specifically the generic target mammal and bird included in the BIOTRAC model, can take in nuclides through ingestion of contaminated feed, water and soil, and through inhalation of contaminated air. The concentration of nuclide i in generic target animal b, CBi_b^i ($Bq \cdot kg^{-1}$ kg wet biomass), is given by

$$\begin{aligned} CBi_b^i = & \left(C_s^i \cdot Bv^i \cdot F_j^i \cdot Qf_j\right) \\ & + \left\{D^i \cdot \left(r_j / Y_j\right) \cdot F_j^i \cdot Qf_j \cdot \left[1 - \exp\left(-\lambda_E^i \cdot te_j\right)\right] / \lambda_E^i\right\} \\ & + \left(C_s^i \cdot F_j^i \cdot Qs_j\right) + \left(C_w^i \cdot F_j^i \cdot Qdw_j\right) + \left[\left(C_a^i\right)_o \cdot Fl_j^i \cdot Qa_j\right] , \end{aligned} \tag{7.1}$$

where 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 of nuclide i ($Bq \cdot kg^{-1}$ wet biomass per $Bq \cdot kg^{-1}$ dry soil),

F_j^i is the ingestion transfer coefficient for animal j = mammal and bird ($d \cdot kg^{-1}$ wet biomass),

Qf_j is the rate of feed ingestion by animal j (kg wet biomass$\cdot d^{-1}$),

D^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 (unitless),

Y_j is the plant yield (kg wet biomass$\cdot m^{-2}$ soil),

λ_E^i is the effective removal constant of nuclide i from vegetation (d^{-1}),

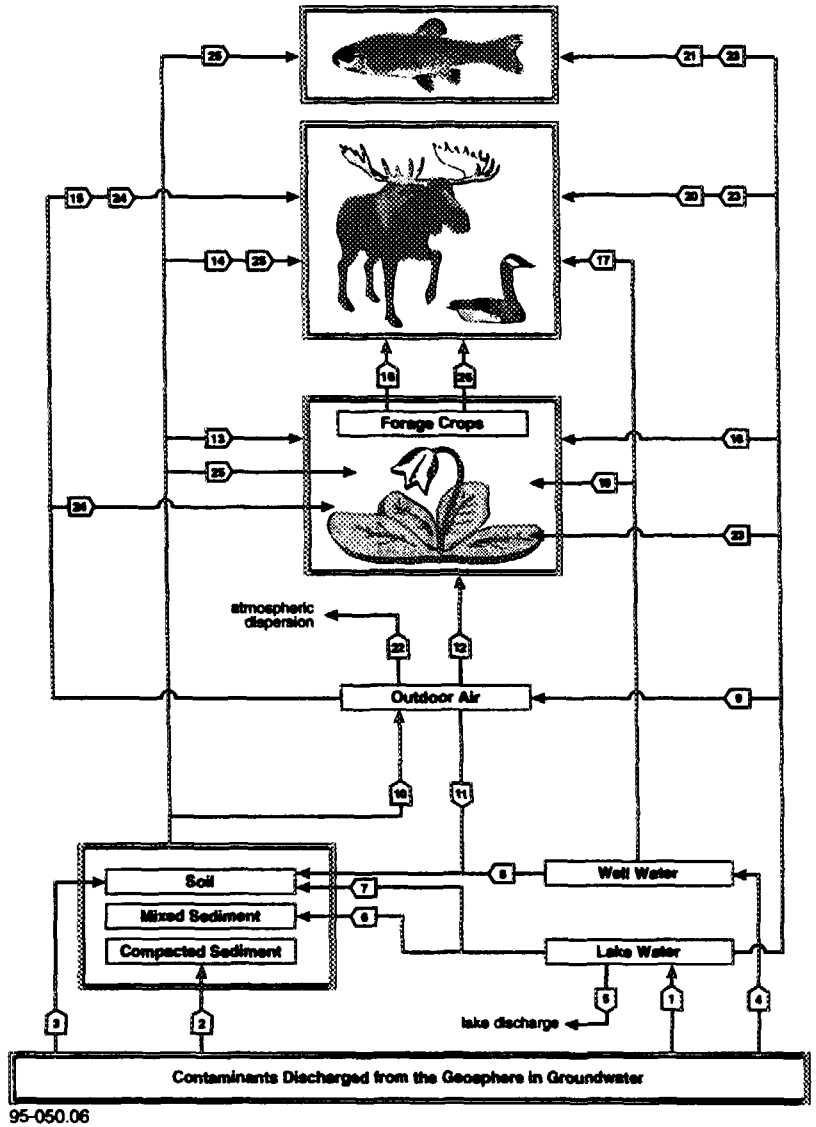
te_j is the time of above-ground exposure of vegetation (d),

Qs_j is the rate of soil ingestion by animal j (kg dry soil$\cdot d^{-1}$),

C_w^i is the annual average surface water concentration of nuclide i ($Bq \cdot m^{-3}$ water), and

Qdw_j is the rate of drinking water ingestion by animal j (m^3 water$\cdot d^{-1}$).

Some of the parameters used in Equation (7.1) are defined under Equation (6.1).



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 lake water to air
10 Transfer from soil/sediment to air
11 Transfer from air to soil (deposition)
12 Transfer from air to plant/forage crops (deposition)
13 Transfer from soil/sediment to plant/forage crops (root uptake)
14 Transfer from soil/sediment to mammal/bird (ingestion)
15 Transfer from air to mammal/bird (inhalation)
16 Transfer from forage crops to mammal/bird (ingestion)
17 Transfer from well water to mammal/bird (ingestion)
18 Transfer from lake water to plant/forage crops (irrigation)
19 Transfer from well water to plant (irrigation)
20 Transfer from lake water to mammal/bird (ingestion)
21 Transfer from lake water to fish
22 Loss through atmospheric dispersion
23 External exposure of target organisms to lake water
24 External exposure of target organisms to air
25 External exposure of target organisms to soil/sediment
26 External exposure of mammal/bird to forage crops

Figure 7-1. Diagram of all the Nuclide Transport and Exposure Pathways for the Non-Human Biota Treated in the BIOTRAC2 Model.

Equation (7.1) accounts for nuclide intake through feed, contaminated by root uptake and deposition onto leaves, water ingestion, soil ingestion, and inhalation. The animals considered are the generic target mammal and bird, which are very similar to the human food types j = TE MEAT and TE BIRD included in the BIOTRAC model. Equation (7.1) does not apply for tritium, ^{39}Ar, ^{81}Kr and ^{85}Kr because these nuclides are treated with special models in the BIOTRAC model (Davis et al. 1993a). Tritium involves a specific-activity model (Section 7.1.3) and for all the inert gas nuclides only air immersion is important (Section 7.1.3). For radon, with its four radon progeny nuclides, only the last part of Equation (7.1) concerned with inhalation applies. Radon is an inert gas and so is not subject to food-chain transfer.

The internal dose to generic target animal b from nuclide i, DBi_b^i ($Gy \cdot a^{-1}$), is given by

$$DBi_b^i = \left(CBi_b^i + CB_b^{i-1}\right) \cdot DFI^i , \qquad (7.2)$$

where DFI^i is the internal dose conversion factor for nuclide i ($Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass).

The term CBi_b^{i-1} in Equation (7.2) accounts for radioactive ingrowth, and is included only when nuclide i is a progeny nuclide with a half-life between 1 d and 20 a. For ingrowth, the BIOTRAC model conservatively assumes that the progeny nuclide, i, is in secular equilibrium with the precursor nuclide, i-1. This entire procedure is analogous to that used in the human food-chain and dose model (Section 6.1.1, Zach and Sheppard 1992).

Equation (7.2) applies to the generic target mammal and bird because only they are subject to the inhalation pathway. However, in principle it also applies to the generic plant and fish included in the non-human food-chain and dose model of BIOTRAC (Davis et al. 1993a). Most of the parameters of Equations (7.1 and 7.2) are described and assigned values by Davis et al. (1993a), including Bv^i, F_j^i, Qf_j, r_j, Y_j, λ_E^i, te_j, Qs_j and Qdw_j. However, Bv^i and F_j^i for ^{36}Cl are defined in Sections 6.2.1.1 and 6.2.1.2. Furthermore, Fl_j^i is defined in Section 6.2.1.3 for all the nuclides, Qa_j, in Section 6.2.5.1, and DFI^i in Section 7.2.1. The input term C_s^i is calculated by the soil model, D^i and $(C_a^i)_o$ by the atmosphere model, and C_w^i by the surface water model.

7.1.2 Groundwater Limits to ^{14}C, ^{36}Cl and ^{129}I Internal Doses for Non-Human Biota

The BIOTRAC model includes groundwater limits to human internal doses for ^{14}C and ^{129}I (Davis et al. 1993a), and now also for ^{36}Cl (Section 6.1.2). The presence of stable carbon, chlorine and iodine in groundwater sets an upper dose limit for non-human biota as well, i.e., for the generic target plant, mammal, bird and fish (Figure 6-3). This limit corresponds to a theoretical maximum dose. The models applicable for non-human biota are essentially the same as those for humans and there are several common parameters. Furthermore, the models for non-human biota for ^{14}C, ^{36}Cl and ^{129}I are all essentially the same.

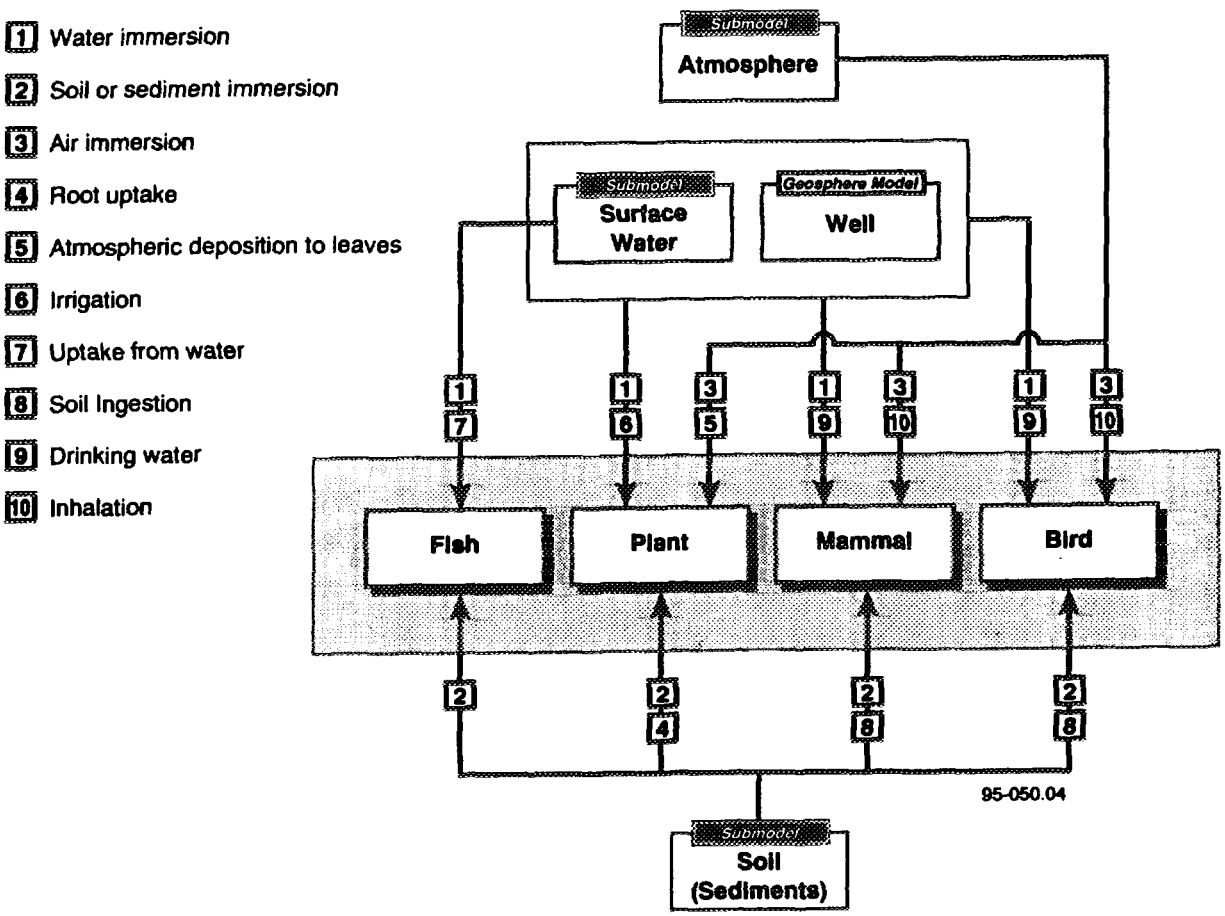


Figure 7-2. Interfaces Between the Non-Human Food-Chain and Dose Submodel and Some of the Other BIOTRAC Submodels.

The upper dose limit to the internal dose for non-human biota from ^{14}C for generic target organism b, $(DB_b^C)_U$ ($Gy \cdot a^{-1}$), is given by

$$(DB_b^C)_U = (DFI^C \cdot C_R^G \cdot Cb_b^C)/gc \quad , \tag{7.3}$$

where C_R^G is the ratio of ^{14}C to the total carbon in groundwater (unitless),

Cb_b^C is the concentration of carbon in generic target organism b (kg carbon$\cdot kg^{-1}$ wet biomass), and

gc is the mass activity conversion factor for ^{14}C (kg carbon$\cdot Bq^{-1}$).

For ^{36}Cl the dose limit for generic target organism b, $(DB_b^{Cl})_U$ ($Gy \cdot a^{-1}$), is given by

$$(DB_b^{Cl})_U = (DFI^{Cl} \cdot Cl_R^G \cdot Cb_b^{Cl})/gd \quad , \tag{7.4}$$

where Cb_b^{Cl} is the concentration of chlorine in generic target organism b (kg chlorine$\cdot kg^{-1}$ wet biomass).

Note that Cl_R^G and gd are defined in Equation (6.3).

Finally, for ^{129}I the dose limit for generic target organism b, $(DB_b^I)_U$ ($Gy \cdot a^{-1}$), is given by

$$(DB_b^I)_U = (DFI^I \cdot I_R^G \cdot Cb_b^I)/gb \quad , \tag{7.5}$$

where I_R^G is the ratio of ^{129}I to the total iodine in groundwater (unitless),

Cb_b^I is the concentration of iodine in generic target organism b (kg iodine$\cdot kg^{-1}$ wet biomass), and

gb is the mass activity conversion factor for ^{129}I (kg iodine$\cdot Bq^{-1}$).

C_R^G, gc, I_R^G and gb in Equations (7.3) and (7.5) are fully documented by Davis et al. (1993a), but Cl_R^G and gd in Equation (7.4) are documented in Equation (6.3) and Section 6.2.4.2. DFI^i and Cb_b^I used in all three equations are defined in Sections 7.2.1 and 7.2.2. Note that C_R^G, Cl_R^G, and I_R^G are quantities calculated by the BIOTRAC model and are not input parameters.

Unlike the groundwater dose limits for humans, the limits for non-human biota are always based on water discharging from the geosphere to the lake at the most contaminated discharge zone (Section 2.1.1). Thus, C_R^G, Cl_R^G and I_R^G are never based on water from the bedrock well included in the BIOTRAC model, regardless of the source of domestic water for the critical group.

In each BIOTRAC model simulation the total internal dose predicted by the transport model, DBi_b^i, is compared with the appropriate groundwater dose limit, $(DBi_b^i)_U$, and we then adopt the smaller dose as the total internal dose (Figure 6-3). This procedure is carried out for ^{14}C, ^{36}Cl and ^{129}I in turn for each of the four generic target organisms. Furthermore, these tests are carried out independently of those for humans (Section 6.1.2) so the outcome of any given test can not influence the outcome of any of the other tests.

7.1.3 Model for Tritium

The model to estimate internal doses from tritium for non-human biota is essentially the same as our specific-activity model for humans (Davis et al. 1993a). The model assumes the specific activity of tritium in these biota is the same as the specific activity in surface water. For humans, surface water can be represented by either well or lake water, but for non-human biota it is always lake water because we are mainly concerned with natural plants and animals. The internal dose to generic target organism b from tritium, DBi_b^{H3} ($Gy \cdot a^{-1}$), is given by

$$DBi_b^{H3} = (C_l^{H3}/C_w^H) \cdot Cb_b^{H3} \cdot DFI^{H3} \quad , \tag{7.6}$$

where C_l^{H3} is the annual average tritium concentration in lake water ($Bq \cdot m^{-3}$ water),

C_w^H is the concentration of hydrogen in water (g hydrogen$\cdot m^{-3}$ water),

Cb_b^H is the concentration of hydrogen in generic target organism b = plant, mammal, bird and fish (g hydrogen$\cdot kg^{-1}$ wet biomass), and

DFI^{H3} is the internal dose conversion factor for tritium ($Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass).

This model accounts for all the pathways and mechanisms whereby non-human biota can become internally contaminated with tritium. Thus, there is no need to consider any other exposure pathways. For external exposure, tritium is treated in the same way as all the other nuclides (Section 7.1.4).

Most of the parameters in Equation (7.6) are defined by Davis et al. (1993a). The exception is Cb_b^H, which is discussed and defined in Section 7.2.3.

7.1.4 External Exposure of Non-Human Biota

Equations (7.1 and 7.2) give the internal dose for a given nuclide. The total dose for a nuclide is obtained by the sum of the internal dose and the greatest of the four potential external doses: water immersion, air immersion, soil/sediment immersion or vegetation immersion (Amiro 1995a). This procedure is conservative because it assumes that an organism continuously inhabits the medium that leads to the greatest dose. This is more

realistic than our previous procedure used in the BIOTRAC model in which organisms were assumed to continuously inhabit all four media simultaneously (Davis et al. 1993a). Note that for the inert gas nuclides ^{39}Ar, ^{81}Kr, ^{85}Kr and radon the only external dose calculated is for air immersion, as is the case for humans (Davis et al. 1993a).

7.2 PARAMETER VALUE CHANGES

Previously, the BIOTRAC model included DCF values for non-human biota for only three nuclides, ^{14}C, ^{129}I and ^{99}Tc (Davis et al. 1993a). We have now expanded this database by including DCF values for all the nuclides considered in postclosure assessment, including ^{137}Cs, ^{239}Np and ^{243}Am (Section 7.2.1). Furthermore, we have established PDFs for the carbon, chlorine and iodine contents of non-human biota (Section 7.2.2). We have also done the same for hydrogen in support of the specific-activity model for tritium (Section 7.2.3).

7.2.1 Internal and External Dose Conversion Factors for Non-Human Biota

These DCF values are required to calculate doses to the four generic target organisms included in the BIOTRAC model (Section 7.1). High DCF values are conservative because they correspond to high doses. We consider internal exposure, and external exposure by water, air, soil/sediment and vegetation immersion. Generally speaking, for a given nuclide and mode of exposure the same DCF value applies for all the generic target organisms, and we use the following revised designations for the DCFs for nuclide i.

Internal	DFI^i ($Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass)
Water immersion	DFW^i ($Gy \cdot a^{-1}$ per $Bq \cdot m^{-3}$ water)
Air Immersion	DFA^i ($Gy \cdot a^{-1}$ per $Bq \cdot m^{-3}$ air)
Soil/sediment immersion	DFS^i ($Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ dry soil or sediment)
Vegetation immersion	DFV^i ($Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass)

The basic methodology for establishing these values is identical to the approach outlined by Davis et al. (1993a). The full description of the methodology is given by Amiro (1992b, 1995a, 1995b, 1996) and by Amiro and Zach (1993). The internal DCF values were derived assuming all the energies emitted from nuclides in the body are absorbed by the tissues of the organism. All the external DCF values are concerned with immersion and they are based on the exposure geometry established by Holford (1989). It assumes organisms are immersed in a semi-infinite volume of contaminated air or water. Our DFS^i values for soil/sediment immersion are assumed to be the same as the values for water immersion with a conversion of 6.7×10^{-4} from m^3 water to kg dry soil or sediment. This conversion factor is based on a soil bulk density of 1500 kg dry soil$\cdot m^{-3}$ soil. The vegetation immersion values, DFV^i, are assumed to be the same as the air immersion DCF values. These values remain unaffected by the conversion from m^3 air to kg wet biomass because we assume there is 1 kg wet plant biomass per m^3 air.

Unlike Davis et al. (1993a), we established identical DCF values for water immersion for all four generic target organisms, including the fish, based on Holford's (1989) exposure geometry. The fish immersion data used by Davis et al. (1993a) came from the National Research Council of Canada (NRCC 1983), but this database includes too few nuclides and it does not treat radioactive decay chains consistently with the BIOTRAC model (Amiro 1995a). The DCF values established in this section are consistent for all the generic target organisms. Furthermore, the derivation of all the external DCF values can now be readily traced to Holford (1989). Thus the ^{14}C, ^{129}I and ^{99}Tc DCF values for water immersion by fish established previously are now superseded by the new values given in Table 7-1.

We treat radioactive decay chains for establishing DCF values for the generic target organisms identically to the way they were treated for our human DCFs (Zach and Sheppard 1992, Davis et al. 1993a). This relates to the fact that BIOTRAC only models explicitly nuclides with a radioactive half-life greater than one day. Therefore, we derived DCF values for all the nuclides with a radioactive half-life greater than one day. Nuclides with half-lives of less than one day are included in the DCF values of their precursor nuclides. Thus, progeny nuclide DCF values were added to those of their precursor nuclides. None of the DCFs include a radiation quality factor, Q, normally applied for calculating human DCFs because these factors are only valid for humans.

The DCF values for the four generic target organisms are given in Table 7-1 for both internal and external exposures. Note that the internal ^{129}I DCF for the generic target mammal, bird and fish must be increased by a factor of ten over that for the plant to account for accumulation of iodine in the thyroid gland. This accumulation is not taken into account in food-chain transfer through the transfer coefficients or through the aquatic concentration ratio for these animals (Amiro 1992b). For the inert gas nuclides ^{39}Ar, ^{81}Kr, ^{85}Kr and radon, DCF values for air immersion are presented only because the BIOTRAC model does not calculate doses for the other external exposure pathways. Air immersion tends to dominate for these nuclides (ICRP 1977).

7.2.2 Carbon, Chlorine and Iodine Contents of Non-Human Biota, Cb_b^i

(kg carbon, chlorine or iodine·kg^{-1} wet biomass)

This parameter represents the average concentration of carbon, chlorine and iodine in the generic target plant, mammal, bird and fish used to assess radiological doses for non-human biota in the BIOTRAC model. The parameter is used in Equations (7.3) to (7.5) to determine the groundwater dose limit for these four generic target organisms. High Cb_b^i values correspond to high groundwater dose limits and so high Cb_b^i values are conservative (Section 7.1.2). Cb_b^I does not involve any parameter correlation.

TABLE 7-1

INTERNAL AND EXTERNAL DOSE CONVERSION FACTORS

FOR THE FOUR GENERIC TARGET ORGANISMS

Nuclide	Internal	External (Immersion)			
	DFI[i]	Water DFW[i]	Air DFA[i]	Soil DFS[i]	Veget. DFV[i]
^{225}Ac*	1.5×10^{-4}	2.2×10^{-9}	1.7×10^{-6}	3.3×10^{-6}	1.7×10^{-6}
^{227}Ac*	4.6×10^{-7}	1.4×10^{-11}	1.1×10^{-8}	2.1×10^{-8}	1.1×10^{-8}
^{241}Am	2.9×10^{-5}	1.5×10^{-10}	7.7×10^{-8}	2.2×10^{-7}	7.7×10^{-8}
^{243}Am	2.8×10^{-5}	2.5×10^{-10}	1.3×10^{-7}	3.7×10^{-7}	1.3×10^{-7}
^{39}Ar	0.0**	0.0	3.3×10^{-7}	0.0	0.0
^{10}Be	1.3×10^{-6}	4.4×10^{-10}	4.0×10^{-7}	6.5×10^{-7}	4.0×10^{-7}
^{208}Bi	1.3×10^{-5}	1.0×10^{-8}	6.3×10^{-6}	1.5×10^{-5}	6.3×10^{-6}
^{210}Bi	2.0×10^{-6}	5.9×10^{-10}	5.4×10^{-7}	8.8×10^{-7}	5.4×10^{-7}
^{210m}Bi*	3.0×10^{-5}	2.3×10^{-9}	1.8×10^{-6}	3.5×10^{-6}	1.8×10^{-6}
^{14}C	2.5×10^{-7}	6.5×10^{-12}	6.0×10^{-9}	9.8×10^{-9}	6.0×10^{-9}
^{41}Ca	1.4×10^{-8}	1.9×10^{-12}	1.7×10^{-9}	2.8×10^{-9}	1.7×10^{-9}
^{113m}Cd	9.4×10^{-7}	2.8×10^{-10}	2.6×10^{-7}	4.2×10^{-7}	2.6×10^{-7}
^{36}Cl	1.4×10^{-6}	5.8×10^{-10}	5.3×10^{-7}	8.7×10^{-7}	5.3×10^{-7}
^{135}Cs	3.4×10^{-7}	2.7×10^{-11}	2.5×10^{-8}	4.0×10^{-8}	2.5×10^{-8}
^{137}Cs	4.1×10^{-6}	2.7×10^{-9}	1.7×10^{-6}	4.0×10^{-6}	1.7×10^{-6}
^{3}H	2.9×10^{-8}	0.0	0.0	0.0	0.0
^{182}Hf	1.6×10^{-6}	1.0×10^{-9}	6.2×10^{-7}	1.6×10^{-6}	6.2×10^{-7}
^{129}I	4.5×10^{-7}+	1.2×10^{-10}	5.7×10^{-8}	1.8×10^{-7}	5.7×10^{-8}
^{40}K	3.4×10^{-6}	1.8×10^{-9}	1.4×10^{-6}	2.6×10^{-6}	1.4×10^{-4}
^{81}Kr	0.0	0.0	3.3×10^{-8}	0.0	0.0
^{85}Kr	0.0	0.0	4.1×10^{-7}	0.0	0.0
^{93}Mo	8.2×10^{-8}	5.0×10^{-11}	2.9×10^{-8}	7.5×10^{-8}	2.9×10^{-8}
^{93m}Nb	1.5×10^{-7}	8.9×10^{-12}	5.2×10^{-9}	1.3×10^{-8}	5.2×10^{-9}
^{94}Nb	8.8×10^{-6}	6.6×10^{-9}	4.0×10^{-6}	9.8×10^{-6}	4.0×10^{-6}
^{59}Ni	3.5×10^{-8}	1.1×10^{-11}	9.4×10^{-9}	1.7×10^{-8}	9.4×10^{-9}
^{63}Ni	8.7×10^{-8}	0.0	0.0	0.0	0.0
^{237}Np	2.5×10^{-5}	1.6×10^{-10}	8.6×10^{-8}	2.3×10^{-7}	8.6×10^{-8}
^{239}Np	2.2×10^{-6}	9.5×10^{-10}	6.2×10^{-7}	1.4×10^{-6}	6.2×10^{-7}

continued...

TABLE 7-1 (continued)

Nuclide	Internal	External (Immersion)			
	DFI[i]	Water DFW[i]	Air DFA[i]	Soil DFS[i]	Veget. DFV[i]
^{32}P	3.5×10^{-6}	1.6×10^{-9}	1.4×10^{-6}	2.4×10^{-6}	1.4×10^{-6}
^{231}Pa	2.6×10^{-5}	2.1×10^{-10}	1.3×10^{-7}	3.2×10^{-7}	1.3×10^{-7}
^{233}Pa	2.0×10^{-6}	1.0×10^{-9}	6.6×10^{-7}	1.6×10^{-6}	6.6×10^{-7}
^{205}Pb	4.6×10^{-8}	1.1×10^{-11}	7.7×10^{-9}	1.6×10^{-8}	7.7×10^{-9}
^{210}Pb	2.2×10^{-7}	2.2×10^{-11}	1.3×10^{-8}	3.3×10^{-8}	1.3×10^{-8}
^{107}Pd	4.7×10^{-8}	0.0	0.0	0.0	0.0
^{210}Po	2.7×10^{-5}	3.4×10^{-14}	2.1×10^{-11}	5.2×10^{-11}	2.1×10^{-11}
^{238}Pu	2.8×10^{-5}	8.5×10^{-12}	4.9×10^{-9}	1.3×10^{-8}	4.9×10^{-9}
^{239}Pu	2.6×10^{-5}	3.7×10^{-12}	2.4×10^{-9}	5.6×10^{-9}	2.4×10^{-9}
^{240}Pu	2.7×10^{-5}	8.1×10^{-12}	4.7×10^{-9}	1.2×10^{-8}	4.7×10^{-9}
^{241}Pu	2.7×10^{-8}	1.2×10^{-14}	6.7×10^{-12}	1.7×10^{-11}	6.7×10^{-12}
^{242}Pu	2.5×10^{-5}	6.7×10^{-12}	3.9×10^{-9}	1.0×10^{-8}	3.9×10^{-9}
^{223}Ra*	1.4×10^{-4}	3.3×10^{-9}	2.6×10^{-6}	5.0×10^{-6}	2.6×10^{-6}
^{224}Ra*	1.5×10^{-4}	7.8×10^{-9}	5.3×10^{-6}	1.2×10^{-5}	5.3×10^{-6}
^{225}Ra	6.1×10^{-7}	1.4×10^{-10}	1.0×10^{-7}	2.1×10^{-7}	1.0×10^{-7}
^{226}Ra*	4.9×10^{-4}	3.2×10^{-11}	2.0×10^{-8}	4.8×10^{-8}	2.0×10^{-8}
^{228}Ra*	7.1×10^{-6}	4.7×10^{-9}	3.1×10^{-6}	7.1×10^{-6}	3.1×10^{-6}
^{87}Rb	5.6×10^{-7}	9.9×10^{-11}	9.1×10^{-8}	1.5×10^{-7}	9.1×10^{-8}
^{187}Re	3.3×10^{-9}	0.0	0.0	0.0	0.0
^{222}Rn*	1.1×10^{-4}	0.0	6.0×10^{-6}	0.0	0.0
^{125}Sb	2.7×10^{-6}	1.9×10^{-9}	1.1×10^{-6}	2.8×10^{-6}	1.1×10^{-6}
^{126}Sb	1.6×10^{-5}	1.2×10^{-8}	7.3×10^{-6}	1.8×10^{-5}	7.3×10^{-6}
^{79}Se	2.8×10^{-7}	1.0×10^{-11}	9.3×10^{-9}	1.5×10^{-8}	9.3×10^{-9}
^{32}Si	3.3×10^{-7}	2.4×10^{-11}	2.3×10^{-8}	3.7×10^{-8}	2.3×10^{-8}
^{126}Sn*	1.2×10^{-5}	8.0×10^{-9}	5.2×10^{-6}	1.2×10^{-5}	5.2×10^{-6}
^{90}Sr	9.9×10^{-7}	3.1×10^{-10}	2.8×10^{-7}	4.6×10^{-7}	2.8×10^{-7}
^{182}Ta	7.6×10^{-6}	5.4×10^{-9}	3.3×10^{-6}	8.0×10^{-6}	3.3×10^{-6}
^{99}Tc	5.1×10^{-7}	8.6×10^{-11}	8.0×10^{-8}	1.3×10^{-7}	8.0×10^{-8}
^{125m}Te	7.3×10^{-7}	1.9×10^{-10}	9.8×10^{-8}	2.8×10^{-7}	9.8×10^{-8}

concluded...

TABLE 7-1 (concluded)

Nuclide	Internal	External (Immersion)			
	DFI^i	Water DFW^i	Air DFA^i	Soil DFS^i	Veget. DFV^i
^{227}Th	3.1×10^{-5}	4.9×10^{-10}	2.9×10^{-7}	7.3×10^{-7}	2.9×10^{-7}
^{228}Th	2.8×10^{-5}	1.5×10^{-11}	8.7×10^{-9}	2.3×10^{-8}	8.7×10^{-9}
^{229}Th	2.6×10^{-5}	4.3×10^{-10}	2.5×10^{-7}	6.5×10^{-7}	2.5×10^{-7}
^{230}Th	2.4×10^{-5}	7.2×10^{-12}	4.3×10^{-9}	1.1×10^{-8}	4.3×10^{-9}
^{231}Th	9.5×10^{-7}	1.7×10^{-10}	1.1×10^{-7}	2.5×10^{-7}	1.1×10^{-7}
^{232}Th	2.1×10^{-5}	6.2×10^{-12}	3.8×10^{-9}	9.3×10^{-9}	3.8×10^{-9}
^{234}Th*	4.6×10^{-6}	2.0×10^{-9}	1.8×10^{-6}	3.1×10^{-6}	1.8×10^{-6}
^{232}U	2.7×10^{-5}	1.0×10^{-11}	6.1×10^{-9}	1.5×10^{-8}	6.1×10^{-9}
^{233}U	2.5×10^{-5}	6.3×10^{-12}	3.8×10^{-9}	9.4×10^{-9}	3.8×10^{-9}
^{234}U	2.5×10^{-5}	8.1×10^{-12}	4.8×10^{-9}	1.2×10^{-8}	4.8×10^{-9}
^{235}U	2.4×10^{-5}	6.6×10^{-10}	3.8×10^{-7}	1.0×10^{-6}	3.8×10^{-7}
^{236}U	2.3×10^{-5}	7.3×10^{-12}	4.4×10^{-9}	1.1×10^{-8}	4.4×10^{-9}
^{238}U	2.2×10^{-5}	6.3×10^{-12}	3.8×10^{-9}	9.5×10^{-9}	3.8×10^{-9}
^{90}Y	4.7×10^{-6}	2.2×10^{-9}	2.0×10^{-6}	3.3×10^{-6}	2.0×10^{-6}
^{93}Zr	9.9×10^{-8}	0.0	0.0	0.0	0.0

* Precursor nuclides include one or more short-lived progeny nuclides with a half-life of less than one day (Amiro 1995a).

** Entries of 0.0 indicate DCF values are less than 10^{-15} in the relevant units or that no doses are calculated in the BIOTRAC model.

\+ For animals, DFI^i is 4.5×10^{-6} $Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass to account for the accumulation of iodine in the thyroid gland.

Note: Units for DFI^i are $Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass, for DFW^i they are $Gy \cdot a^{-1}$ per $Bq \cdot m^{-3}$ water, for DFA^i they are $Gy \cdot a^{-1}$ per $Bq \cdot m^{-3}$ air, for DFS^i they are $Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ dry soil or sediment, and for DFV^i they are $Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass.

Most organisms are composed of cells that carry out similar functions and this leads to broad similarity in body composition. The four most widely distributed elements (oxygen, carbon, hydrogen and nitrogen) make up about 95% of the wet biomass of organisms, and about 30 other elements (including chlorine and iodine) contribute to the remaining 5% (Weisz 1967). Most of the elements are present in the form of compounds, complexes, or ions. Some of the elements form hard deposits and others are mainly in solution. Carbon is an important

structural and metabolic element, chlorine is an important solute related to water balance and electrical activity in animals, and iodine is important in hormones produced by the thyroid gland of vertebrates. Carbon and chlorine are essential macronutrients for plants and animals, and iodine is an essential micronutrient for animals but not for plants. Essential elements tend to be regulated by organisms within relatively narrow bounds, particularly in animals. Therefore, carbon, chlorine and iodine do not always reflect availability in the environment, but there can be differences between various organisms. This means Cb_b^i may need to have different values for our four generic target organisms. Variability within these target organisms needs to reflect differences between species. In the case of the plant, differences in environmental concentrations can also contribute to this variability.

Generally speaking, organisms have a carbon content of 20% wet biomass (Weisz 1967). An average value for a variety of terrestrial plants is about 11% (Bowen 1966). However, in watery shoots, this value can be considerably lower and in nuts considerably higher (NCRP 1983). The value for reference man is 23% (ICRP 1975), a value also representative for mammals in general (Bowen 1966) and likely for birds. Based on data by Vanderploeg et al. (1975) and Stephenson et al. (1994), fish have a carbon content of about 9 to 11%. Because water has little or no carbon, the carbon content of organisms is inversely related to water content. Usually, the carbon content of organisms is determined as a percentage of the dry weight. This percentage is remarkably constant among organisms at about 45% (Bowen 1966, Jorgensen et al. 1991). The dry weight as a percentage of the wet weight of organisms is about 25% for plants (Zach and Sheppard 1992), 30 to 50% for mammals, 30 to 45% for birds (Brisbin 1968, Holms 1976, Reinecke and Stone 1982), and 25 to 45% for fish (Holmes and Donaldson 1969, Vanderploeg et al. 1975). The variability in this percentage in animals mainly relates to the amount of skeleton and body fat, tissues that do not contain much water. Furthermore, young animals have usually a higher water content than adults (Robbins 1993).

Given these values, the carbon content as a percentage of wet biomass is 11% for plants, 14 to 23% for mammals, 14 to 21% for birds, and 11 to 20% for fish. Taking all this information into consideration, we assume Cb_b^C is uniformly distributed in all cases. The range of these uniform distributions in the BIOTRAC model for the generic target plant is 0.08 to 0.15, and for the generic target mammal, bird and fish the range is 0.12 to 0.25 kg carbon·kg^{-1} wet biomass.

Organisms have a chlorine content of about 0.16% wet biomass (Weisz 1967). The range for plants is about 0.05 to 0.5% (Chapman 1966). The value for reference man is 0.14% (ICRP 1975). This value is close to the 0.18% reported by Berthet (1963) and to the 0.15% reported by Guthrie (1983) for humans in general. The range of values for mammals in general is about 0.06 to 0.18% (Bowen 1966), a range likely representative of birds too. Rainbow trout have a chlorine content of about 0.19% (Eddy and Bath 1979). The highest chlorine levels occur in marine organisms, which are not included in the BIOTRAC model with its focus on an inland exposure situation on the Shield (Davis et al. 1993a). Based on this information we assume Cb_b^{Cl} is uniformly distributed in all cases. The range of values for the generic target

plant is 5.0 x 10^{-4} to 5.0 x 10^{-3}, and for the generic target mammal, bird and fish it is 1.2 x 10^{-3} to 2.2 x 10^{-3} kg chlorine·kg^{-1} wet biomass.

Organisms in general have an iodine content of about 0.014% wet biomass (Weisz 1967). However, data by Robens et al. (1988) for plants indicate an iodine content of 4 x 10^{-5}% and those from Handle et al. (1990) a value of 1.4 x 10^{-4}%. Data presented by Kabata-Pendias and Pendias (1984) delineate a range for plants of 1.3 x 10^{-7} to 2.6 x 10^{-4}%. The value for reference man is 2 x 10^{-5} % (ICRP 1975). For humans in general a more representative value is 1 x 10^{-4} % (Berthet 1963). According to Maynard et al. (1979), farm animals have a value of 4 x 10^{-5} %. Values for white-tailed deer range from 4 x 10^{-6} to 4 x 10^{-5} %, as calculated from data by Ballard et al. (1976). It is reasonable to assume mammalian values are also representative of birds. Rainbow trout have a value of about 1.0 x 10^{-4}% (Hunt and Eales 1979), but the range of values could be considerable judging from the high variability in the aquatic concentration ratio for fish (Zach and Sheppard 1992). In vertebrates, most of the iodine resides in the thyroid gland. However, our model calculates the ^{129}I concentration for the animals as a whole (Section 7.1.1) and so Cb_b^I must be defined in the same way. The high iodine concentration in the thyroid is accounted for in the derivation of the internal DCF for animals (Section 7.2.1). As for the other two elements, we assume the distributions of Cb_b^I for the generic target organisms are all uniform with a range of 1.0 x 10^{-9} to 3.0 x 10^{-6} for the plant, 1.0 x 10^{-5} to 2.0 x 10^{-4} for the mammal and bird, and 1.0 x 10^{-5} to 1.0 x 10^{-3} kg iodine·kg^{-1} wet biomass for the fish.

All the recommended PDFs for Cb_b^i for use in the BIOTRAC model are summarized in Table 7-2.

7.2.3 Hydrogen Content of Non-Human Biota, Cb_b^H
(g hydrogen·kg^{-1} wet biomass)

This parameter is the average concentration of hydrogen in the generic target plant, mammal, bird and fish used to assess radiological doses for non-human biota in the BIOTRAC model. The parameter is used in Equation (7.6) to determine the internal tritium dose for these organisms. High Cb_b^H values correspond to high tritium concentrations and high doses to the four generic target organisms. Thus, high Cb_b^H values are conservative.

Hydrogen is an important component of all organisms (Weisz 1967). It is found in the body water in which it accounts for about 111 g·kg^{-1} water. Because water is a major body constituent, the water content of organisms, which can be quite variable, is an important determinant of Cb_b^H. Hydrogen is also a major component of the dry weight of organisms, which mainly includes carbohydrates, fats, proteins and ash. The relative contributions of some of these components can vary greatly, depending on the organism and the time of the season involved. We assume hydrogen makes up 6% of all carbohydrates, 8% of proteins and 13% of fats (Robbins 1993) to calculate Cb_b^H values for the generic target organisms. We further assume that ash makes up 1 to 2% of the dry weight (Zach et al. 1989) and does not have any hydrogen.

TABLE 7-2

RANGE OF UNIFORM PROBABILITY DENSITY FUNCTIONS FOR CARBON, CHLORINE AND IODINE CONTENTS OF THE GENERIC TARGET ORGANISMS, Cb_b^i

Target Organism	Carbon	Chlorine	Iodine
Plant	8.0×10^{-2} to 1.5×10^{-1}	5.0×10^{-4} to 5.0×10^{-3}	1.0×10^{-9} to 3.0×10^{-6}
Mammal	1.2×10^{-1} to 2.5×10^{-1}	1.2×10^{-3} to 2.2×10^{-3}	1.0×10^{-5} to 2.0×10^{-4}
Bird	1.2×10^{-1} to 2.5×10^{-1}	1.2×10^{-3} to 2.2×10^{-3}	1.0×10^{-5} to 2.0×10^{-4}
Fish	1.2×10^{-1} to 2.5×10^{-1}	1.2×10^{-3} to 2.2×10^{-3}	1.0×10^{-5} to 1.0×10^{-3}

Note: Units are kg carbon, chlorine or iodine·kg^{-1} wet biomass.

Plants have a water content of about 75% by weight (Section 7.2.2), which corresponds to 83 g hydrogen·kg^{-1} wet biomass. We assume the 250 g of dry weight per kg^{-1} wet biomass includes 84% carbohydrate, 7% protein and 7% fat (Robinson and Bolen 1989). This corresponds to 14 g hydrogen. The plant as a whole, including the water fraction, has then 97 g hydrogen·kg^{-1} wet biomass, which represents an average value. For the BIOTRAC model, we assume Cb_b^H for the generic target plant has a uniform PDF, ranging from 90 to 120 g hydrogen·kg^{-1} wet biomass (Table 7-3).

Mammals, birds and fish have all similar water contents (Section 7.2.2). We assume that these values are on average 60, 65 and 65% respectively. The dry weight of these animals is mainly made up by protein and fat, and, unlike plants, carbohydrate is unimportant (Robinson and Bolen 1989). We assume the dry weight is made up by 89% protein and 10% fat for mammals, by 75 and 24% for birds, and by 85 and 14% for fish. Proceeding in the same way as for the plant, we obtain 101, 104 and 102 g hydrogen·kg^{-1} wet biomass for the generic target mammal, bird and fish respectively. Our value for mammals comes close to the 93 g hydrogen·kg^{-1} wet soft tissue reported by Bowen (1979). ICRP reference man (ICRP 1975) has a value of about 105 g hydrogen·kg^{-1} wet soft tissue (Zach and Sheppard 1992). By including bone, these values would be lower because bone is relatively low in hydrogen content. The three generic target animals have the same uniform PDF for Cb_b^H in the BIOTRAC model, which ranges from 100 to 130 g hydrogen·kg^{-1} wet biomass (Table 7-3).

TABLE 7-3

RANGE OF UNIFORM PROBABILITY DENSITY FUNCTIONS FOR HYDROGEN CONTENT OF THE GENERIC TARGET ORGANISMS, Cb_b^H

Target Organism	Hydrogen $g \cdot kg^{-1}$ wet biomass
Plant	90 to 120
Mammal	100 to 130
Bird	100 to 130
Fish	100 to 130

8. COMPARISON WITH BIOSPHERE MODEL FOR CONCEPT ASSESSMENT

8.1 BIOTRAC MODEL VERSIONS

A code of the BIOTRAC model has been implemented in SYVAC3 for the EIS postclosure assessment case study (Goodwin et al. 1994). For this case study, the BIOTRAC model has been fully documented by Davis et al. (1993a). The BIOTRAC model has now been updated and expanded for the present study, as documented here. The name of the updated model is BIOTRAC2 (BIOTRAC - Version 2) to avoid confusion with the original BIOTRAC model. The complete documentation of the BIOTRAC2 model includes Davis et al. (1993a) and this report. A code of BIOTRAC2 has been implemented in SYVAC3 for the present postclosure assessment study (Goodwin et al. 1996).

The BIOTRAC model has been substantially updated for the present study through the addition of nuclides, the addition of a pathway, and the addition and improvement of parameter values and PDFs. The human and the non-human food-chain and dose submodels have been most affected, but all the other submodels have also been subjected to changes. Figures 6-1 and 7-1 summarize all the exposure pathways included in the BIOTRAC2 model for humans and for non-human biota.

8.2 IMPACT OF CHANGES ON SENSITIVITY ANALYSIS RESULTS

We have previously carried out sensitivity analyses for most of the individual submodels of the BIOTRAC model (Zach and Sheppard 1991, Amiro 1992a, Bird et al. 1992, Sheppard 1992) and also for BIOTRAC as a whole (Reid and Corbett 1992, 1993, Davis et al. 1993a). Furthermore, the BIOTRAC model was subject to an extensive sensitivity analysis for the entire systems model, which includes the vault, the geosphere and the biosphere models (Goodwin et al. 1994). All these analyses focus on the dose to humans. With only one major exception, the results of all these analyses also apply to the BIOTRAC2 model. This exception concerns ^{36}Cl, which was not included in most of the sensitivity analyses, and which is now known to be an important nuclide in postclosure assessment (AECL 1994a, Johnson et al. 1995).

The results of our sensitivity analyses can be summarized in terms of nuclide, exposure pathway, and parameter importance (Davis et al. 1993a, Johnson et al. 1995). The most important nuclide by far is ^{129}I, followed by ^{14}C or ^{36}Cl and, finally, by ^{99}Tc. Depending on the exact nature of the simulation or time span under consideration, ^{14}C and ^{36}Cl exchangeably assume the second or third ranks. Given these results, pathways associated with ^{129}I are most important. These are: transfer from soil to plants to humans, deposition from air or irrigation water to plants to humans, and transfer from drinking water to humans (Figure 6-1). Not surprisingly, several of the most important parameters concern ^{129}I.

1. Lake well-water switch, LW.
2. Iodine mass loading from lake to air, AIML (Section 5.2.3).
3. Iodine evasion rate from soil to air, η_s^I (Section 4.2.2).
4. Iodine plant/soil concentration ratio, Bv^I (see Section 6.2.1.1).
5. Sediment/soil switch, PS.
6. Garden irrigation switch, PI.

Together, these six parameters account for about 75% of the variation in the dose to humans predicted by the BIOTRAC model. All these parameters are defined in detail by Davis et al. (1993a). The switch parameter LW decides whether the critical group (Section 1.3) uses well water or lake water for domestic needs, PS decides whether organic soil is derived from lake sediment or from a terrestrial source, and PI decides whether or not the garden of the critical group is irrigated.

Animals inhalation pathway is the only new exposure pathway added to the BIOTRAC model (Section 6.1.1). We have previously shown that this pathway tends to be unimportant relative to human inhalation (Zach 1985b). Our sensitivity analyses results indicate human inhalation is not an important pathway and it is reasonable to conclude that animals inhalation is even less important for human dose prediction. This means the relative importance of the various exposure pathways for human dose prediction remains unchanged.

Only two of the six most important parameters identified by Davis et al. (1993a) have been modified for the BIOTRAC2 model. For η_s^I, we have reduced both the GM and the GSD values of its lognormal PDF (Section 4.2.2). The effects of this are relatively minor. This means the importance of the pathway from soil to plants to humans is decreased relatively to the transfer of deposition from air to plants to humans. It could also mean some small shifts in the importance rankings of some of the minor parameters listed by Davis et al. (1993a). For AIML, we have introduced a truncation limit for its lognormal PDF to screen out unreasonably high values (Section 5.2.3). This physical limit is very high and so would only be applied in very few simulations. When it is applied, it would decrease the concentration of ^{129}I in air, but there would be no corresponding increase in lake water concentration because we do not consider gaseous depletion of the lake (Davis et al. 1993a). The limit has little influence on dose prediction, but it corrects a physical impossibility in atmospheric suspension.

The importance of the nuclides, exposure pathways and parameters indicated above are based on the BIOTRAC model, as used for the EIS postclosure assessment case study (Davis et al. 1993a). These findings are of general significance. However, the exact importance of nuclides, exposure pathways and parameters for the present study, based on the entire system model, are given by Goodwin et al. (1996).

8.3 IMPACT OF CHANGES ON MODEL PREDICTIONS

Here we are concerned with a quantitative comparison of BIOTRAC and BIOTRAC2 model predictions. The test was selected to meet two objectives: (1) quality assurance of the SYVAC codes of the two model versions and (2) estimation of the effects of the revisions made for the BIOTRAC2 model on human dose prediction. To facilitate this, we ran the two codes stochastically 1000 times each to 10 000 years in comparable ways, together with the same vault and the geosphere models. The test included all the group 1 nuclides considered in the EIS postclosure assessment case study, i.e., ^{14}C, ^{135}Cs, ^{129}I, ^{59}Ni, ^{107}Pd, ^{79}Se, ^{99}Tc and ^{238}U (Goodwin et al. 1994).

In both cases, we used the vault and geosphere models from the EIS postclosure assessment case study (Goodwin et al. 1994), linked to either the BIOTRAC or the BIOTRAC2 models. However, to establish the same geosphere/biosphere interface for the BIOTRAC model as for the BIOTRAC2 model, we used a special code that established separate aquatic and terrestrial nodes for the GEONET geosphere model (see Section 2.1.2). The two sets of simulations were based on different sequences of random numbers and so predicted doses cannot be compared on a one-to-one basis. However, the two dose distributions can be compared directly.

Figure 8-1 contrasts the two dose distributions, which are dominated by ^{129}I. The two histograms have the same shape, but the BIOTRAC2 simulations have slightly higher peak doses. These doses are important in determining the arithmetic mean dose. Thus, the arithmetic mean dose for the BIOTRAC2 simulations is about 2.6 times greater than that for

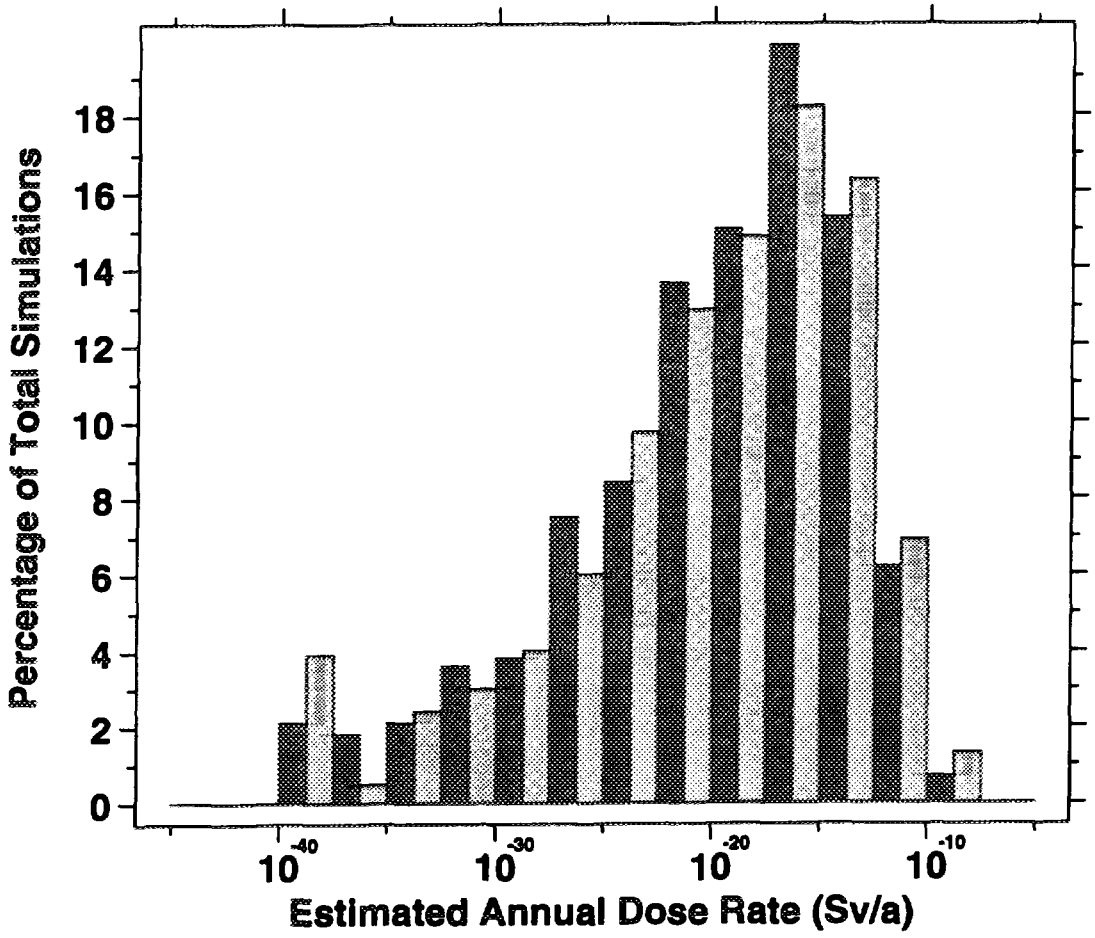


Figure 8-1. Plot of Dose Predictions ($Sv \cdot a^{-1}$) at 10 000 Years for 1000 Comparable BIOTRAC (dark) and BIOTRAC2 (light) Simulations. Note that most of the doses shown are so low they are mainly of theoretical interest.

the BIOTRAC simulations (Table 8-1). The maximum dose shows a similar increase and so does the SD. It is difficult to pinpoint the exact causes of these differences but the increase in doses is at least partially related to the increase in the internal ^{129}I DCF for humans (Section 6.2.2.3).

TABLE 8-1

SUMMARY OF DOSE PREDICTIONS ($Sv \cdot a^{-1}$) AT 10 000 YEARS FOR 1000 COMPARABLE BIOTRAC AND BIOTRAC2 SIMULATIONS

Model Version	Arithmetic Mean	SD	Maximum Value
BIOTRAC	5.1×10^{-12}	7.1×10^{-11}	2.0×10^{-9}
BIOTRAC2	1.3×10^{-11}	1.6×10^{-10}	4.0×10^{-9}

Results of the test show that the SYVAC codes of the two model versions give similar results. This is not surprising because the BIOTRAC and the BIOTRAC2 models are fundamentally the same and also similar in most details. The effects of the changes made to the BIOTRAC model do not have a strong influence on human dose predictions because the importance of nuclides, pathways and parameters remain largely unchanged. Together these results inspire confidence in the predictions by the BIOTRAC2 model because they confirm the previous expectation that the two models should make similar predictions.

9. SUMMARY AND CONCLUSIONS

We have previously developed the BIOTRAC biosphere model for assessing the safety of the disposal concept for Canada's nuclear fuel waste deep in a vault in the Canadian Shield, and used the BIOTRAC model in the postclosure assessment case study for the environmental impact statement (EIS). We have now improved and expanded the BIOTRAC model for use in the present study, designed to further demonstrate the safety and flexibility of the disposal concept, and the flexibility of our assessment methodology. The improved model is named BIOTRAC2 to avoid confusion with other model versions. Because the size and location of the hypothetical vault in the Whiteshell Research Area remain unchanged from those in the

EIS postclosure assessment case study, the biosphere/geosphere interface submodel of BIOTRAC remains largely unchanged. However, several important changes were made to the BIOTRAC model, particularly to the food-chain and dose submodels for humans and non-human biota. There are about ten main improvements.

- Addition of ^{137}Cs, ^{239}Np and ^{243}Am to the suite of postclosure assessment nuclides.
- Full inclusion of ^{36}Cl with a groundwater dose limit for humans similar to the limits for ^{14}C and ^{129}I.
- Improved gaseous evasion rates of ^{14}C and ^{129}I from soil to air.
- Improved indoor release fractions of nuclides from domestic water supply.
- Inclusion of animals inhalation pathway to predict doses for humans and non-human biota.
- Switch from ICRP 26 to ICRP 60/61 human internal dose conversion factors.
- Full incorporation of the food-chain and dose submodel for non-human biota in BIOTRAC.
- Implementation of ^{14}C, ^{36}Cl and ^{129}I groundwater dose limits for non-human biota.
- Improved procedure for calculating external doses for non-human biota.

We did not subject the BIOTRAC2 model to a sensitivity analysis because results from previous analyses of the BIOTRAC model remain largely relevant. The major exception is that ^{36}Cl is now one of the three most important nuclides, which also include ^{14}C and ^{129}I.

A quantitative comparison of the BIOTRAC and the BIOTRAC2 models shows that the two models make similar dose predictions for humans, although those of the BIOTRAC2 model are slightly higher. This is at least partially related to the increase in the internal dose conversion factor for ^{129}I.

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* Unpublished report available from SDDO, AECL, Chalk River Laboratories, ON, K0J 1J0

APPENDIX A

LIST OF ACRONYMS, NAMES AND ABBREVIATIONS

AECB	Atomic Energy Control Board
AECL	Atomic Energy of Canada Limited
ALI	Annual Limit on Intake
BIOTRAC	BIOsphere TRansport And Consequence model
BIOTRAC2	BIOsphere TRansport And Consequence model - Version 2
CEAA	Canadian Environmental Assessment Agency
COG	CANDU Owners Group
CSA	Canadian Standards Association
DCF	Dose Conversion Factor
EIS	Environmental Impact Statement
FW FISH	Freshwater fish food type
GEONET	GEOsphere NETwork model
GM	Geometric Mean
GSD	Geometric Standard Deviation
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
LD1	Low-Dipping fracture zone 1 in the Whiteshell Research Area
NRCC	National Research Council of Canada
PDF	Probability Density Function
SD	Standard Deviation
SDDO	Scientific Document Distribution Office
SYVAC3	SYstems Variability Analysis Code - Generation 3
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
TR	Technical Record
WL	Whiteshell Laboratories
WRA	Whiteshell Research Area

APPENDIX B

LIST OF SYMBOLS

AIML	aquatic iodine mass-loading parameter (m^3 water·m^{-3} air)
Bcl	chlorine content of human soft tissue in the body (kg chlorine)
B_j^{Cl}	aquatic concentration ratio for chlorine (m^3 water·kg^{-1} wet biomass)
B_j^i	aquatic concentration ratio for nuclide i (m^3 water·kg^{-1} wet biomass)
Bs	mass of human soft tissue in the body (kg soft tissue)
Bv^{Cl}	plant/soil concentration ratio for chlorine (Bq·kg^{-1} wet biomass per Bq·kg^{-1} dry soil)
Bv^I	plant/soil concentration ratio for iodine (Bq·kg^{-1} wet biomass per Bq·kg^{-1} dry soil)
Bv^i	plant/soil concentration ratio for nuclide i (Bq·kg^{-1} wet biomass per Bq·kg^{-1} dry soil)
b	generic target organism
$(C_a^i)_o$	annual average outdoor air concentration of nuclide i (Bq·m^{-3} air)
C_{gw}^{Cl}	concentration of ^{36}Cl in groundwater (Bq·m^{-3} water)
C_{gw}^{sCl}	stable concentration of chlorine in groundwater (kg chlorine·m^{-3} water)
$C_{gw}^{s\,i}$	stable concentration of nuclide i in groundwater (kg nuclide·m^{-3} water)
C_l^{H3}	annual average tritium concentration in lake water (Bq·m^{-3} water)
C_R^G	ratio of ^{14}C to the total carbon in groundwater (unitless)
C_s^i	annual average soil concentration of nuclide i (Bq·kg^{-1} dry soil)
C_w^H	concentration of hydrogen in water (g hydrogen·m^{-3} water)
C_w^i	annual average surface water concentration of nuclide i (Bq·m^{-3} water)
Cb_b^C	concentration of carbon in generic target organism b (kg carbon·kg^{-1} wet biomass)
Cb_b^{Cl}	concentration of chlorine in generic target organism b (kg chlorine·kg^{-1} wet biomass)

Cb_b^H concentration of hydrogen in generic target organism b (g hydrogen·kg^{-1} wet biomass)

Cb_b^I concentration of iodine in generic target organism b (kg iodine·kg^{-1} wet biomass)

Cb_b^i concentration of stable nuclide i in generic target organism b (kg nuclide·kg^{-1} wet biomass)

CBi_b^i concentration of radionuclide i in generic target organism b (Bq·kg^{-1} kg wet biomass)

Cl_R^G ratio of ^{36}Cl to the total chlorine in groundwater (unitless)

D^i aerial deposition rate of nuclide i to vegetation (Bq·m^{-2} soil·d^{-1})

$(D^{Cl})_U$ upper ^{36}Cl internal dose limit for humans (Sv·a^{-1})

$(D^I)_U$ upper ^{129}I internal dose limit for humans (Sv·a^{-1})

$(D_j^i)_{AA}$ dose to humans from nuclide i from ingestion of food type j contaminated by animal inhalation (Sv·a^{-1})

$(DB_b^C)_U$ upper internal dose limit for generic target organism b for ^{14}C (Gy·a^{-1})

$(DB_b^{Cl})_U$ upper internal dose limit for generic target organism b for ^{36}Cl (Gy·a^{-1})

$(DB_b^I)_U$ upper internal dose limit for generic target organism b for ^{129}I (Gy·a^{-1})

$(DBi_b^i)_U$ upper internal dose limit for generic target organism b for nuclide i (Gy·a^{-1})

DBi_b^{H3} internal dose to generic target organism b for tritium (Gy·a^{-1})

DBi_b^i internal dose to generic target organism b for nuclide i (Gy·a^{-1})

Des number of radioactive disintegrations (disintegration per Bq·a)

$DF^{Cl'}$ human internal dose conversion factor for ^{36}Cl (Sv·a^{-1} per Bq· kg^{-1} soft tissue)

DF^{H3} human internal dose conversion factor for tritium (Sv·a^{-1} per Bq· kg^{-1} soft tissue)

DF^I human internal dose conversion factor for ^{129}I (Sv·a^{-1} per Bq·kg^{-1} thyroid)

DFA^i air immersion dose conversion factor for generic target organisms for nuclide i ($Gy \cdot a^{-1}$ per $Bq \cdot m^{-3}$ air)

DFa^i human air immersion dose conversion factor for nuclide i ($Sv \cdot a^{-1}$ per $Bq \cdot m^{-3}$ air)

DFb^i human building exposure dose conversion factor for nuclide i ($Sv \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ dry material)

DFe^i human ingestion dose conversion factor for nuclide i ($Sv \cdot Bq^{-1}$)

DFg^i human ground exposure dose conversion factor for nuclide i ($Sv \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet soil)

DFh^i human water immersion dose conversion factor for nuclide i ($Sv \cdot a^{-1}$ per $Bq \cdot m^{-3}$ water)

DFI^c internal dose conversion factor for generic target organisms for ^{14}C ($Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass)

DFI^{Cl} internal dose conversion factor for generic target organisms for ^{36}Cl ($Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass)

DFI^{H3} internal dose conversion factor for generic target organisms for tritium ($Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass)

DFI^I internal dose conversion factor for generic target organisms for ^{129}I ($Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass)

DFI^i internal dose conversion factor for generic target organisms for nuclide i ($Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass)

DFi^{Rn} human inhalation dose conversion factor for radon ($Sv \cdot Bq^{-1}$)

DFi^i human inhalation dose conversion factor for nuclide i ($Sv \cdot Bq^{-1}$)

DFS^i soil immersion dose conversion factor for generic target organisms for nuclide i ($Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ dry soil)

DFV^i vegetation immersion dose conversion factor for generic target organisms for nuclide i ($Gy \cdot a^{-1}$ per $Bq \cdot kg^{-1}$ wet biomass)

DFW^i water immersion dose factor for generic target organisms for nuclide i ($Gy \cdot a^{-1}$ per $Bq \cdot m^{-3}$ water)

$(E_j^i)_{AA}$	human intake of nuclide i via food types j contaminated by animal inhalation ($Bq \cdot a^{-1}$)
Enc	amount of energy absorbed ($MeV \cdot disintegration^{-1}$)
ecf	energy conversion factor ($Sv \cdot kg \cdot MeV^{-1}$)
F	indoor or outdoor equilibrium factor for radon progeny nuclides (unitless)
F_j^{Cl}	ingestion transfer coefficient for chlorine for animal j ($d \cdot kg^{-1}$ wet biomass)
F_j^i	ingestion transfer coefficient for nuclide i and animal j ($d \cdot L^{-1}$ or $d \cdot kg^{-1}$ wet biomass)
Fl_j^i	inhalation transfer coefficient for nuclide i and animal j ($d \cdot L^{-1}$ or $d \cdot kg^{-1}$ wet biomass)
gb	mass activity conversion factor for ^{129}I (kg iodine$\cdot Bq^{-1}$)
gc	mass activity conversion factor for ^{14}C (kg carbon$\cdot Bq^{-1}$)
gd	mass/activity conversion factor for ^{36}Cl (kg chlorine$\cdot Bq^{-1}$)
I_R^G	ratio of ^{129}I to the total iodine in groundwater (unitless)
INFILT	building air infiltration rate (s^{-1})
i	nuclide
j	food type or animal
Kd^{Cl}	soil partition coefficient for chlorine (m^3 water$\cdot kg^{-1}$ dry soil or L water $\cdot kg^{-1}$ dry soil)
Kd^i	soil partition coefficient for nuclide i (m^3 water$\cdot kg^{-1}$ dry soil or L water$\cdot kg^{-1}$ dry soil)
Kd_{ds}^i	compacted sediment partition coefficient for nuclide i (m^3 water$\cdot kg^{-1}$ dry sediment)
Kd_{ds}^{Cl}	compacted sediment partition coefficient for chlorine (m^3 water$\cdot kg^{-1}$ dry sediment)
Mw^{Cl}	molecular weight of chlorine ($kg \cdot mol^{-1}$)

Q	radiation quality factor (unitless)
Qa_j	rate of inhalation by animal j (m^3 air·d^{-1})
Qdw_j	rate of drinking water ingestion by animal j (m^3 water·d^{-1})
Qf_j	rate of feed ingestion by animal j (kg wet biomass·d^{-1})
Qs_j	rate of soil ingestion by animal j (kg dry soil·d^{-1})
$RELFRAC^i$	release fraction of nuclide i from domestic water to indoor air (unitless)
Rlg	inhalation/ingestion ratio of a nuclide absorbed per unit intake (unitless)
r_j	plant interception fraction for animal j (unitless)
te_j	time of above-ground exposure of vegetation for animal j (d)
tha_j	inhalation holdup time for animal j (d)
U_j	human ingestion rate of food type j (L·a^{-1} or kg wet biomass·a^{-1})
W_T	organ or tissue weighting factor (unitless)
Y_j	plant yield for animal j (kg wet biomass·m^{-2} soil)
α^{Cl}	sediment transfer rate for chlorine (a^{-1})
α^i	sediment transfer rate for nuclide i (a^{-1})
η_s^I	gaseous evasion rate from soil for iodine (a^{-1} or s^{-1})
η_s^i	gaseous evasion rate from soil for nuclide i (a^{-1} or s^{-1})
λ^{Cl}	radioactive decay constant for ^{36}Cl (d^{-1})
λ^i	radioactive decay constant of nuclide i (d^{-1})
λ_E^i	effective removal constant of nuclide i from vegetation (d^{-1})
δ	terrestrial fraction of a discharge zone (unitless)

APPENDIX C

CORRECTION OF ERRORS IN THE BIOTRAC BIOSPHERE MODEL DOCUMENTATION

The BIOTRAC model was fully documented by Davis et al. (1993). During the subsequent review of the model, we have become aware of a number of errors in the documentation. Most of these errors are unimportant and of little consequence because they are not reflected in the BIOTRAC code itself. The exception is error 5, as indicated below. The errors are corrected in this appendix mainly for future reference.

1. Page 88, Figure 5-1

The bottom most solid arrow in this figure is unnecessary and should be deleted.

2. Page 100, Figure 5-4

On the y-axis of this figure the three bottom most designations are incorrect. They should be from the bottom up "μ-3σ", "μ-2σ", "μ-σ", and not "-μ+3σ", "-μ+2σ" and "-μ+σ".

3. Page 137, Equation (6.45)

This equation is apparently missing the atmospheric deposition term, as suggested on page 138, paragraph 3. Rather than modifying the equation, the paragraph should be changed as follows.

"Shallow soils are subject to contamination by atmospheric deposition in the same way as deep soils. The nuclide concentration resulting from deposition is calculated by using Equation (6.38) and the calculated concentration is combined with the concentration from Equation (6.47) to give the total shallow soil concentration."

4. Page 160, paragraph 4

The reference quoted in the first line of this paragraph is incorrect. It should be "(Sheppard M.I. and Hawkins 1991)" and not "(Sheppard M.I. and Hawkins 1991a)".

5. Page 198, Section 7.5.3.6

The last value quoted in this section is incorrect. It should be "9.7×10^{-5} s^{-1}" and not "0.0058 s^{-1}". This error has also been corrected in the BIOTRAC2 model version (Section 5.2.2).

6. Page 228, Equation (8.44)

This equation is incorrect because the opening and closing brackets for the third term are missing. The correct equation is:

$$Udw = En \cdot ewc - \left\{ \sum_{j=1}^{n} (U_j \cdot Ywc_j) + \sum_{j=1}^{n} U_j \cdot \left(Cym_j \cdot Cmw + Fym_j \cdot Fmw + Pym_j \cdot Pmw \right) \right\} .$$

7. Page 240, paragraph 3

The reference quoted in the second line of this paragraph is incorrect. It should be "(Linauskas 1992 a, b, c)" and not "(Linauskas 1989 a, b, c)".

8. Page 267, item 5

In line 11 of this item it should say "preclosure" and not "postclosure".

REFERENCES

Davis, P.A., R. Zach, M.E. Stephens, B.D. Amiro, G.A. Bird, J.A.K. Reid, M.I. Sheppard, S.C. Sheppard and M. Stephenson. 1993. The disposal of Canada's nuclear fuel waste: The biosphere model , BIOTRAC, for postclosure assessment. Atomic Energy of Canada Limited Report, AECL-10720, COG-93-10.

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