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D 9.20 - Addressing the uncertainties in urban/inhabited scenarios

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Abstract

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Coping with uncertainty for improved modelling and decision making in nuclear emergencies

Addressing the uncertainties in urban/inhabited scenarios

Final

Version 1.2

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Executive Summary

This document discusses the uncertainties under which stakeholders and decision-makers operate during and beyond the transition phase of a nuclear accident when developing a strategy for recovery in urban areas. In this document the term ‘urban’ is used very broadly to mean any inhabited area where people live, work, spend leisure time, attend school and do many other activities.

Chapter 1 defines the transition phase, sets out the main issues for recovery in the urban environments, defines the steps of a generic decision-making process and defines different types of uncertainty that come into play.

Chapter 2 discusses the stochastic, judgmental and modelling uncertainties when defining the radiological situation both as it stands at transition and how it will develop in the weeks, months and years following transition. Chapter 2 focusses on projections of residual dose because it is of prime significance to decision-makers, though it is recognised that other information about the radiological situation are also used, and also subject to uncertainty. Chapter 2 explores the uncertainties associated with the urban dose model ERMIN.

Chapter 3 considers how the stochastic, judgmental and modelling uncertainties of the current and projected radiological situation are integrated with wider uncertainties by the stakeholders and decision-makers in defining criteria and objectives for the recovery of the urban area. It is the intention of WP4 of the CONFIDENCE project to further elucidate this interaction by assembling panels of stakeholders to consider various recovery scenarios; so this chapter can be considered as setting the scene for this work.

Chapter 4 considers the implementation of recovery strategies. As with Chapter 2, this chapter focusses on the projections of residual dose and particularly on the significance of stochastic, judgmental and modelling uncertainties. This Chapter extends the exploration of the uncertainties associated with the urban dose model ERMIN to include a small range of management options.

Chapter 5 gives a brief introduction to environmental and social aspects implications, and also outlines a number of ‘social countermeasures’, where the primary objective may not be to reduce contamination levels or even exposure, but to assist in coping with a contamination situation during the transition phase, the recovery phase, and beyond. An example is given of how social/ethical concerns can influence the effect of dose-reductive countermeasures.

Contents

1 Introduction

In the framework of the European project CONFIDENCE, the work package WP4 (Transition to long-term recovery, involving stakeholders in decision-making processes) is devoted to improve the preparedness and response during the transition phase after a nuclear accident, identifying and trying to reduce the uncertainties in the subsequent management of the long-term exposure situation, reflecting the requirements of the new European Basic Safety Standards (BSS) [EURATOM, 2013].

For that purpose, a framework of structured collaboration involving technical experts (partners) and stakeholders in a sequential process has been established. Three tasks have been distinguished to accomplish the work [Montero & Trueba, 2017]:

1. Establishment and optimisation of remediation strategies in generic scenarios. (**Recovery scenarios planning**)
2. Involvement of stakeholders in decisions to recover acceptable living conditions (**Scenario-based stakeholder engagement**).
3. Elaboration of guidelines and recommendations to address the planning and decision making during the transition phase. (**Guidelines and recommendations**)

In agreement with the general work plan of the WP4, the first task has been carried out during the first half of the project with the following objectives:

- to identify and assess the criteria and factors (including the spatial and temporal influence in the establishment of the reference levels and the evaluation of the uncertainties in the optimisation process), that improve/affect the selection, efficiency and ending of remediation strategies, in both urban/inhabited and agricultural areas through modelling and literature review.
- to agree on scenarios and identify remediation strategies as well as the questions and issues to be addressed by national stakeholder panels through a structured brainstorming process, concluding with a dedicated workshop.

This document discusses the uncertainties under which stakeholders and decision-makers operate during and beyond the transition phase of a nuclear accident when developing a strategy for recovery in urban areas. A similar document has been elaborated to address this same objective in agricultural areas [Montero et al, 2018]. In this document the term 'urban' is used very broadly to mean any inhabited area where people live, work, spend leisure time, attend school and do many other activities.

This chapter defines the transition phase, sets out the main issues for recovery in the urban environments, defines the steps of a generic decision-making process and defines different types of uncertainty that come into play.

Chapter 2 discusses the stochastic, judgmental and modelling uncertainties when defining the radiological situation both as it stands at transition and how it will develop in the weeks, months and years following transition. Chapter 2 focusses on projections of residual dose because it is of prime significance to decision-makers, though it is recognised that other information about the radiological situation are also used, and also subject to uncertainty. Chapter 2 explores the uncertainties associated with the urban dose model ERMIN.

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1.1 Transition phase from emergency to recovery following a nuclear emergency

When an emergency involves a significant release of radioactive material to the environment (e.g. nuclear power plant accidents as in Chernobyl or Fukushima-Daiichi), we are faced, in accordance to the situation-based approach introduced by ICRP in their 2007 recommendations [ICRP, 2007], with an emergency exposure situation (EmES). The presence of residual radioactive material in the long-term results in an existing exposure situation (ExES).

Therefore, following the course of the nuclear emergency, the "transition" from EmES to ExES requires efforts to cease the emergency response and establish specific plans to begin the recovery and/ or long-term rehabilitation of the affected areas. The main objective is to facilitate the timely resumption of social and economic activities, as far as possible [IAEA, 2015].

The IAEA explains the concept of "transition phase" as:

> *"The process and the time period during which there is a progression to the point at which an emergency can be terminated"* [IAEA, 2018].

This means that there is no clear-cut boundary, neither temporal nor geographical, between both situations and the difference come from the way they are managed. This report considers the overall emergency management timeline proposed by NEA/OCDE [NEA, 2010] with three phases to identify the progression of the situation (see Figure 1) where a range of various stages and types of actions can be identified (elements in the middle of the scheme). The early and intermediate phases comprise the emergency response and the late phase is associated with long-term recovery, in concordance with the proposals of ICRP [Michiaki, 2016, Nisbet, 2017].

Preparedness	Response				Recovery
	Early		Intermediate		Late
Planning stage	Event/ response initiation	Crisis management	Consequence management	Transition to recovery (including recovery planning)	Recovery/long-term rehabilitation
	Emergency exposure situation				Existing exposure situation

Figure 1. View of the emergency management timeline and emergency phases (Source [NEA, 2010]).

If it is assumed that the transition phase commences once the situation is stable,

> *"… when the source has been brought under control, no further significant accidental releases or exposures resulting from the event are expected and the future development of the situation is well understood"* [IAEA, 2018],

Therefore, the term "transition phase" used by IAEA is equivalent to the whole "intermediate phase" as used by other organisations as ICRP and NEA/OCDE. This phase is divided between a stage of "consequence management" and a specific "transition to recovery" differentiating the various activities to address in each one.

In this context, the actuations are not driven by urgency and allow, as emergency evolves:

- For the planning and implementation of activities to enable the emergency to be declared terminated in order to prepare the long-term recovery.
- For adapting, justifying and optimizing specific protection strategies, to prepare and begin the late phase recovery and
- For the engagement of the interested parties in decisions regarding the long-term recovery.

NEA [NEA, 2010] identifies the "consequence management" as the first period in this transition/intermediate phase when the response efforts will focus on mitigating the consequences of the emergency on populations, infrastructures, environment and socio-economic structures through actions such as population protection measures, agricultural and food countermeasures, decontamination, etc. During this time, characterisation of the contamination, review or lifting of initial countermeasures and consideration of new actions are ongoing. Urban and/or agricultural countermeasures, dietary aspects, stakeholder involvement mechanisms and international coordination become increasingly important, and activities addressing the transition to recovery will begin. The last period of the intermediate phase is defined by NEA [NEA, 2010] as the "transition to recovery", when the emergency should be nearby to be terminated and the efforts will be directed to prepare plans and strategies to deal the management of following ExES and recovery of the contaminated areas.

The transition will end when all areas under an EmES have changed to ExES. The actions taken during this time should be address to approach the residual doses to the lower bound of the reference level for an EmES, is to say, to an effective dose around 20 mSv, acute or annual.

1.2 Main issues for recovery in an urban setting

An urban area is more than a physical landscape; it also comprises interlocking spheres of domestic, social, recreation, commercial, industrial, transportation, education and other human activities. Disrupting an urban area for any reason is costly (both economically and socially) and will have consequences that are hard to predict but may extend beyond the physical boundaries of that disruption. Disruption beyond a certain time will see the systems that support the urban area, both physical systems such as water or electricity supplies and non-physical such as social cohesion or services (police, health, education, retail etc.), start to degrade, and consequently ending the disruption and 'restarting' human activities within the area will become very much harder.

This presents a considerable challenge for a decision-maker faced with an urban area that has been contaminated following a nuclear accident. For severe contamination, the principal countermeasure available to the decision-maker is either full or partial restriction of access into the contaminated area. However, restriction is a very disruptive option; under full restriction the resident population has to be relocated from their homes in the area and housed elsewhere (with further disruption to the host community), businesses in the area cannot operate and facilities, infrastructure and services in the area (schools, hospitals, shops, churches etc.) are unobtainable to the wider population. Faced with such disruption, further radiological measures such as clean-up to reduce the period of restriction or non-radiological measures to reduce the impact of disruption (e.g. maintain or augment the systems that support the urban area) may be deployed, but may themselves cause further disruption or other negative consequences such as waste generation or environmental degradation. Even for less severe contamination where restriction is brief or not justified, the fact of contamination together with any management options may lead to public concerns and stigma that directly impacts commercial and economic activity that has further indirect consequences through all the spheres of human activity.

Recovery is a complex optimisation task undertaken under considerable uncertainty in order to balance the risks from the radiological situation with the various direct and indirect costs of disruption to the area and the various options to mitigate that disruption, and this must be undertaken in partnership with many different stakeholders who perceive the radiological situation and the priorities differently.

The questions that decision-makers and stakeholders need to consider include:

- How to define the affected area. What criteria in terms of dose, dose-rate and residual contamination are appropriate?
- What activities are undertaken in this area and what are the consequences both direct and indirect of disruption and options?
- What different populations use the area; residents, people who work or go to school, people travelling through the area? How are they going to be supported through the disruption?
- Are there vulnerable populations who need special support through the period of disruption; for example elderly people, disabled, sick, prison communities and migrant communities?
- Are there any services or important items of infrastructure whose operation needs to be maintained; either because it is important for activities beyond the contaminated area or because it would degrade unacceptably in the projected period of disruption and impede the eventual recovery?
- Remembering that disruption and consequences occurs beyond the physically contaminated area, the same questions above need to be considered for these indirectly affected areas. For example, how are host communities going to be supported and how are services in these areas going to be maintained and augmented with the extra demands placed on them?
- If clean-up options are being considered, what is the capacity to perform the option and handle the waste.

These are questions that can be considered both in planning and in responding. For example, there could be planning for clean-up options focusing on development of feasible, effective and acceptable strategies.

1.3 Decision making process

During the intermediate/transition phase, the actuations are not driven by urgency and allow, as emergency evolves:

- For the planning and implementation of activities to enable the emergency to be declared terminated in order to prepare the long-term recovery.

- For adapting, justifying and optimizing specific protection strategies, to prepare and begin the late phase recovery and
- For the engagement of the interested parties in decisions regarding the long-term recovery.

These plans need to be developed through a process of national dialogue with stakeholders, taking into account the inherent uncertainties on:

- the knowledge of the real consequences of an accident,
- the strategies to be implemented, and
- the potential socioeconomic impact on the affected population.

Management efforts are therefore complex because of the multiple objectives, actions, metrics, participants and so on and because the implementation takes place in a constrained world (location, money, time, resources, knowledge). The management of these complexities is the main challenge to deal with.

The success of the recovery plan will be measured by the ability of the recovery actions to be implemented in a timely manner, meeting the stakeholders' main concerns and the objectives pursued. It depends on the following:

- How is the problem addressed?
- Who (stakeholders) are involved in the recovery plan?
- What concerns are considered: health, environmental, social, economic, …?
- What are the objectives pursued in the recovery plan?
- What are the evaluation criteria?
- What are the possible options?

The challenge lies precisely in being able to take the correct decisions, considering these issues. According to [SDM, 2013], an organized and Structured Decision-Making (SDM) can help to address to identify and evaluate alternatives that focuses on engaging stakeholders, experts and decision makers in productive decision-oriented scenario-analysis as an iterative process as much as the evolution of the radiological situation requires. Figure 2 shows a scheme of the different key steps to follow:

Figure 2 The key steps of a typical Structure Decision Making (SDM) process. (Source: [SDM, 2013]).

1. **Define the Problem / Clarify the Decision Context:** Define what question or problem is being addressed and why, identify who needs to be involved and how, establish scopes

and bounds for the decision (constrains, goals or targets), and clarify the roles and responsibilities of the decision team.

2. **Define Objectives and Evaluation Criteria:** Together they define "what matters" about the decision (issues), drives the search for creative alternatives (preferred direction), and becomes the framework for comparing alternatives and making trade-offs between alternatives.
3. **Develop Alternatives:** A range of creative policy or management alternatives designed to address the objectives is developed. Alternatives should reflect substantially different approaches to the problem or different priorities across objectives, and should present decision makers with real options and choices. A "strategy" or "portfolio" is a logical combination of actions designed to be implemented as a package.
4. **Estimate Consequences:** Analytical exercise in which the performance of each alternative is estimated in terms of the evaluation criteria developed in Step 2 using available knowledge and predictive tools. Care must be taken to determine the focal areas of uncertainty and to ensure that these are represented properly in the analysis.
5. **Evaluate Trade-Offs and Select:** The next step involves evaluating the trade-offs and making value-based choices (Social, Technological, Environmental, Economic and/or Ethical values). Who is consulted and who participates in making choices may vary by the decision. Explicit choices about which alternative is preferred, could be made directly. Alternatively, structured methods for more explicitly weighting the evaluation criteria, making trade-offs, and scoring and ranking the alternatives may be used.
6. **Implement and Monitor:** The last step in the decision process then is to identify mechanisms for on-going monitoring to ensure accountability with respect to on-ground results, research to improve the information base for future decisions, and a review mechanism so that new information can be incorporated into future decisions. A key challenge will be to both reduce critical uncertainties and build in institutional flexibility to respond to new information without overextending management and political resources.

1.4 Residual dose and intervention justification and optimisation

Residual dose is defined by ICRP as:

"*The dose expected to be incurred after protective measure(s) have been fully implemented (or a decision has been taken not to implement any protective measures).*" (ICRP, 2007).

The ICRP judges that reference levels (the level of residual dose, above which it is deemed inappropriate to plan to allow exposures to occur) for the highest planned residual dose from a radiological emergency would be in the range between 20 and 100 mSv (acute or annual dose). On the use of reference levels, the view of ICRP is:

"The use of predetermined specific reference levels can facilitate timely decisions on interventions and the effective deployment of resources; however, an improper use may lead to inconsistencies with the principles of justification and optimisation" (ICRP, 2000).

It is thus clear that their application requires great caution. It should also be noted that naturally occurring radionuclides may contribute significantly to the annual dose in some areas, but not in others, which means that different degrees of intervention may be called for in similar areas with the same existing annual dose to reach a given residual dose rate. NB for the purposes of this document ingestion dose and dose from naturally occurring radionuclides are not included in residual dose estimates.

Selecting the optimal countermeasure strategy for a specific emergency situation is by no means an easy task, as countermeasure implementation can impact on society in a wide range of more or less foreseeable ways. Some aspects can relatively unproblematically be quantified in for instance monetary terms (e.g., use of machinery, consumables, transport, and worker wages), thereby facilitating intercomparison between methods. However, as stated by the ICRP in their latest recommendations (ICRP, 2007), also problems like social disruption, loss of property value and loss of income due to the contamination situation as well as due to countermeasure implementation should be taken into account. Depending on the specific scenario, there may be a wealth of such 'indirect' factors, the importance of which is in general very difficult to describe. When evaluating the possible implications of a countermeasure strategy, it is important always to measure these against the implications of doing nothing. If nothing is done to reduce the exposure problems there will be an equally long (quite possibly longer) list of different types of adverse effects on society, both radiological and non-radiological.

In addition to securing that any strategy for restoration is justified and optimised considering a wide range of aspects, it is of course essential to secure that the strategy achieves the ultimate goal of intervention: that the health consequences and other adverse impacts of the contamination are reduced sufficiently to allow the affected population and society as a whole to resume their lives and functions in the area. In extreme cases, contamination levels might be so high that even the most effective existing countermeasures would be insufficient in reducing the problems to an acceptable level that would permit a population to remain (or return) and function in the area, and any clean-up effort would thus then be in vain. A clean-up strategy that would not reduce residual doses to less than an agreed reference level should never be carried out (ICRP, 2007). However, practically any remedial means for mitigation would generally be justified to avoid permanent removal (or desertion) of (parts of) a population, as this can have immense societal repercussions.

In order to explore the uncertainty of the ERMIN residual dose calculation, a parameter response investigation was undertaken, which is described in this report and appendices.

1.5 Uncertainties

As stated in [French et al., 2018], uncertainty is interpreted differently by different people and disciplines. It can include stochastic, epistemological, endpoint, judgemental, computational and modelling uncertainties, but there are also those related to ambiguities and partially formed value judgements as well as social and ethical uncertainties.

This generic interpretation of uncertainty can be specifically adapted to the transition phase, identifying those uncertainties related to the different challenges to face in the recovery process. There are therefore, uncertainties associated:

1. To the radiological situation of the scenario, contributing to the overall uncertainty associated with the estimated impact. They are referred specifically to:
    - Space-time evolution of the contamination and the prediction of the radiological situation in the long term
    - Results of the monitoring
    - Possible changes in the future use of the scenario
2. To the goals and criteria used in the design of the protection strategy:
    - Objectives pursued

- Radiological criteria: reference levels
- Indicator Units (time to carry out the implementation of the strategy, area affected, nº of persons affected…..)

3. To the protection strategy regarding:

- Effectiveness
- Side-effects
- Generated wastes and their disposal
- Costs
- Flexibility and adaptation of the strategy in order to take into account the evolution of the radiological situation.

4. To the social pressure regarding:

- Trust and confidence: Will the protection strategy really allow the resumption of social and economic activities; stigmatization of the affected area
- Acceptability of the recovery actions
- Conflicting interests among the affected population and/or affected economic activities of the affected area

However, the involvement of stakeholders in decision-making, another important challenge to be faced in the transition phase is also subject to uncertainties, in particular, on *"how to learn from the stakeholders and the public their preferences on clean-up and recovery strategies and integrate them into decision-making, recognising that they may be unclear on their valuation of these"* [French et al., 2018]. This implies the need to help them discuss, think about and, indeed, form their values and preferences. Many of the approaches to stakeholder engagement and public participation in decision making use multi-criteria decision analysis (MCDA) to articulate such exploratory discussions [Gregory et al., 2012; Papamichail & French, 2013].

The decision making process has to operate under uncertainty concerning, for example, how long will the disruption last, what clean-up options are appropriate, and how will the urban area respond to the situation and to different management options that may be applied etc. Generally, there are two possible responses to uncertainty; either reduce the uncertainty or accommodate that uncertainty within the decision-making process; typically by making cautious or conservative assumptions.

Uncertainty can be reduced by gathering more information, ideally before the event but also during. Of course, when gathering information before the event there is the uncertainty on the event itself. Such information gathering whether in preparation or response could involve, for example, enumerating vulnerable people, identifying clean-up capacity and identifying crucial infrastructure and vulnerable services. In response to an event, uncertainty can be reduced by additional monitoring of the environment and pilot studies to demonstrate how, physically, the area responds to different management options.

Uncertainty can be reduced by widening the pool of stakeholders consulted, this is particular useful to identify unexpected consequences of proposed actions for example, by having representatives of different communities that may be impacted by those actions.

Uncertainty can be reduced by defining better models to project the current situation into the future. 'Better' could mean more accurate, for example, by using situation specific parameters rather than generic default parameters (an example of gathering more information). But 'better' could mean tools that handle and communicate the unresolvable uncertainty in a way that is useful, informative and does not overload the information consumer.

Typically, managing uncertainty involves making cautious or conservative assumptions in order to develop a robust response, i.e. a decision maker may use the 'worst case' scenario to guide actions. This works well in an emergency situation because of the restricted time scales. However, a precautionary approach during the transition phase and beyond can lead to suboptimal choices being made; extending the disruption into much larger areas and over much longer times, and therefore an optimising approach is preferable.

The iterative nature of the decision-making process (outlined Section 1.3) in finding that optimum is an implicit response to uncertainty; it follows from the recognition that situations need to be re-evaluated in the light of new information and that actions may not achieve the goals expected or may have unexpected consequences that require further actions to mitigate.

Finally, it is important to recognise that decision-making is not confined to the authorities; each person impacted by the event will make decisions about how they respond to their perception of the event at any given time, and to the management options that are being suggested or applied. Therefore, information gathering and stakeholder engagement are not one-way processes.

2 Radiological assessment

This Chapter discusses the stochastic, judgemental and modelling uncertainties in defining the radiological situation both as it stands at transition and how it will develop in the weeks, months and years following transition.

The starting point for decision-making process is the spread of contamination and the dose-rates that accompany it both of which will be imperfectly characterise as discussed in Section 2.1. However, this is not enough and for decision makers to define criteria, set objectives and develop strategies the crucial information that is needed is what the future residual doses in the subsequent days, months and years will be. To project the residual dose a model is required; one such model is ERMIN. All model outputs have uncertainty that derives both from the inputs (in this case the current radiological situation) and the model itself, and that uncertainty grows the further into the future one projects the outputs. The uncertainty associated with the projection of residual doses in ERMIN is discussed in Section 2.2.2.

Projections of residual dose are the focus of this chapter because it is on this information that many decisions will be based. However, it is recognised that the situation is wider than just the projection of residual dose. It includes the social and economic disruption and impact, which are a function of the numbers of people and size and duration of the areas affected, but this can only be evaluated when criteria are applied to the current and projected future residual doses as discussed in Chapter 3.

2.1 The spread of contamination

In the first few hours and days of a nuclear accident there will be large uncertainties about the spread of contamination across the landscape, however by the transition phase it is expected that this uncertainty will have been considerably reduced by monitoring. The magnitude of the remaining uncertainty will depend on the monitoring resources available (together with the size of the area that has been affected), the radionuclide mix, the availability of monitoring capability and the complexity of the urban landscape. There will be uncertainties about the absolute levels of deposition, the mix of radionuclides, the mode of deposition (i.e. wet, dry or mixed), the relative levels of deposition onto different urban surfaces and the physico-chemical properties of the deposited particles.

Specifically, in the transition phase before the recovery phase, there is a need to assess as well as possible the likely radiological impact (e.g., in terms of residual dose; see section 2.2) of a number of management options that are deemed to be potentially relevant, available and applicable in the local and scenario specific context. For this purpose, the early local contamination levels just after deposition on a type of reference surface in the environment (typically a short cut lawn) are relatively easily and quickly assessed, either in situ or through sampling using standard methods. If the grass is first cut as short as at all possible, and collected, and a soil sample (to 5-10 cm in depth) covering the same underlying area is subsequently taken, this can give important information as to the extent to which dry and/or wet deposition of the contaminants has occurred. Ideally, such measurements and samples should be collected on a grid that is sufficiently fine to accommodate local weather variations, but measurements on a wider grid may instead be combined with more local visual-based information on precipitation during the plume passage. This type of information may be assumed to a useful extent to be available by the time in the transition phase where considerations on recovery are first brought up. The uncertainty of these assessments depends on a range of local conditions as well as the grid size and quality/scale of any applied visual observations, and is difficult to predict. Through the associated spectral analyses with modern peak allocation software, the radionuclide mix at the given time and place will also be expected to be known (the relative amounts of the various radionuclides in the deposited mix may differ at different distances from the release point, due to, e.g., differences in physicochemical forms and local precipitation). Dosimeter readings, which will probably be the most abundant at this time, are difficult to use in detailed planning/optimising for intervention on specific surfaces, as the inhabited / urban areas normally comprise a multitude of different types of surface with different material characteristics and orientations, on which varying depositions of contaminants contribute differently to dose rate readings recorded in different positions. However, dose rate readings in reference positions such as open lawns are more easily comparable. Such measurements, which can rapidly be made in large open lawns, can give an impression of the local variation in overall deposition.

Due to the complexity of different types of contaminated surfaces in an urban area, surface specific contamination measurements would need to be carried out in collimation in different geometries, which is very time and resource consuming. It is here possible to cut corners by making use of available knowledge on the likely deposition relations on different types of surface of contaminants with different physicochemical characteristics (e.g., particles of different sizes and different types of gases), as for instance specified in the ERMIN model (Charnock et al, 2016). This builds on experience in aerosol physics and various known sorption mechanisms. As large (supermicron) particles can due to gravity and impaction be much more rapidly depleted from the contaminating plume than for instance typical ambient particles to which contaminants may have accumulated (typically in the ca. 0.5-1 μm range), the physicochemical forms (notably particle size spectra) of the contaminants can differ depending on the distance to the release point, and the relationships between concentrations of different contaminants may vary accordingly.

If the deposition of contaminants takes place in precipitation, the level of contamination will generally be much higher than if the deposition takes place in dry weather. It has for instance been calculated that in areas of Russia where it rained heavily when the contaminating plume from the Chernobyl accident passed, dry deposition only accounted for a few percent of the total deposition, since washout by precipitation is a very powerful mode of deposition (Andersson and Roed, 2006). Nevertheless, close to the Chernobyl reactor, dry deposition still led to comparatively very high levels, as the plume passing was there highly concentrated and contained very large contaminant particles that fell out very effectively. Depending on the rain intensities during the plume passage, contamination level maps may have a very 'patchy' appearance, as for instance recorded after the Chernobyl accident, where heavy rain in the Bryansk region resulted in comparatively high levels of contamination hundreds of kilometres from the point of release. Overall contamination levels may therefore vary significantly (potentially exceeding one order of magnitude in difference) over distances of less than 1 km, and there will also be some small scale inhabited area land contamination distribution variation (both vertically and horizontally, e.g., due to stem flow from trees, dripzones around dwellings, terrain unevenness, and anthropogenic alterations in the landscape), which is important to record to optimise countermeasure implementation (e.g., the content of airborne radiocaesium in 3" diameter vertical soil cylinder samples taken within a few square metres may differ by about one order of magnitude (Andersson, 1991).

The particle sizes and aerodynamic characteristics can be easily assessed using impactors, light-based particle size counters or analyses of air filters. Also other characteristics important in predicting the environmental behaviour of the contaminants, including particle solubility, could quite rapidly be determined with good accuracy (Salbu et al, 1994). In both the Fukushima and Chernobyl accident, the dominating contaminant determining long-term external dose at distances exceeding some tens of kilometres from the release point has been radiocaesium in more or less readily soluble form associated with particles in the ca. 0.7 µm range, and this is for simplicity what has also been assumed for the purpose of the scenario calculations in this report.

Cationic caesium is very effectively captured and retained in mineral traps in soil clay (particularly in the micaceous mineral illite, in which the caesium cation may be virtually irreversibly fixed). The migration thus depends on the soil type. This type of fixation also occurs in a wide range of urban construction materials (e.g., typically in clay tiles, but generally not bricks that are normally fired at a higher temperature (Andersson, 2009).

2.2 Projecting residual dose; ERMIN model uncertainty

ERMIN (European Model for Inhabited Areas, Charnock et al (2016)) has been implemented within both the RODOS (Ievdin et al, 2010) and ARGOS (PDC, 2018) decision support systems. Its purpose is to predict long-term residual doses within inhabited areas along with other endpoints useful to managing recovery in urban areas. It can handle a large number of possible management options.

In order to explore the uncertainty of the ERMIN residual dose calculation a small investigation was undertaken. Appendix 1 examines the likely distributions of the important ERMIN parameters and is summarised in Section 2.2.1. Appendix 2 uses preliminary sensitivity and uncertainty analysis to explore the impact of those on the principle endpoint of residual dose as well as looking at wider sources of uncertainty, this work is summarised in Section 2.2.2.

ERMIN is a complicated model with several components including data libraries of parameters; a brief description is given in Appendix 2.1. ERMIN produces a large number of different endpoints. However, dues to time constraints the investigation focusses on what are judged to be the most important scenarios and endpoints. The principle scenario investigated was a reactor accident involving the deposition of cationic ^{137}Cs (see Appendix 2.2) and the principal endpoint investigated was the projected average normal living effective dose as a function of time from exposure to external radiation from deposited radioactivity, to the population or subset of the population living in a contaminated zone (see Appendix 2.3). In addition, the relative contribution to this dose from different urban surfaces was investigated as this is a prime endpoint for identifying surface for clean-up option as discussed further in Chapter 4.

The normal living dose that ERMIN calculates is not to an individual but represents an average dose to a population in a built up area. Within that population there will be variations that arise from variability in the deposition, the inhabited environment (shielding, surface materials, weathering etc.) and where the individuals spend time. In addition, there is physiological variability between individuals. Some of this variability is represented within the model; for example, different urban and semi-urban environments have different shielding properties and the user is able to choose the most appropriate idealized environment from a limit selection in the ERMIN database. However, most of the variability is not represented and for most processes ERMIN uses average or representative values, for example; average deposition to a reference surface, average weathering rates, average occupancy and standard adult physiological parameters (e.g. breathing rate).

The investigation focusses on external gamma doses. While ERMIN predicts beta doses in skin and effective doses from internal exposure to inhaled resuspended radioactivity, for reactor accident scenarios these are usually less important pathways and within ERMIN are subject to considerable additional uncertainties. In the case of beta doses, the most significant is judged to arise from stochastic uncertainty particularly in the unit doses rates from beta emitting radionuclides on different surfaces in different inhabited environments. For resuspension, the most significant probably arise from model uncertainty. The special cases of deposition during or onto snow are also not considered.

2.2.1 Parameter uncertainty

Appendix 1 is an extensive review of parameter uncertainty (combined stochastic and judgemental uncertainties) of those ERMIN components identified as most significant in Appendix 2.4. The focus of this study are the parameters describing the relative initial deposition of contaminants on different urban surfaces, parameters determining the natural weathering and migration of contaminants in the environment over time, and the occupancy by people of different positions indoors and outdoors in the environment. These, together with the uncertainty of the dose-reductive effect of the clean-up process (here the main variation will be due to the degree of case-specific countermeasure appropriateness and the extent to which proper recommendations for optimisation of the countermeasures have been given and followed; it is clear that virtually all countermeasures may have no significant dose reductive effect at all if implemented wrongly) are the most important in that context.

The relative initial deposition figures on different surfaces would at the transition phase not be likely to have been measured. Since early recovery countermeasures need planning and implementation at this time, such values must be derived from past experience. Deposition (wet or dry) to different types of surface in the urban complex depends greatly on the physicochemical form(s) of the contaminants. Appendix 1.1 first gives an account of the physicochemical forms of contaminants encountered after the Chernobyl and Fukushima accidents (and beyond). For different ranges of physicochemical properties (notably aerosol size), Appendix 1 section 1.1.1 describes the state-of-the-art literature relating to deposition in absence of precipitation. On this background statistical normal distributions (in terms of mean and standard deviation) of the dry deposition relative to that on the shortcut grassed reference surface are described in Appendix 1.1.1, table 1.1, for elemental iodine, as well as for 4 different relevant aerosol size ranges. Likewise, an account of wet deposition relations and statistical distributions are given in Appendix 1 Table 1.2, which includes estimations of the fractions of contaminants that would be expected to be removed immediately from the surface with the depositing rainwater. In Appendix 1 Table 1.3, the corresponding figures are given for a case, where wet and dry deposition contributions are about equal in magnitude. Indoor deposition may also have importance, particularly in dry deposition scenarios, where people living in a contaminated urban area spend much of the subsequent time in locations close to unshielded contaminated surfaces. Appendix 1, section 1.2 gives an account of parameters governing the indoor deposition, and thus the resultant dose, with indications of uncertainties.

Appendix 1 section 1.2 gives an overview of parameters governing the post-deposition migration, including estimates of parameter uncertainties. The post-deposition migration information is needed to project doses into the future and, for example, to estimate residual doses that would remain after implementation of a recovery intervention strategy. The work here not only extends the traditionally deterministic ERMIN model to become probabilistic with proper parameter uncertainty estimates, but also introduces new measures to reduce the uncertainties associated with estimates of changes over time of contamination levels on different roof materials, and introduces a new model parameter set to enable distinction between vertical contamination migration in different types of soil.

Also, an account is given in section 1.3 of typical occupancy factor data (including uncertainty estimates) for different European cities. Particularly, the fraction of time spent outdoors is important; as people will then not be protected against outdoor contamination by shielding structures of for example a dwelling.

Where appropriate for the selected scenarios in the current project, these parameters have been applied in the sensitivity analyses described in sections 2.3.3, 2.3.4 and in Appendix 2.

Finally, some remarks on the likely uncertainties on expected countermeasure effect are given in Chapter 4. If site and case specific characteristics could be identified, it would be possible to considerably reduce the uncertainty ranges that these countermeasure effect parameters are associated with.

2.2.2 Sources of ERMIN uncertainty

Appendix 2 reports on an investigation into the relative significance of different sources of ERMIN on the residual dose predictions. It looks at the wider sources of uncertainty and qualitatively assesses their significance, it also documents a limited sensitivity analysis and uncertainty analysis that was used to further explore the parameter uncertainty using the findings of the parameter uncertainty investigation summarised in Section 2.2.1 and detailed in Appendix 1.

When looking at the different kinds of uncertainty; stochastic, judgemental, epistemological, computational, model-uncertainty and ambiguity (see Appendix 2, Table 2.1 to Table 2.5), it was concluded that stochastic and judgemental are the most significant sources and that computational and ambiguity are probably the least significant. The impacts of model-uncertainty are hard to quantify but situations such as extreme weather occurs very soon after deposition or a real environment that differs significantly from any of the idealised environments could add significant uncertainty to the predicted residual doses because of simplifications in the model formulation. Similar the impact of epistemological uncertainty is difficult to quantify but judged to be relatively small.

Appendix 2 also reports on a restricted uncertainty analysis to further explore the impact of stochastic and judgemental uncertainty of individual parameters of the ERMIN model. The analysis considered the parameters describing:

- occupancy,
- redistribution of initial deposition onto urban sources,
- weathering on the urban surfaces, and
- Soil migration.

Initial deposition to reference surface was not included although it is expected to be significant; however, it is know that dose will vary linearly with deposition and there is little information on the level of uncertainty of this input at the time of transition. Differences in unit dose rates and in environment, shielding properties were also not included, however the sensitivity was repeated for a number of different ERMIN environments in order to include some of the gross variability. Where possible the analysis was repeated for both wet and dry deposition situations. Figure 3 gives an example of some of the results from the analysis.

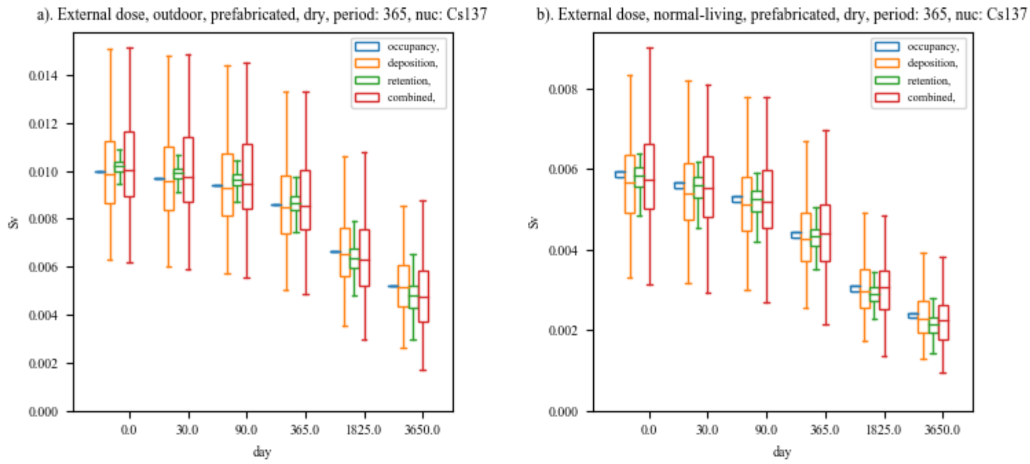

Figure 3. Example, box plots of a Monte Carlo analysis of predicted annual dose for outdoor and normal living assumptions following dry deposition. This is an example showing the effect of parameter uncertainty for different groups of parameters; occupancy, initial surface ratios, subsequent surface retention and all groups combined. This example shows a prefabricate building environment following dry deposition of ^{137}Cs. The full analysis is reported in Appendix 2.5.5 and Figure 2.25.

The unsurprising conclusion was reached that if a surface is an important contributor to dose in a particular environment, under particular deposition conditions (wet or dry) and under particular occupancy assumptions then uncertainty in the processes that operate on the surface become important.

Generally the analysis that the uncertainty of the redistribution of initial deposition onto urban sources to have the most significant impact on the uncertainty of the output residual dose prediction. The uncertainty on the initial ingress of radioactivity into buildings is particularly significant in high shielded environments, when dose-rates from internal surface are relatively significant proportion of the total dose, in low shielded environments it was less important since even to indoor location external surfaces contribute the most dose.

Whilst generally less important than the initial redistribution, retention on surfaces can have a significant impact; again it is retention on interior surface in high shielded environments where this is most pronounced, echoing the important of this surface under these conditions. Soil migration was found to be significant particularly under wet conditions and at longer times. The investigation also highlighted that there might be variations between the soil migration of different elements that are not satisfactorily captured by the current ERMIN model.

Occupancy parameter uncertainty was found to have the least impact on uncertainty of the residual dose predictions.

The impact of stochastic uncertainty can be overstated. For example, there are likely to be small areas of soil, certain roofs or building interiors which receive and retain radioactivity in greater amounts than the average parameters of ERMIN would suggest, similarly there will be surfaces that receive and retain much less. However, people generally move around and such stochastic variation is smoothed out. Judgemental uncertainty is therefore probably more important; the correct choice of the average parameters that ERMIN needs to predict residual doses that represent this smoothing. For example, incorporating the functionality to use different average soil migration parameters depending on soil type is arguably a worthwhile development for ERMIN, whereas a full uncertainty analysis that incorporates values from the extremes of the distributions for these soil type specific parameters may well just create spurious uncertainty.

Notwithstanding the constraints of the analysis and the uncertainties that could be included (deposition ratio, retention parameters and occupancy), it is tentatively judged the uncertainty on projected residual doses in ERMIN is less than an order of magnitude and for most environments much less. The most uncertainty in the predictions will be in the most heavily built up areas where shielding is high. Under these conditions the uncertainty on the ingress of material in the buildings and it subsequent retention is most significant.

3 Objectives and criteria

"…the primary goal of the entire recovery process will be to develop an agreed strategy for returning areas affected by the emergency to a state as close as possible to that existing before the release of radioactivity and the population to a lifestyle where the accident is no longer a dominant influence" UK Inhabited Areas Recovery Handbook (Nisbet and Watson, 2015).

The stochastic, judgemental and modelling uncertainties surrounding the assessment of the radiological situation, and in particular the residual dose, were examined in Chapter 2. Stakeholders and Decision makers at all levels must operate within those uncertainties to define a criteria and objectives and ultimately a strategy for achieving those objectives. Of course, the strategy may include options aimed at reducing uncertainties if that is appropriate.

3.1 Criteria

Criteria, likely to be expressed as some measure of projected residual dose, are applied to the radiological situation to delineate the extent and predict the duration of the situation.

"*Reference level; In emergency or existing controllable exposure situations, this represents the level of dose or risk, above which it is judged to be inappropriate to plan to allow exposures to occur, and below which optimisation of protection should be implemented. The chosen value for a reference level will depend upon the prevailing circumstances of the exposure under consideration*." (ICRP, 2007).

Setting, refining and revising criteria is a complex optimising process undertaken not only under the uncertainty of the evolving radiological situation but also the uncertainty of heterogeneous multi-use urban landscape, uncertain prognosis for interdependent social, economic and environmental spheres and the uncertainty of a decision process with many different stakeholders participating.

3.2 Objectives

Objectives define what is intended to be achieved. They can be split into fundamental objectives which define goals, and means objectives which define how fundamental objectives should be achieved. For example, a fundamental objective could be to bring the residuals doses down to below the criteria within a given time frame. Additional means objectives may require that waste is to be minimised. Not all fundamental objectives are radiological in nature, for example, there may be an objective for economic recovery.

Setting, refining and revising objectives is a process taken against a backdrop of uncertainty.

4 Implementation of strategies

A strategy is a package of management options brought together to achieve a set of objectives. This chapter looks at the difficulties decision makers have in putting together strategies and the tools available to them (Section 4.1), it discusses the factors that impact on the effectiveness of options in reducing residual dose (Section 4.2) and finally it considers the uncertainty that arises in predicting the reduction of residual dose when combined (Section 4.3).

4.1 Developing strategies

Decision-makers for recovery in urban areas have available a wide range of management options to achieve their objectives. For example, the EURANOS Handbook for Management of Contaminated Inhabited Areas in Europe contains a compendium of 59 radiological management options for inhabited areas, to which can be added a large number of non-radiological options.

In exercises and studies, decision-makers have found it hard to select and combine options. There are many reasons for this, but they all fundamentally reduce to profound uncertainty over which combination would be optimal. Optimal in this case means the one that most effectively meets the objectives. This is because there will be considerable uncertainty of how a heterogeneous urban area with intersecting physical, social, economic and environmental spheres will respond to combinations of radiological and non-radiological options. It should be noted there may be ambiguity, inconsistency and uncertainty in the objectives themselves.

Several tools have been developed to assist in the process of selecting and combining options, for example:

- the EURANOS Generic Handbook for Assisting in the Management of Contaminated inhabited areas in Europe following a radiological emergency (Nisbet et al, 2010),
- the ERMIN model (Charnock et al, 2009) (see section 2.2), and
- the HARMONE Guidance Handbook for Recovery after a Radiological Incident (Charnock et al, 2017) .

The EURANOS handbook contains a compendium of management options that includes information beyond the radiological effectiveness, such as environmental impact and social factors. While the handbooks have been customised for a number of difference European countries (see for example the UK recovery handbook (Nisbet and Watson, 2015)), the information is still perforce generic, and it is up to the stakeholders and decision-makers to consider how it applies to the actual local situation.

ERMIN is a model and therefore to some extent it is customisable to a local situation (if the situation specific parameters are available); the user can choose between different environment types, specify the radionuclide mix and whether the deposition was wet or dry, for example. However, ERMIN provides little beyond radiological and broad resource requirement endpoints. A useful endpoint from ERMIN is the projected residual dose given by urban surface; by identifying which surface contributes the most to dose the user is able to reduce the number of possible options to a much more manageable number.

In the development of the HARMONE guidance handbook, a defined decision-making process was applied to greatly simplified and idealised situations and objectives, to construct an 'example' recovery strategy. The intention is that the example strategy provides a starting point that can be adapted for a specific situation and modified and augmented to account for more complex sets of objectives.

4.2 Factors effecting optimal implementation

When dose reducing management options / countermeasures are implemented, it is important not only to select the optimal management strategy, but also to implement it optimally. To do so, simple local investigations of the contamination and its horizontal and vertical distributions are needed. It has been demonstrated that such practical investigations to guide the countermeasure implementation process, and ensure that contamination is for instance removed to the optimal depth in surfaces, can be decisive in whether the implemented strategy reduces the dose rate by a factor of about 5 or gives no dose rate reduction at all (Andersson, 2009). They are also important in securing that countermeasure implementation does not lead to production of excessive amounts of radioactive waste to be managed. However, in spite of the obvious need, guidelines for measurement strategies to guide countermeasure implementation are not available.

It is of great importance to enable implementation of at least some recovery countermeasures very early, as they will then be very highly cost-effective and more dramatic and expensive countermeasure implementation can be avoided. An example is lawn mowing (particularly in dry deposition scenarios), which can, if carried out within few days or weeks, remove much of the contamination on a lawn before it reaches the underlying soil, where the treatment will be much more problematic. A requirement in this context is of course that prior considerations have been made of what would be practically possible and acceptable to implement, so that it could be secured that the required resources to implement the countermeasure are in place when needed. Also for this purpose, generic European guidelines are needed.

Wider social and environmental uncertainties are discussed in Chapter 5. This section focusses on the uncertainties of the projected residual dose with management options applied by extending the sensitivity analysis of ERMIN in Chapter 2. Again, residual dose is the focus because whilst it is recognised that there are many other radiological and non-radiological endpoints that might be of concern to the decision-maker, the residual dose is likely to be of primary concern.

The countermeasures from the HARMONE guidance handbook included in this study are; roof brushing, vacuuming and washing of internal (indoor) surfaces, cutting of grass, removal of plants, rotovating of soil, and removal and replacement of topsoil.

Looking first at roof brushing, the available data from testing suggests that the decontamination factor (DF; reduction factor in contamination level) by this method would be in the range of 2-7. This is provided that the method is carried out in agreement with the instructions given in the EURANOS handbook. If the method is applied wrongly, it may have much less effect, and this goes for all methods. As written in the EURANOS handbook, the quoted range of DF can in the short term (first few weeks) be considered to be the same for all radionuclides (except elemental iodine and tritium for which removal is likely to be virtually full), but with time, specifically cationic caesium will become stronger bound to the surface, and a value in the low end of the DF range (2-3) should be expected. However, this depends on the surface material and state of the surface (e.g., extent of cover with removable organic matter). The DF range thus represents a methodological effectiveness uncertainty, largely depending on the exact type and condition of the brush and surface as well as on the water temperature, and an implementation uncertainty (although things are assumed to be done in accordance with the method description, and a given amount of time and water is invested, the work can still be done more or less thoroughly). These two types of uncertainty components exist for essentially all remedial countermeasures. As the method is in this study meant for implementation in the late phase, the effect on a cationic caesium contamination, which is the focus radiocontaminant in this study, may be expected to be uniformly distributed between 2 and 3.

Vacuuming of indoor surfaces, although in this study (see Appendix 2 Table 2.13) assumed to be carried out late, would be likely to be a routine household procedure, that would be carried out on floors of dwellings at least within few weeks of the contaminating event. As the submicroneous aerosols assumed in this study to be carrying the cationic caesium will deposit and accumulate on somewhat larger house dust particles, which are more easily removed by vacuuming, a surface decontamination factor of 5-10 may be achieved (large uncertainties depending on vacuum cleaner type and type and state of surface). The floor is likely to become the single most contaminated indoor surface, but if other indoor surfaces are not treated, the overall indoor surface DF should probably be divided by a factor of 3 (Andersson et al, 2004; Lange, 1995).

Indoor washing would also be considered to be associated with considerable uncertainty, and a uniform distribution of DF between 1.5 and 3 (as indicated in the EURANOS handbook) may be expected. If this is not carried out within the first few weeks, the DF range is likely to be only 1.2-1.5.

The effect of cutting (and removal) of grass is very highly sensitive to the weather (precipitation) between the time of deposition and the time of countermeasure implementation. Generally, the method should be implemented before the first significant rain occurs, and the essentially useful time scale may therefore be from 0 days to a few weeks. The method can, for instance as a 'self-help' countermeasure (see Chapter 5), be carried out early enough to have a good effect. If the weather has remained dry, the available literature shows that the natural weathering removal of the grass contamination will occur with a half-life of about 2-3 weeks (Nielsen and Andersson, 2011). Depending on primarily the weather conditions (also wind), the length of the grass as deposition and the cutting height, the DF can, as stated in the EURANOS handbook, vary anywhere between 2 and 10 assuming very early countermeasure application after dry deposition. In wet deposition scenarios, the countermeasure would be likely to have very little effect, as much of the contamination would have been washed directly into the underlying ground.

Removal of plants is like grass cutting assumed to be carried out very soon after the deposition and before the first rain. If so, a decontamination factor of 2-10 is possible, as mentioned in the EURANOS handbook. The wide range reflects the weather conditions, type and size of vegetation, and removal methods applied.

Rotovating is a mechanical countermeasure which mixes the upper layers of soil fairly uniformly within a relatively shallow depth. The effect is according to the EURANOS handbook a reduction of the dose rate at 1 m above a large contaminated surface by a factor (surface dose rate reduction factor; DRF) of 2-3 if the contamination was still in the upper few cm (depends on contaminant, soil type and time, as well as exact mixing depth of the equipment). If it is assumed that the countermeasure will be implemented at least within a year or two, this will likely be the case for the targeted caesium contamination.

Removal and replacement of topsoil is a method that has been tested widely in contaminated areas. If the operation is optimised according to the penetration of contaminants in the soil (requires an in situ measurement strategy to avoid removing excessive soil masses) the DF that can be achieved could according to the EURANOS handbook be up to 30. However, considering that lack of local experience, differences in soil texture, weather conditions, ground evenness, heterogeneity of the vertical contaminant distribution and time can limit the success, and all these influences are largely unknown/undefined in the case scenarios (except that the work is not done very early), it may on the basis of field studies on aged contamination (Andersson, 2009; Roed et al, 2006) be more prudent to assume a DF range of 6-10.

In general, it would be problematic to suggest other statistical distributions than uniform for the above DFs and DRFs, as they depend on a range of case specific parameters, which cannot be nearly fully defined generically or even for a given location. The available data is inadequate to suggest that one part of the range is more likely than another.

4.3 Predicting residual dose after clean-up

A key question when considering applying a clean-up option or set of clean-up options is, by how much will the residual dose be reduced? The sensitivity analysis summarised in Section 2.2.2 and detailed in Appendix 2 was extended to include clean-up options applied separately and together as strategies. Since it is not possible to consider all combinations of options the analysis focusses on the HARMONE 'model' strategies.

In ERMIN, the action of the different types of option is described by sets of parameters. For example, removal of radioactivity is described by particle group and element dependent decontamination factors (DF), as well as further parameters that describe how that DF varies with time following deposition, whereas a soil mixing technique is defined by a matrix that describes how layers of soil are redistributed into new layers. Section 4.2 describes some of the variations in effectiveness that have been seen when clean-up operation have been applied and are typically expressed as, for example, ranges of DF.

A full sensitivity analysis might look at the uncertainties involved with these parameters to explore how they impact on project residual dose following clean-up, for example by sampling the given ranges in some way. However, there are problems with this approach because, as Appendix 2.5.6 demonstrated, much of the uncertainty associated with an option effectiveness may be derived from the uncertainty in model parameters particular surface retention including soil migration (already considered in Section 2.2.2), combined with the uncertainty in the time of application. Therefore, the sensitivity analysis considered only those sources of certainty.

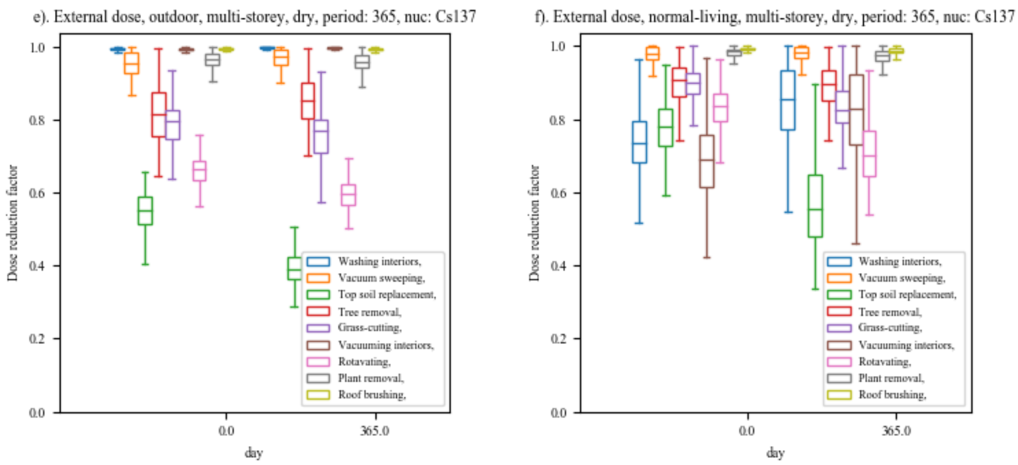

Figure 4. Example box plots of a Monte Carlo analysis of predicted dose reduction factor on annual dose for various environments and locations following dry deposition from all surfaces. Each series represents an option applied using the default effectiveness parameters from the ERMIN database and sampling model parameter and timing parameter distributions. The doses given are the 1st year dose staring from time 0 and the second year starting from time 365 days. This example shows a multistorey building environment following dry deposition of ^{137}Cs. The full analysis is reported in Appendix 2.5.6 and Figure 2.29.

The striking result when considering options separately is that there is considerable likelihood of many of the options being ineffective in reducing dose (Figure 4, and also see Appendix 2, Figure 2.29). This maybe because they are targeting a surface that is an insignificant contributor, or it may be that the combination of timing and retention factors is far from optimal. Generally speaking the soil and grass surface techniques (grass cutting, rotovating and top-soil removal and replacement) are the most consistently effective and are seldom completely ineffective. The more effective a technique the wider the range of uncertainty, for example top-soil replacement normal-living dose reduction factor ranges from about 0.3 to 0.9 following dry deposition in the multi-storey environment in the second year.

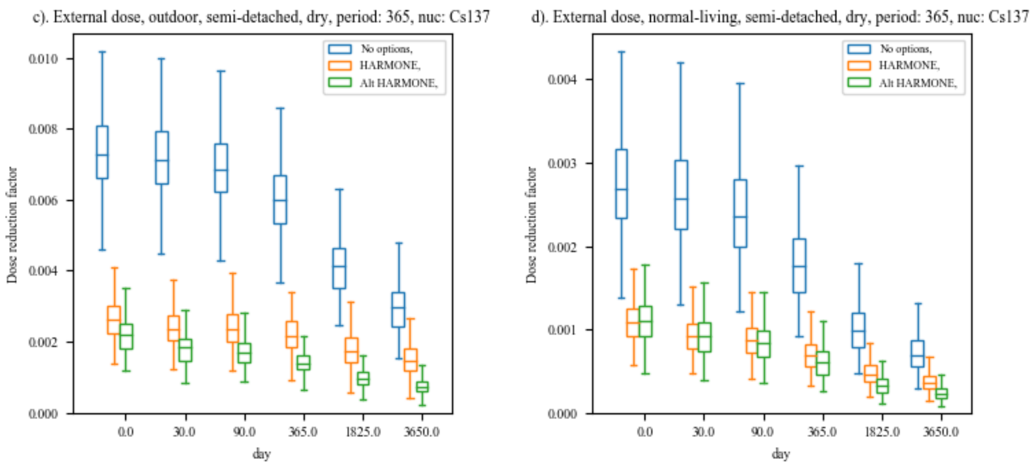

Figure 5. Box plots of a Monte Carlo analysis of predicted annual dose for various environments and locations following dry deposition with no options applied and with two possible strategies applied. The full analysis is reported in Appendix 2.5.6 and Figure 2.30

When considering the options applied together as strategies (see Appendix 2, Figure 2.30) they were always very effective. This is because the strategies were comprehensively applied to most of the urban surfaces; so even if a single option is relatively ineffective because the surface that it targets contributes only a small part of the total dose, it naturally follow other options will be more effective because the other surfaces they target must contribute more.

5 Environmental and social aspects

It is essential in justifying and optimising any intervention to ensure that the disadvantages introduced by the intervention (including costs, harm and social disruption) are offset by the advantages, so that the net benefit of intervening is positive. Indeed, the optimum protection option is not necessarily the option that results in the lowest residual doses. Some options may result in a lower residual annual dose but give a smaller net benefit than the optimum option (ICRP, 2000).

The impact on the environment and on society and individuals of intervention must be taken into account. Countermeasures may impact on social values in different ways (Hunt and Wynne, 2002), and can impact in different ways on different affected population groups / stakeholders. Ethical considerations are therefore important in reaching decisions on intervention strategies. Particularly important issues to consider in this context are (Oughton and Forsberg, 2009):

- Disruption of everyday life and self-help
- Free informed consent of workers to risks of radiation and chemical exposure and consent of private owners for access to property
- Distribution of dose, costs and benefits
- Liability and/or compensation for unforeseen health or property effects
- Change in public perception or use of an amenity
- Environmental risk from ecosystem changes, groundwater contamination, etc.
- Environmental consequences of waste generation and management

Dose reductive countermeasures can have considerable impact on the environment - e.g., with respect to pollution and future land (soil digging, ploughing) - and may affect cultural heritage, damage property and expose population groups (e.g., clean-up workers) to risks. They may for example also affect population behaviour, lead to loss of amenities, and have other (both positive and negative) side effects including cleaning and renewal of surfaces (Nisbet et al., 2007).

Also, there are social/ethical factors that may influence the dose-reductive effect of some countermeasures. For instance, reliance on voluntary behaviour may not be straightforward. Acknowledging the ethical key principle of free informed consent has for instance in connection with clean-up of village areas after the Chernobyl accident been found to limit the dose-reductive effect of countermeasure implementation. When the Russian army decontaminated 93 contaminated settlements in the Bryansk region in 1989, the most important countermeasure was topsoil removal. However, in an area with small, open ground lots, if a ground lot owner chose not to allow their garden area to be treated, the dose contribution from contamination in their garden to the neighbour's dose remained significant (Andersson, 2009).

Specifically in relation to acceptability of waste treatment and storage issues, there may also be important issues to consider. The Fukushima accident and the subsequent clean-up work demonstrated the importance of this problem, and of waste minimisation strategies, which require measurements to guide optimised practical (in field) countermeasure implementation, as outlined in section 4.

As stated by Oughton and Forsberg (2009), 'voluntary countermeasures that are carried out by the public or affected individuals themselves, or that increase personal understanding or control over the situation, are usually deemed positive as the action respects the fundamental ethical values of autonomy (i.e., respect for freedom of choice of individuals), liberty and dignity'. People are enabled to actively do something to positively affect their own situation, while in the same process gaining a better understanding of the actual physical problem. 'Self-help' countermeasures can also be comparatively attractive in terms of labour costs. On the negative side, there will always be a risk of misunderstanding or misinterpretation of instructions given, which might in some cases lead to an irreversible worsening of the situation. It is therefore of great importance to provide carefully worded and detailed instructions to the people that participate. With a 'self-help' work force it would be possible to more rapidly carry out a countermeasure like grass-cutting or plant removal (both considered in the countermeasure strategies considered in this project; see Appendix 2, section 2.4.5), which must be carried out within days or weeks (depending on local weather) to be effective. The possibility to draw upon such a work force may thus affect the countermeasure effectiveness.

So-called 'social countermeasures' (primarily focusing on other benefits than dealing with the environmental contamination) may also be implemented. These may be seen as tools to enable the affected populations to better cope with the exposure situation, for instance also by avoiding exposure where possible. 'Social countermeasures' of potential relevance for consideration for implementation in the transition phase in a contaminated inhabited area include (Oughton and Forsberg, 2009):

- Provision of radioactivity or dose measurement equipment
- Altering intervention limits (can backfire in losing trust in authorities)
- Compensation scheme
- Information/advice bureau
- Education programmes in schools
- Medical check-up
- Stakeholder and public consultation methods

On top of the assessments of uncertainties in estimating not only the dose reductive effect, but also other cost-benefit evaluation elements, it is necessary to have a strategy for how to deal with the uncertainties (also those that cannot be quantified in absolute figures) in reaching decisions. Essentially, this is an ethical issue, and the principle of respect for autonomy, as well as the precautionary principle, require that measures with uncertain consequences be discussed with those potentially affected (Oughton and Forsberg, 2009).

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D 9.20 Appendix 1.

Urban Scenario Parameter Uncertainty

Final

Version 1.0

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1 Appendix Urban Scenario Parameter Uncertainty

The ERMIN model for urban dose estimation in the European state-of-the-art decision support systems was originally not built for probabilistic assessments, and although rough indications of parametric uncertainties have previously been indicated, it was clear that an effort was needed in the current project to improve on the quality of these parameter uncertainty indications, taking the latest information into account and considering a wider range of case specific parametric options, thus reducing the overall prognostic uncertainty.

As discussed in the analyses in Appendix 2 of the manifold different types of uncertainties associated with modelling of doses received in urban contamination scenarios, due to practical constraints, the focus in the analyses carried out in this project of the impact on dose endpoints has been limited to cover those stochastic and judgmental parameter uncertainties that are judged to have the greatest impact. The most important urban dose calculation parameters are from experience found to be those describing the deposition of contaminant particles and gases with different physicochemical forms on different types of surface in the environment, those describing the retention and migration with time after the deposition of the same contaminants on/in the different types of urban surface, and the occupancy, that is the fractions of time spent by people in different locations outdoors and indoors in the environment, as dwellings can, depending on their construction, protect inhabitants well against contamination present on surfaces in the outdoor environment.

1.1 Contaminant characteristics and deposition

In the transition phase to long term recovery, some initial measurements of overall contamination level on easily measurable reference surfaces in the environment may be expected to have been made, but a detailed account of levels of contamination on the different types of surface of different orientation and texture in the inhabited complex is unlikely to be readily available. Nevertheless, such data is needed as a starting point in preparing for recovery where necessary, and it would then be necessary to predict the levels of deposition on other types of surface that could occur for a given accumulated air concentration or reference surface contamination concentration.

The contaminant deposition is greatly dependent on the weather conditions during the passage of the contaminated plume, the physicochemical properties of the contaminants (particle size, reactivity) and the types and orientations of the surfaces in the inhabited area to which the deposition occurs (Andersson, 2009). It should thus first be considered which physicochemical forms the various likely contaminants from a major NPP (nuclear power plant) accident could be expected to have. A ‘consensus’ list of contaminants considered potentially important by Slovakia, France, Germany, Finland, Czech Rep. and USA for evaluation of radiological consequences in case of severe NPP accidents comprises (apart from noble gases) radionuclides of the following elements: Am, Ba, Ce, Cs, Cm, I, La, Mo, Nb, Np, Pu, Rb, Ru, Sb, Sr, Te and Zr (Bujan, 2014). The physicochemical forms of these in future accidental releases will depend on a complexity of processes and conditions during the release, and are difficult to predict. However, the experience from history’s two large nuclear power plant accidents, the Fukushima and the Chernobyl accident, provide very useful information on what might be expected in some different types of scenarios, and for instance which sizes, materials and thus aerodynamic behaviour the produced aerosols might be expected to have under different conditions. A literature search has been made on the characteristics of radiocontaminants that might be released to the environment in a nuclear power plant accident, and on which relative initial contamination levels on different surfaces in the inhabited environment would be likely to occur for contaminants with different characteristics (Hinrichsen & Andersson, 2018).

The radionuclide composition of released contaminants will depend on the source, while contaminant characteristics such as particle/gas release fractions, particle size distribution, solubility and oxidation states will also depend on the release processes, in particular on the temperature, pressure and the presence of air/oxygen (Lind, 2006; Lind et al, 2009; Salbu, 2001).

One of the most volatile contaminants (except noble gases) is iodine, which may be released in its elemental gas form (which has a very high deposition velocity to surfaces), in organic gas forms (where the deposition velocity is comparatively insignificant and thus in practice unimportant), and as condensed vapour on ambient aerosols, typically resulting in an AMAD (activity median aerodynamic diameter) in the range of 0.5-1 µm, which would have an intermediate deposition velocity (Andersson, 2009). Comparatively very high release fractions of iodine were as expected reported both in connection with the Chernobyl (0.2; IAEA, 1991) and Fukushima (0.0002; Le Petit et al., 2014) accident. Iodine aerosol spectra obtained at different distances after the Chernobyl accident show a perfect Gaussian distribution with no signs of bimodality (e.g., Reineking et al., 1987; Jost et al., 1986), with an AMAD of about 0.5 µm, which is slightly smaller than that of the corresponding Cs aerosol. This iodine aerosol size distribution compares well with that registered after the Fukushima accident (Kaneyasu et al., 2012). However in these measurements the size distribution is a complete match with that for caesium, indicating that insignificant quantities of larger (fuel fragment) particles containing traces of caesium were at the times of measurement released at Fukushima. This suggests that the aerosol iodine can essentially be assumed to be purely condensed mode (on ambient particles). This is in-line with the high solubility and initial post-deposition mobility recorded for all the deposited iodine from Chernobyl at different distances (see, e.g., Roed, 1990).

At the other end of the volatility spectrum, it was in connection with the Chernobyl accident found that contaminants of certain elements, which were not reported after the Fukushima accident, where the explosions were less powerful, were only released to the atmosphere in the form of comparatively large low solubility fuel particles, indicating that these would in general be expected to be highly refractory (undepleted from the fuel). These comprised ^{95}Zr, ^{95}Nb, ^{140}Ba, ^{140}La, $^{141/144}$Ce, $^{237/239}$Np, $^{238\text{-}242}$Pu, $^{241/243}$Am and $^{242/244}$Cm (Bobovnikova et al., 1990, Loschilov et al., 1992, Kuriny et al., 1993, Kashparov et al., 2003; Salbu et al., 1994). It cannot be ruled out that future accident scenarios might lead to releases of fuel particles. Apart from the fuel particles with sizes allowing them to follow air streams, part of the released fuel from the Chernobyl accident was in the form of either very large fuel fragments spread ballistically by the power of the release process, or very large conglomerates of nuclear fuel fused with melted zirconium (Kashparov et al., 2003). This part of the contamination was mainly in a form with a size range from several tens to more than a thousand microns (Kashparov et al., 2003), and mostly deposited within the nearest 2 km (Kashparov et al., 2003) - a zone where it makes absolutely no sense to attempt to model the contaminant distribution through atmospheric dispersion modelling. These huge particles/fragments, although probably locally dominant in some areas over very small distances, are estimated to contain only a small fraction of the total contamination (Kashparov et al., 1999). It can thus be assumed that nearly all atmospherically dispersed particles carrying Zr, Nb, Ba, La, Ce, Np, Pu, Am and Cm are fuel aerosol particles. Measurements made after the Chernobyl accident showed that the smallest of these particles (which reached great distances) had a size of about 4 µm (Reineking et al., 1987; Rulik et al., 1989, Mala et al., 2013). Kashparov et al. (1996, 1999) reported of a fuel aerosol particle median diameter of some 5-6 µm corresponding to a crystallite size of the fuel. This actually seems consistent with results of smaller explosive tests (although clearly much less powerful) interacting on a matrix of uranium dioxide (Harper et al., 2007), where the smallest particles were found to be some 4 µm, but the greatest part of the aerosolised mass was in the ca. 5-20 µm range. In addition to pure fuel (uranium oxides) particles, also, fuel mixes with construction materials and fire extinguishing materials have been reported in the near zones after the Chernobyl accident, which could have a different environmental mobility (Dobrovolsky & Lyalko, 1995; Lind, 2006).

Quite large (and comparable) fractions of Cs, Te and Rb (and to a somewhat lesser extent Sb and Mo) were released in connection with the Fukushima accident (Le Petit et al., 2014), and these should, based on Chernobyl data (e.g., Bobovnikova et al., 1990, Loschilov et al., 1992, Kuriny et al., 1993) be expected to a considerable extent (probably somewhat less for Sb and Mo) to be volatilised from the fuel, forming submicroneous condensation particles. In the powerful Chernobyl explosion case investigations by Kuriny et al. (1993) show that even at distances up to about 50-60 km in some directions from the Chernobyl NPP, most of the deposited caesium was in the form of fuel particles. This agrees with results of experimental investigations of the effect of decontamination operations (water hosing on impermeable surfaces) carried out in Pripyat and hundreds of km away from the Chernobyl NPP, where the contamination was much easier removed in the nearest areas where it was associated with large low-solubility fuel particles (Andersson, 2009). The data of Salbu et al. (1994) show that the relationship between Sr-90 and Cs-137 in fuel particle deposition dominated areas was roughly 10 times higher than that in condensation particle deposition dominated areas. This can be taken as an indication that the fuel particles may have been depleted about 10 times more with respect to Cs than with respect to Sr. Some association with fuel particles could explain the slightly bimodal Cs-137 aerosol distribution measured by Reineking et al. (1987) as far away as Göttingen in central Germany after the Chernobyl accident, clearly showing the presence of some supermicron particles, which would be expected to have low solubility (Andersson, 2009). Again, the depletion fraction would be expected to vary according to the exact accident scenario conditions. The caesium aerosol measured after the Fukushima accident was generally submicron and characteristic of condensation mode (Kaneyasu et al., 2012), even though surprising processes some days after the start of the Fukushima accident also seem to have resulted in creation of some homogeneously caesium-containing spherical low solubility particles in the 2 µm range (Adachi et al., 2013). In connection with the Chernobyl accident, single element particles (e.g., ruthenium, caesium) were recorded more than a thousand km from Chernobyl (Salbu, 1988), indicating the complexity of processes during the release.

As for strontium, both fuel particle and small condensation aerosol fallout has been reported from the Chernobyl accident (Kashparov et al., 2003; Salbu et al., 1994). In the Chernobyl 30 km zone Konoplev et al. (1993) and Askbrant et al. (1996) reported that 80-90 % of the strontium was associated with fuel particles. Even more than, a hundred km away from the Chernobyl NPP, fuel particles constituted a significant part of the strontium contamination (Kuriny et al., 1992). The 'duality' of the fuel particles and condensation aerosols carrying strontium from the Chernobyl accident can be illustrated through the results of modified Tessier type sequential extractions (see Tessier, 1979) carried out on soils contaminated with Chernobyl Sr at various distances from the Chernobyl NPP (Salbu et al., 1994). In the nearest investigated areas (at 50 km distance), by far the greatest part of the strontium in the soil was in strongly bound forms that could only be extracted with hydrogen peroxide or nitric acid, whereas in areas at greater distances (170-450 km), by far the majority of the strontium was in much more easily soluble forms. Parallel tests with stable Sr were used to rule out effects of the different specific soil types. It should be noted that since Sr-89, Sr-90 and Y-90 cannot be determined in straightforward gamma spectrometry, but usually require chemical separation of strontium from other radionuclides in the sample, prior to radiometric analysis, they are 'inconvenient' to study for instance in aerosol samples, where they have to a large extent been ignored both after the Chernobyl and the Fukushima accident (Steinhauser, 2014). However, even in the Fukushima case, also ^{90}Sr contamination has been measured, in the vicinity of the Fukushima NPP (Steinhauser et al., 2014), at reported levels of about 1 kBq/kg soil (note: as this figure was published without indications of the depth/dimensions of the soil sample taken, it only qualitatively indicates the presence of strontium).

Ruthenium is special in that it has a very high elemental boiling point (2700°C), which would in practically any conceivable incident scenario prevent it from being volatilised and depleted from fuel material. However, if oxygen is present, it can be oxidised to its tetraoxide form, which is highly volatile (Kashparov et al., 1996; Hunt et al., 1994). From the Chernobyl accident, ruthenium radionuclides were in great amounts dispersed as condensation particles. This would be expected to have occurred in connection with the fire that followed the explosion. In fact, more ruthenium than caesium was released in connection with the Chernobyl accident (IAEA, 1991), and this had a considerable impact on doses over the first few years (^{106}Ru has a half-life of very close to 1 year). The explanation offered by Le Petit et al. (2014) as to why only small amounts of ruthenium were measured in the environment after the Fukushima accident was that it seems that the fuel remained under water in the spent fuel pools (thus no air ingress). Instead, the low volatility of ruthenium is reported to be consistent with overheating and fuel melting of reactor cores. Oxidation could in reality occur in all accident scenarios currently represented in RODOS (Bujan, 2014). However, since this is a critical parameter, and oxidation obviously may not always be expected, it would be useful to run the DSS with different assumptions in this respect, both for training purposes and for early prognostic runs, when actual scenario specific processes have not yet been disclosed through measurements. It is well known that ruthenium in irradiated UO_2 fuel appears in small metallic alloy precipitations together with other fission product elements such as molybdenum, technetium, rhodium, and palladium (Ver et al., 2007). Such precipitations are in metallographic images seen as generally spherical white inclusions.

It is difficult to predict the physicochemical forms that would arise in any future nuclear power plant accidents, as these would be largely dependent on the exact inventory and accident processes at the NPP. Although for example the international Phebus Fission Product Programme (Gonfiotti and Paci, 2018) shed some new light on possible releases in different NPP accident processes, the results reflect specific conditions and do not provide the range of details needed in operational nuclear preparedness for a specific NPP construction. However, perhaps in the future, results of such investigations could be used together with for example the Rapid Source Term Prediction (RASTEP) system (Knochenhauer, 2013), focusing on estimating the state of the specific NPP at the time of the accident using a Bayesian belief network to provide a probabilistic overview of possible accident states. By estimating the processes at the NPP, also the physicochemical forms of the various potentially released contaminants could be estimated. In a recent publication Havskov Sørensen et al. (2018) comment on the requirements to do this.

1.1.1 Dry deposition parameter uncertainty

The concentration whereby dry deposition (deposition in absence of precipitation) occurs is usually expressed in terms of a dry deposition velocity (v_d) which was defined originally by Chamberlain (1953):

V_d (m s^{-1}) = Deposition Flux (Bq m^{-2} s^{-1}) / Atmospheric Concentration at Reference Height, c (Bq m^{-3}),

in which the deposition flux is defined as

Deposition Flux = K_p dc/dz, where dc/dz is the concentration gradient at the height z above the surface in question, and the negative sign denotes that the deposition flux is in a direction opposite to the concentration gradient. K_p is a diffusion coefficient, which is usually assumed to be the same for particles and gases, and can approximately be described by

$K_p = k^2 z^2 du/dz$,

in which k is von Karman's constant (0.41), and u is the horizontal wind speed. Thus, du/dz is the wind speed gradient according to height (Nicholson, 2009).

It is important to note that the background data for the deposition parameter values given in Table 1.1 may possibly not reflect the full range of possible parametric variation, as they are generally taken from a limited number of actual sets of environmental observations of deposition velocity of elemental iodine and relevant aerosols with different AMADs on different surfaces in connection with the Chernobyl accident, the Fukushima accident and various experimentation (Atkins, 1967; Belot, 1977; Bonka, 1989; Bonka & Horn, 1980; Chamberlain, 1953; Chamberlain, 1967; Clough, 1975; Collins et al., 2004; Freer-Smith et al., 2003; Garland, 2001; Horn et al., 1988; Jonas, 1984; Jonas & Vogt, 1982; Kashparov et al., 1999; Lai & Nazaroff, 2005; Little, 1977; McMahon & Denison, 1979; Maro et al., 2014; Mück et al., 2002; Nicholson, 1989; Nicholson & Watterson, 1992; Petroff, 2005; Roed, 1985; Roed, 1987; Roed, 1988; Roed, 1990; Roupsard et al., 2013; Schwartz, 1986; Sehmel, 1973; Tschiersch et al., 2009; Vargas et al., 2016; Watterson & Nicholson, 1996). However, these relations between deposition on different surfaces *in the same scenario* are obviously associated with comparatively much less variation than would relations between deposition velocities in general to these surfaces. The deposition relations can be assumed to remain the same regardless of the actual quantity of each contaminant that has deposited. This means that all dose estimate figures can be scaled according to the actual levels of contamination, which is ideal for the purpose of the transition phase scenario evaluations in the present project.

For example, deposition velocity depends on atmospheric stability. It has been demonstrated that under moderately stable atmospheric conditions (e.g., night time with clear sky), the friction velocity will only be about half of its value under neutral conditions (Jensen, 1981). This in turn means that the eddy diffusion part of the deposition velocity will be reduced to about a quarter (IAEA, 1994).

Also, wind velocity can greatly influence deposition velocity. It has been demonstrated (Ahmed, 1979) that between wind velocities of 2 and 14 m s^{-1}, the deposition velocity of naturally occurring radioactive aerosols increases by about a factor of 3, both to smooth (e.g., filter paper) and rough (grass) surfaces. It has also been shown (Freer-Smith et al., 2003; Slinn, 1982) that deposition velocities of ca. 0.8 µm particles to trees can increase by a factor of 3-4 between wind velocities of 3-9 m s^{-1}. Even at moderate wind velocities (< 5 m s^{-1}), the deposition of particles on walls facing the wind direction can be several times higher than that on leeside walls, for particles of sizes between about 10^{-2} and 20 µm (Freer-Smith et al., 2003). As the particle size increases beyond about 20 µm, the influence of wind speed on deposition increases markedly, due to the significance of the inertial impaction mechanism (Ahmadi & Li, 1999). However, such large particles will in any case only remain airborne for short time, due to their large mass, and radionuclides associated with these would thus only contaminate rather small areas, depending on, e.g., the initial plume rise height (Hage, 1961).

Finally, surface roughness is an important parameter. An indication of this influence can be seen from measurements made in the Roskilde area after the Chernobyl accident. Here deposition velocities to grassed surfaces varied rather widely (Roed, 1990) between 1.8 and 8.8 m s^{-1}. However, if the length of the grass is taken into account (by dividing with the grass mass per unit area), the results are consistent within 10 %. It should therefore be noted that grassed areas in inhabited environments must be well-defined with respect to roughness (grass length). Differences of up to about a factor of 2 have been recorded (Lai & Nazaroff, 2005) for deposition velocities of 0.9-9.1 µm particles to vertical sandpaper surfaces, ranging from Sand 60 to Sand 220. As shown in Table 1, dry deposition will vary to roof pavings of different materials having different roughness.

Deposition to coniferous trees and deciduous trees in leaf would according to available literature be similar (Jonas, 1984). However, during the winter period where deciduous trees are leafless, the deposition to these would be very low. According to measurements made after the Chernobyl accident (Roed, 1988), the needles or leaves receive some 98 % of the bulk 0.7 µm aerosol deposition on a tree. However, relatively not quite insignificant deposition velocities of trace particles have been reported to bare trees in forests (Höfken et al., 1981) (ca. 10-30 % of that to the same trees in leaf). This is explained by a higher wind speed in a forest with bare trees, but this effect would not be expected to be relevant for single trees in an inhabited area (Jonas, 1984). Only trees in leaf are thus considered in the table. In the period where they are not in leaf, the deposition to these surfaces would be assumed to be negligible.

Unfortunately, no measurements of deposition velocities on surfaces in inhabited areas were reported after the Fukushima accident, where the focus was on measurements of dose rate, which of course comes to varying extent over time from different contaminated surfaces in the environment.

In the ERMIN model deposition on different surfaces in the inhabited environment is dealt with relatively to the deposition to a defined reference surface - in this case a newly shortcut lawn was selected (here a quick measurement of the relationship between deposition on the grass and the underlying soil can also give a useful indication of the local extent of dry and wet deposition). In ARGOS and RODOS, the deposition process to the reference surface is dealt with in the applied atmospheric dispersion model tool, and not in ERMIN. ERMIN has been designed on the background of the Chernobyl and Fukushima experience to hold information for elemental iodine gas and for aerosols in four characteristic groups with different size ranges (AMAD less than 2 µm, 2 - 5 µm, 5 - 10 µm and 10 - 20 µm). The initial surface contamination relations within each group are all assumed to be representable by normal distributions. Typically reported values of the dry deposition velocity in units of 10^{-4} m/s to the reference surface are for these contaminant groups respectively of the order of 20, 4, 7, 30 and 130 (see references above), but case-specific factorial dependencies and thus overall uncertainties are large as explained above.

In relation to roof pavings of different materials, some new detail variations compared with defaults in ERMIN have been introduced in the values in Table 1.1. These reflect differences in initial deposition and retention due to differences in surface roughness.

Table 1.1. Values for deposition to different surfaces relative to that on the grassed reference surface, for situations when dry deposition dominates. The term 'sd' denotes one standard deviation. All distributions are assumed to be normal. Values are given for elemental iodine gas and for particles with AMAD < 2 µm, 2-5 µm, 5-10 µm and 10-20 µm.

Surface	Elemental iodine	AMAD < 2 µm	AMAD 2-5 µm	AMAD 5-10 µm	AMAD 10-20 µm

	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd
Short grass	1.0	Ref. surf.	1.0	Ref. surf.	1.0	Ref. surf.	1.0	Ref. surf.	1.0	Ref. surf.
Bare soil	0.6	0.4	0.3	0.15	0.3	0.15	0.17	0.10	0.23	0.12
Soil and short grass	1.0	-	1.0	-	1.0	-	1.0	-	1.0	-
Small plants	0.8	0.5	1.4	0.7	1.6	0.8	1.0	0.5	1.2	0.7
Trees and shrubs	0.4	0.25	2.5	1.2	4.3	2.5	1.7	1.2	1.5	1.1
Paved area	0.2	0.1	0.25	0.15	0.75	0.35	0.3	0.15	0.3	0.25
Clay tile roof	1.5	0.3	0.8	0.1	3.0	0.8	1.9	0.5	1.5	0.4
Concrete tile roof	1.8	0.4	1.0	0.2	4.0	1.0	2.2	0.6	1.6	0.4
Fibre cement roof	1.6	0.3	0.9	0.1	3.6	0.9	2.1	0.5	1.6	0.4
Silicon covered fibre cement roof	1.0	0.2	0.7	0.1	2.5	0.6	1.7	0.4	1.4	0.4
Glass roof	0.5	0.1	0.4	0.1	1.4	0.4	1.5	0.4	1.3	0.3
Smooth metal roof	0.7	0.1	0.5	0.1	1.6	0.4	1.6	0.4	1.3	0.3
External walls	0.15	0.1	0.03	0.02	0.07	0.04	0.1	0.07	0.05	0.03

1.1.2 Indoor deposition parameter uncertainty (relevant in dry deposition conditions)

The time integrated outdoor air activity concentration of radioactive matter is given by

$$I_O = \int C_O(t)dt\,,$$

in which I_O is the time integrated outdoor air activity concentration (Bq s/m^3), and $C_0(t)$ is the outdoor air activity concentration at time t (Bq/m^3), and the integration is over the period while the plume persists.

The corresponding time integrated indoor air concentration at equilibrium is given by

$$I_I = \frac{I_O f \lambda_v}{\lambda_v + \lambda_d}$$

(Andersson et al., 2004), where I_i is the time integrated indoor air activity concentration (Bq s/m^3), f is the building filtration factor (the fraction of outdoor contaminant material in air entering the building canopy that actually penetrates into the building), λ_v is the air exchange rate (s^{-1}), and λ_d is the indoor deposition rate (s^{-1}). This assumes that the parameter values do not change during the period considered. All ventilation is assumed to be natural rather than forced, as ventilation systems etc. would be expected to be switched off, and windows closed, during a contamination episode.

The total deposition in a room is given by $I_{itot} = I_i \lambda_d V$, in which I_{itot} is the total indoor deposition in a room (Bq), and V is the volume of the room (m^3).

Deposition on the walls and ceiling of a room is generally considerably less than that on upward-facing horizontal surfaces such as the floor (to some extent depending on contaminant characteristics; Andersson et al., 2004). ERMIN assumes that material only deposits to the horizontal surfaces of area A, which is approximately the area of the floor. Then the average deposition per unit area is given by

$$D_i = \frac{I_i \lambda_d V}{A} \approx \frac{I_i \lambda_d AH}{A} = I_i \lambda_d H = \frac{I_O f H \lambda_d \lambda_v}{\lambda_d + \lambda_v}$$

(Andersson et al., 2004), where D_i is the average deposition indoors per unit area (Bq/m^2), A is the area of the horizontal surfaces in the room, approximately equal to the area of the floor (m^2), and H is the average height of the room.

The deposition on the outdoor reference surface is given by $D_O^r = I_O v_d^r$, where D_O^r is the deposition on the outdoor reference surface (Bq/m^2), and v_d^r is the deposition velocity to the reference surface (m/s).

The ratio of the initial deposition indoors to the initial deposition on the outdoor reference surfaces thus becomes

$$\frac{D_i}{D_O^r} = \frac{1}{v_d^r} \frac{Hf\lambda_d \lambda_v}{\lambda_d + \lambda_v}.$$

In this formula, H seems in dwellings to vary rather widely – typically between about 2.5 m and 4 m (in Denmark, for detached single family houses the minimum requirement is 2.3 m and for multistorey buildings it is 2.5 m; some old buildings probably have about 4 m to the ceiling; Building Regulations, 2018). The variation is assumed to be uniform between these boundaries.

v_d^r is (the deposition velocity to the outdoor reference surface) is assumed to be invariant in this context.

The filtration factor f is for particles smaller than 2 µm about 0.9 (with a sd of 0.1), for particles between 2 and 5 µm it is ca. 0.7 (sd = 0.2), for particles between 5 and 10 µm it is 0.4 (sd = 0.3), and for particles between 10 and 20 µm it is 0.1 (sd = 0.08) (Andersson et al., 2004; Byrne, 2009; Lange, 1995; Roed, 1987a). For elemental iodine, it should be assumed to be 1 (invariant) (Roed, 1987a).

Also, the rate coefficient of indoor deposition is rather strongly dependent on the physicochemical form (gas / particle diameter). For elemental iodine it is ca. 5 h^{-1} with a sd of 2 h^{-1}, for particles smaller than 2 µm it is 0.4 h^{-1} (sd = 0.2 h^{-1}), for particles between 2 and 5 µm it is 2.5 h^{-1} (sd = 1.5 h^{-1}), for particles between 5 and 10 µm it is ca. 8 h^{-1} (sd = 6 h^{-1}), and for particles between 10 and 20 µm it is ca. 30 h^{-1} (sd = 20 h^{-1}) (Andersson et al., 2004; Byrne, 2009; Lange, 1995; Roed, 1987a).

The rate coefficient of ventilation can for relatively modern dwellings of good tight construction be assumed to typically be 0.4 h^{-1}, with a sd of 0.2 h^{-1} (Andersson et al., 1995; Andersson, 2013). However, this can vary somewhat geographically, and appears in Northern Europe to be typically 0.5 +/- 0.3 (1SD) h^{-1}, for central Europe 1.0 +/- 0.4 (1SD) h^{-1}, and for southern Europe 1.8 +/- 1.0 (1SD) h^{-1} (Andersson, 2013).

Sheltering reduces indoor air concentrations by $f\,\lambda_v /(\lambda_v + \lambda_d)$ at equilibrium.

A series of experiments have shown that deposition velocity of particles in the range 0.5 µm to 5.5 µm was about 30 % higher in a room when it was 'normally' furnished than when it was unfurnished. There was no apparent trend in relation to particle size within this range (Lange, 1995). The effect of the degree of furnishing could perhaps be seen as an extra source of uncertainty on the indoor deposition velocity and on the indoor rate coefficient of deposition. In addition there are various phoretic effects etc. that we know can influence the indoor deposition, but which would depend on very specific things in the room such as heat sources, surfaces with high voltage (was more relevant in the old days before flat screen tv's) and surface moisture (Andersson et al., 2004).

1.1.3 Wet deposition parameter uncertainty

Table 1.2 shows estimates of the relative wet depositions to the different surfaces for each type of contaminant (again, the modelling of deposition to the shortcut grassed reference surface is in ARGOS and RODOS included in the atmospheric dispersion estimation tool). Also shown in this table is the fraction of the deposition to each surface which is practically instantaneously carried away, e.g., to sewers, with run-off water. Even during periods of strong rain, deposition to surfaces occurs through a combination of wet and dry deposition. However, unless the rain is extremely light or brief during such a phase or only leads to slight contaminant scavenging from the plume (not assumed for this deposition weather category), wet deposition will clearly be the dominant contamination process. Dry deposition contributions can thus be assumed to be negligible for the deposition weather category covered in this section. The initial run-off of contaminants in rainwater during the wet deposition process may depend on the surface roughness/permeability/porosity and rainfall intensity immediately before as well as during the wet deposition episode (Bonka & Horn, 1980; Karlberg, 1986; Sartor et al., 1974; Shaw et al., 2006). Further, the surface material type has been reported to be able to influence run-off through pH (Göbel et al., 2007).

On roofs compared with the grassed reference surface, the rain intensity incident per unit roof area will be less by cos(v), where v is the roof angle. It is assumed that common roofs have a slope of between 0 and 45 degrees. The initial retention after wet deposition of a range of Chernobyl contaminants (^{134}Cs, ^{137}Cs, ^{103}Ru, ^{106}Ru, ^{140}La and ^{140}Ba) with different physicochemical characteristics was recorded on different types of roof pavements with different slopes in Denmark following the Chernobyl accident (Roed, 1987). Caesium, which is in cationic form retained selectively and strongly in many building materials (Andersson, 2009), seems to be somewhat more efficiently retained on the roof than other contaminants. In general, the initial retention after the deposition process varies greatly with the roof material. For a range of materials and radionuclides, in the region of one-sixth to half of the contaminants were instantaneously removed with the run-off rain water. The exception from this was silicon-treated very smooth roofs with extremely low open porosity, where the run-off percentage was as high as 70-80 %. The main cause of variation here was by far the roof material and not the roof angle nor the radionuclide. Corresponding measurements made in Germany and the United Kingdom of wet-deposited Chernobyl radiocaesium on clay and concrete roofs showed similar values (Roed & Jacob, 1990; Sandalls & Gaudern, 1986). It should be noted that contaminant run-off in rainwater is likely to be more dominant when the roof pores are already filled by rain than when contaminated rain falls on a dry roof (Roed, 1987). Ritchie (1976) found that run-off from artificial surfaces in an urban area (e.g., roofs) would be virtually 100 % for all rainfall above an initially accumulated 3 mm, and if there has been rain within the previous hour the run-off will occur sooner.

Wet contamination levels on walls would in general be expected to be low, but associated with some variation according to factors such as the wind speed and direction during the contaminating process. In the Gävle area, which was wet-contaminated by the Chernobyl accident, a caesium contamination level on walls of slightly less than 1% relative to the reference surface was recorded in 1988 (Andersson, 1991). Figures reported by Roed & Jacob (1990) for the same location were by mistake somewhat higher (up to 3%), as the contamination estimate for the grassed reference surface originated from a direct measurement, not allowing for contaminant penetration.

Only a couple of weeks after the Chernobyl accident, the initial retention on street pavings of wet deposited contaminants was measured in Sweden (Karlberg, 1986; Karlberg, 1992). It was found that at this point, some 40-70 % of the radiocaesium incident on asphalt and differently textured concrete street pavings had been removed, most likely to a very high extent already during the deposition phase, with the run-off water. Somewhat less had been removed from rough concrete paving slabs. Similar figures were found for the more refractory ^{140}Ba and ^{110m}Ag that were according to Rulik et al. (1989) associated with particles with a size of several microns after Chernobyl, indicating that particle size within the range of interest has little influence on the fraction of contamination lost with run-off water. Also Jacob et al. (1990) reported results of measurements of wet deposition of Chernobyl caesium, on different urban pavings in Germany. After 32 days, 28-32% of the caesium remained on concrete pavings, and 36% in an asphalted parking lot. A measurement after 40 days in an asphalted square showed 32% retention. In a different area, the retention on concrete pavings after 160 days was found to be 33%. By extrapolation from the curves obtained for the different locations, it could be estimated that the initial retention was in the German region of 35-50%.

Experience with non-radioactive pollutants demonstrates that rain often leaves comparatively little contamination on vegetation (Gravenhorst & Höfken, 1982). The deposition before run-off for trees is interpreted as the deposition per unit ground area covered by the tree. Contaminants in the precipitation above the tree canopy will either be intercepted by the tree, lost by throughfall (falling directly through leaf gaps or dripping from leaves, needles, twigs and branches), or lost by stemflow (flow down stems or boles). It has been reported (Alexander & Cresser, 1994) that both for birch trees (Betula pubescens) and pine trees (Pinus sylvestris L.) the throughfall precipitation fraction is some 80% of the incident precipitation. This is an average figure for a two-year study in the English Midlands, in an area with an annual precipitation of 930 mm. The interception was greatest for the pine tree during summer. This is in agreement with findings of other workers of 80-90% throughfall and 2-5% stemflow (Carlyle-Moses, 2004; Kryshev, 1996; Neal et al., 1993; Pryor & Barthelmie, 2005). However, contaminants do not follow the water fractions evenly. Ronneau et al. (1987) reported that for Belgian spruce contaminated by a 7.4 mm rainfall episode after the Chernobyl accident significantly less ruthenium and lanthanum than caesium was intercepted. The explanation offered was biological absorption, e.g., by caesium exchange with potassium. It is also known that the rate of penetration of cations through the cuticle of vegetation is inversely related to the radius of the ion, and thus strongly favours caesium (Carini & Bengtsson, 2001). Similar figures have been reported by other workers for caesium on spruce, whereas deciduous trees have somewhat lower caesium interception (Schell et al., 1996). Schimmack et al. (1991) have reported a caesium interception fraction of 20% for beech trees. Deciduous trees would in winter conditions be expected to intercept considerably less than indicated by the numbers in Table 3. A rain interception fraction for a leafless pear tree has been reported, which was about half of that of an evergreen oak (Xiao et al., 2000). The same workers stress that interception fractions vary significantly dependent on factors like the structure of the tree and amount of rainfall.

Small plants would in general in the context interception be expected to be well represented by agricultural crops, due to sizes, shapes and textures. It has been reported that interception fractions will depend on the amount of rainfall, and plant type, as well as the stage of plant development (Müller & Pröhl, 1993). It would seem that a likely interception range relevant to urban small plants would be 10-30% for most radionuclides (IAEA, 1994; Schell et al., 1996). This would correspond to assuming a leaf area index value of about 5; retention coefficient of 0.2-0.3 mm, and rainfall of 4-10 mm (Müller & Pröhl, 1993). The leaf area index is the total one-sided leaf canopy area per projected area ground covered by the plant.

For relatively short urban grass, the leaf area index would be of the order of 1-3 (Kammann et al., 2005; Müller & Pröhl, 1993; Rodriguez et al., 1999), and the retention coefficient would be 0.2 for most radionuclides (Müller & Pröhl, 1993). With the same assumptions as for small plants regarding rainfall, this would give the retention/run-off expressed by the values in Table 1.2 (Müller & Pröhl, 1993).

It should be noted that initial run-off fractions with rain during the deposition can as shown in Table 1.2 vary considerably between different roof paving materials. The default parameterisation of the ERMIN model does not distinguish between material types, but within the current project, an effort has been made to refine the model in this context. Thereby, the overall uncertainty in the parameter governing the run-off fractions has been reduced greatly. For more information see section 1.2.1 in this appendix on the impact of weathering processes on roof pavings, as the two types processes were examined together for different roof paving materials.

Table 1.2. Values for initial deposition to different surfaces relative to that on the grassed reference surface, for situations when wet deposition dominates. The term 'sd' denotes one standard deviation. Also given are the fractions of the contaminants that immediately run off the surface with rain water during the deposition process.

Surface	Elemental iodine		Cationic caesium		Other contaminants		Elemental iodine		Cationic caesium		Other contaminants	
	Rel. deposition		Rel. deposition		Rel. deposition		Runoff fraction		Runoff fraction		Runoff fraction	
	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd
Short grass	1	-	1	-	1	-	0.9	0.1	0.8	0.1	1	0.2
Bare soil	1	-	1	-	1	-	0	-	0	-	0	-
Soil and short grass	1	Ref. surf.	1	Ref. surf.	1	Ref. surf.	0	-	0	-	0	-
Small plants	1	-	1	-	1	-	0.99	0.01	0.7	0.2	0.8	0.2
Trees and shrubs	1	-	1	-	1	-	0.99	0.01	0.5	0.3	0.8	0.2
Paved area	1	-	1	-	1	-	0.97	0.03	0.55	0.15	0.55	0.15
Clay tile roof	0.8	0.2	0.8	0.2	0.8	0.2	0.99	0.01	0.3	0.04	0.35	0.05
Concrete tile roof	0.8	0.2	0.8	0.2	0.8	0.2	0.99	0.01	0.4	0.05	0.45	0.06
Fibre cement roof	0.8	0.2	0.8	0.2	0.8	0.2	0.99	0.01	0.15	0.02	0.18	0.02
Silicon covered fibre cement roof	0.8	0.2	0.8	0.2	0.8	0.2	0.99	0.01	0.8	0.1	0.9	0.1
Glass roof	0.8	0.2	0.8	0.2	0.8	0.2	0.99	0.01	0.95	0.05	0.95	0.05
Smooth metal roof	0.8	0.2	0.8	0.2	0.8	0.2	0.99	0.01	0.9	0.07	0.9	0.07
External walls	0.01	0.01	0.01	0.01	0.01	0.01	0	-	0	-	0	-

1.1.4 Uncertainty in cases where dry and wet deposition are comparable

ERMIN also comprises a case, where contributions of wet and dry deposition are of approximately the same magnitude. Since precipitation is very effective in washing out contaminants from a plume, this case would be associated with very little rain, and comparatively rather little contamination would be removed with the run-off water during the deposition process. This is for instance clear from investigations in areas in Russia, which received some rain as the contaminated plume carrying primarily caesium condensation particles passed from Chernobyl. Here, dry deposition rarely contributed more than one or two percent to the total deposition on the reference surface (Andersson et al., 2002). It would thus in most cases only take very little precipitation during the plume passage to make wet deposition the dominant mechanism of contamination. Table 1.3 shows estimates of the relative depositions to the different surfaces for each type of contaminant assuming that half of the deposition is wet and the other dry (simple averaging with parameters described above). Also, the fractions of the deposit removed by run-off water during the contamination process are estimated on the basis of the same literature as used for the wet deposition mode, but assuming very little water. The rainfall rate is here assumed to be well below 1 mm per hour. Experimental and theoretical work has demonstrated that at low precipitation values (< ca. 0.5 mm), the majority of a contamination deposited in solution on a grassed area will remain on the grass (Bonka & Horn, 1980).

Table 1.3. Values for initial deposition of different contaminant groups to different surfaces relative to that on the grassed reference surface, for situations when wet and dry deposition are about equal. The term 'sd' denotes one standard deviation. Also given are the fractions of the contaminants that immediately run off the surface with rain water during the deposition process.

Surface	Elemental iodine		AMAD < 2 µm		AMAD 2-5 µm		AMAD 5-10 µm		AMAD 10-20 µm		Elemental iodine		Cationic caesium (< 2 µm fraction)		Other contaminants	
	Rel. deposition		Rel. deposition		Rel. deposition		Rel. deposition		Rel. deposition		Runoff fraction		Runoff fraction		Runoff fraction	
	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd
Short grass	1.0	Ref. surf.	1.0	-	1.0	-	1.0	-	1.0	-	0.3	0.1	0.2	0.1	0.2	0.2
Bare soil	0.8	0.2	0.7	0.2	0.7	0.2	0.6	0.2	0.6	0.2	0	-	0	-	0	-
Soil and short grass	1.0	-	1.0	Ref. surf.	1.0	Ref. surf.	1.0	Ref. surf.	1.0	Ref. surf.	0	-	0	-	0	-
Small plants	0.9	0.3	1.2	0.4	1.3	0.4	1.0	0.3	1.1	0.4	0.3	0.3	0.1	0.1	0.15	0.15
Trees and shrubs	0.7	0.2	1.8	1.0	2.5	1.0	1.4	0.7	1.2	0.6	0.3	0.3	0.05	0.05	0.15	0.15
Paved area	0.6	0.1	0.7	0.2	0.9	0.2	0.7	0.1	0.7	0.1	0.3	0.3	0.05	0.05	0.05	0.05
Clay tile roof	1.2	0.8	0.8	0.3	1.8	0.5	1.3	0.4	1.2	0.4	0.3	0.2	0.1	0.06	0.1	0.06
Concrete tile roof	1.5	0.9	0.9	0.3	2.4	0.6	1.5	0.4	1.2	0.3	0.3	0.2	0.2	0.1	0.2	0.1
Fibre cement roof	1.3	0.8	0.8	0.3	2.2	0.5	1.5	0.4	1.2	0.3	0.3	0.2	0.03	0.02	0.03	0.02
Silicon covered fibre cement roof	1.4	0.8	0.7	0.2	1.7	0.4	1.3	0.3	1.1	0.2	0.5	0.3	0.4	0.2	0.4	0.2
Glass roof	0.7	0.2	0.5	0.1	0.7	0.2	0.6	0.2	1.0	0.3	0.6	0.3	0.6	0.2	0.6	0.2
Smooth metal roof	0.9	0.2	0.6	0.2	1.2	0.3	1.2	0.3	1.1	0.3	0.6	0.3	0.55	0.2	0.55	0.2
External walls	0.07	0.05	0.02	0.015	0.04	0.03	0.06	0.04	0.03	0.02	0	-	0	-	0	-

It should be noted that due to the practical and temporal limitations of the project, the analyses that were carried out assumed that the contaminant was representative of the smallest of the aerosol groups described above, and readily soluble. This is in-line with the physicochemical forms recorded of airborne radiocaesium and radioiodine aerosols from the Fukushima accident, as well as long-range transported airborne radiocaesium, radioiodine and radioruthenium aerosols from the Chernobyl accident. Due to the special status that these radionuclides have held in these accidents with respect to radiological implications, they merit special attention in preparedness against the consequences of new future accidents.

1.2 Post deposition migration and weathering in the urban environment

Identifying appropriate time functions representing the natural weathering and migration processes of contaminants on each type of surface is essential in enabling estimation of time integrated doses and for instance residual doses received by the people after treatment in a prescribed way of a given type of surface in the urban/inhabited environment.

Material originally deposited in the typical relative proportions, as described in part 1.1 of this appendix, on the different types of surfaces in an inhabited environment will move away from the surface or migrate deeper into it over time due to natural weathering processes. It is the time constants determining the speed of these movements that are studied here with respect to uncertainty on the basis of a literature study. Prior to the Chernobyl accident, a very small number of systematic assessments were made of the retention of radioactive contaminants on different types of surfaces that specifically characterise the inhabited environment (see e.g., Qvenild & Tveten, 1983; Roed, 1985; Wilkins, 1987). However, a number of very early publications are particularly useful in interpreting the behaviour larger and insoluble contaminant particles on impermeable surfaces (see, e.g., Owen et al., 1960; Clark & Cobbin, 1964; Wiltshire & Owen, 1965; Sartor et al., 1974). After the Chernobyl accident time series studies were carried out in contaminated areas in

Germany (Jacob & Roed, 1990) and Sweden (Andersson, 1989, 1991, 2009, 2016; Andersson et al., 1995, 2002; Andersson & Roed, 2006; Brown et al., 2006), but long term studies spanning over more than 2 years were only carried out by the Technical University of Denmark around the Swedish town of Gävle where strong rain during the passage of the contaminated plume from Chernobyl led to comparatively high contamination levels facilitating measurements with good accuracy. The parameter values for migration on impermeable surfaces in the ERMIN model are in general mainly based on this information.

Unfortunately, measurements have not been made of the post-deposition migration of radioactive matter from the Fukushima accident on each of the various types of man-made surfaces representative of inhabited areas. Instead, the focus of the Japanese authorities has been on rapid carborne surveillance of dose rate in affected areas using KURAMA II detection systems (Kinase et al., 2015), which measure an uncollimated dose rate in a position close to the road surface. In relation to the actual average exposure of the local population, this measurement geometry would give an overrepresentation of the radiation from the nearby contaminated street surface. And since the natural decline in radioactivity on street surfaces has previously been found to be comparatively very rapid (Andersson, 2009), such repeated measurements would lead to overestimation of the rate at which the average dose rate declines in the area through natural processes. Qualitatively, however, these Japanese measurements illustrate that the decline in dose rate is as expected faster in urban areas, comprising surfaces with rapid natural weathering processes, than in rural areas, where the decline in dose rate level is largely dependent on the slow downward migration in soil.

Experience shows that the outdoor surfaces of primary importance in connection with long-term doses in inhabited areas are generally the (sometimes limited) surfaces of soil and the roof pavings, since the natural decline in contamination level on paved surfaces as streets is rapid, external walls seldom received much contamination due to the vertical orientation, and vegetation mostly only stays in leaf over relatively short time. It is therefore the soil vertical migration model and roof retention model that have been in focus in these uncertainty studies.

1.2.1 Uncertainty in weathering of contamination from roof pavings

The natural weathering of radiocaesium in cationic form from a range of roof paving types has been measured in the period following the Chernobyl accident, particularly in Denmark (Andersson, 2009) and Germany (Roed & Jacob, 1990). In these two places, the deposition generally took place in precipitation. The figure below shows the decline in roof contamination level due to the natural weathering process over the first ca. 15 years, as measured in Denmark. The measurements made in Germany (only for concrete and clay tile roofs), although so far giving very similar results to those in Denmark, were discontinued after only some four years. It is quite clear looking at the picture that the initial retention on these surfaces after the initial rain shower varies rather much. The two surfaces, for which there was least initial retention, were the silicon treated eternit surfaces. On these very smooth surfaces the early rain water (run-off) during deposition took with it nearly all the contamination, leaving only some 20 %. It can be seen that both for the silicon treated eternit and the corrugated eternit, the slope of the roof does not have much influence on the initial retention nor on the subsequent weathering speed. It is well-known (Andersson, 2009) that most clay tiles contain quite large amounts of the clay mineral illite (in intact crystal form in spite of the firing process, which generally occurs at comparatively low temperatures of < 1000°C). As in the soil, this mineral is known to have a highly selective capacity for effectively retaining cationic caesium at all thinkable concentrations following a major NPP accident. Also concrete and eternit contain minerals with similar selective ability to specifically fix the caesium cation (in concrete for example the mineral tobermorite can be mentioned). It is thus not surprising that the retention function after the earliest rainshower is very similar for these materials, as can be seen in the figure.

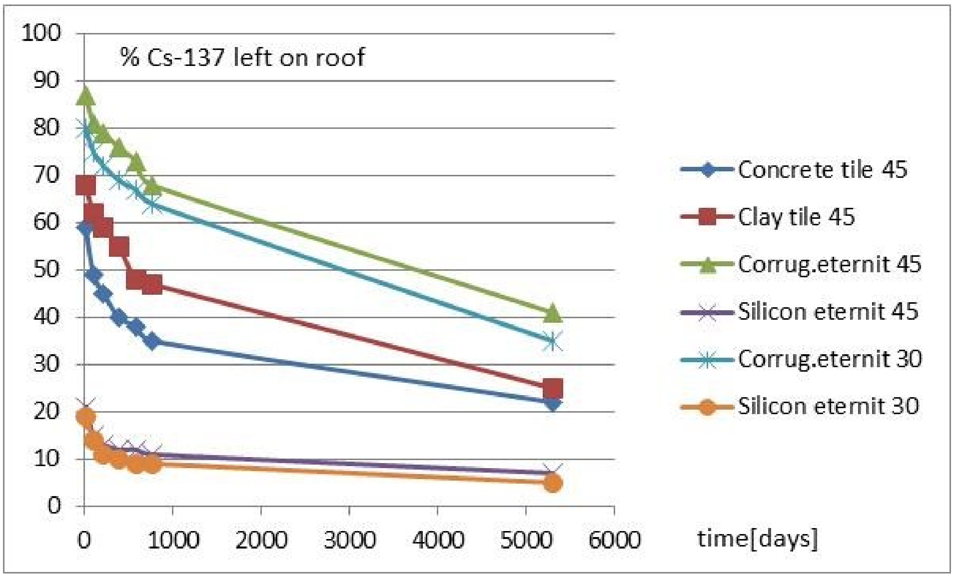

Fig. 1.1. Retained cationic caesium measured on different roof surfaces after the Chernobyl accident

The current weathering modelling in the ERMIN urban dose calculation tool does not take into account differences in roof material characteristics. It assumes that half of the material (Cs in cationic form) is weathered off with a half-life of 730 days and the other half with a half-life of 12800 days (based on the same dataset). This weathering function varies very little between materials and the average value can be assumed to be as previously modelled. However, according to this data, it should be assumed that the slow component half-life is associated with a standard deviation of some 20 %, whereas the long component has a standard deviation of ca. 12 %. However, the initial retention of cationic caesium is slightly more than 80 % on a corrugated eternit surface, ca. 68 % on a clay tile surface, ca. 60 % on a concrete tile surface, and only 20 % on a silicon-treated surface, which would inevitably cause significant differences in dose contributions as shown in Table 1.2 of this appendix. Generally, the initial retention increases with material roughness / open porosity.

According to experimentation made by Brown et al. (2015), only respectively 6 % and 15 % of a cationic caesium contamination was in a controlled natural weathering experiment found to remain on surfaces of glass and wood after a first light rain (which occurred after 7 days). After only 5 months, the levels had declined on these two types of surface to respectively 2 and 7 %. This corresponds to a weathering half-life of only respectively 95 and 135 days. Given the likely low relative significance of the doses from such surfaces, it seems to be reasonable to skip uncertainty evaluation here (partly because the data is really insufficient for that evaluation). Soluble strontium was found to weather off a clay roof with a half-life of only ca. 2.5 months (Brown et al., 2016), giving a similar indication of the retention of ionic contamination on roof surfaces with no specific trapping mechanisms for the ion in question (i.e. a value that can be used for all other ionic contaminants than caesium). In-line with this, Galkin (1993) found that cationic caesium deposits on smooth metal surfaces (steel, aluminium) were very easy to remove with water.

For fuel particles, very little data exists, and no differences are expected according to roofing material.

1.2.2 Uncertainty in vertical migration of contamination in soil

The vertical migration of contaminants in soil is described in ERMIN by a convection-dispersion model, as suggested by Bunzl et al (2000) and Kirchner et al. (2009). The crucial parameters are D_s and v_s, which are defined respectively as

$$D_s = \frac{D}{1 + K_d \dfrac{\rho}{\varepsilon}} \qquad (1)$$

and

$$v_s = \frac{v_w}{1 + K_d \dfrac{\rho}{\varepsilon}} \qquad (2)$$

where

D is the dispersion coefficient
v_w is the mean pore water velocity
K_d is the distribution coefficient of the contaminant in the soil
ρ is the bulk soil density
ε is the soil porosity

Bunzl et al. (2000) fitted the model to a set of measurements of Chernobyl ^{137}Cs in 100 soil samplings within an area of 100 m by 100 m pasture (distric cambisoil), assuming that the dispersion and convection parameters did not vary with time. They found that the observed small scale distribution in D_s could be adequately described by a log-normal frequency function, with a geometric mean of 0.6 cm^2 year^{-1} and a coefficient of variation of 0.77. The distribution of the parameter v_s might be approximated by a normal function, although this was not very convincing. This would imply a mean close to 0 cm year^{-1}, with a standard deviation of 0.2 cm year^{-1}. No convincing correlation was found between the two parameters. Values recorded by a few other workers including Schuller et al (1997) and of Szerbin et al (1999) were also used in the original ERMIN parameterisation of the model, suggesting overall average values of D_s and v_s, which were supposed to be applicable for all thinkable soil types and conditions. Because of the lack of available published data at the time on the variation across Europe in the soil parameters that govern the values of D_s and v_s, a uniform variation was assumed between the highest and lowest of the known values. However, it should be noted that the large majority of the data behind the values in ERMIN were derived from one small pasture area with essentially no soil type variation. This could not be expected to be representative of other soil types. Also, the values all related to cationic caesium contamination, and no effort was then made to derive values for other radionuclides through use of different Kd factors. The values from ERMIN are shown in Table 1.4.

Table 1.4 Default values assumed in ERMIN of the parameters in the soil model

Quantity	Value	unit	uncertainty
Parameter D_s	0.6	$cm^2\ year^{-1}$	a uniform distribution from 0.2 – 1
Parameter v_s	0.15	$cm\ year^{-1}$	uniform distribution from 0 – 0.3

Since then, the model type seems to have come into wider use, and particularly Bossew and Kirchner (2004), and Kirchner et al. (2009) have made thorough reviews of D_s and v_s by soil type on the basis of numerous assessments over different parts of Europe. For radiocaesium from Chernobyl, the resultant values were found to be as shown in Table 1.5. The migration of fuel particles in soils of different textures seems from field data to roughly match the migration of cationic caesium, although due to very different fixation mechanisms (Andersson, 2016). For other radionuclides, the migration while they are embedded in fuel particles can be assumed to be the same, but once the radionuclides are released from the fuel particles, they will migrate faster, as there are much weaker retention mechanisms (due to differences in Kd, see Table 1.6 below). The transition from fuel particles to ions in soil solution takes place according to the findings of Kashparov et al. (2004):

If the material was initially oxidised, the dissolution rate constant after deposition in soil will be:

$k\ (years^{-1}) = 0.6 * 10^{(-0.15*pH)}$ at $pH < 7.0$, and $k = 0.05$ at $pH > 7.0$ (3)

If the material was NOT initially oxidised, the dissolution rate constant after deposition in soil will be:

$k\ (years^{-1}) = 40 * 10^{(-0.45*pH)}$ at $pH < 6.5$, and $k = 0.05$ at $pH > 6.5$ (4)

The pH values to be used in Kasparov et al.'s formulae would be based on easily made actual measurements in case of an accident, but may be assumed to mostly be in the range 5.0-6.5 for mineral soils and 6.5-8.0 for organic soils.

Table 1.5 Results of a review of values of D_s and v_s by soil type in different types of soil, based on Chernobyl (cationic) caesium assessments.

Soil group	GM	GSD	AM	SD	Min	Max
Parameter: D_s (cm^2 per year)						
All soils	0.22	3.1	0.37	0.4	0.02	1.9
Clay/Loam	0.20	4.6	0.36	0.3	0.02	0.8
Sand	0.11	2.3	0.16	0.2	0.03	0.6
Organic	0.94	1.8	1.07	0.7	0.63	1.9
Unspecified	0.27	2.6	0.37	0.3	0.04	0.8
Parameter: v_s (cm per year)						
All soils	0.18	3.3	0.27	0.2	0.00	0.9
Clay/Loam	0.06	17.5	0.24	0.3	0.00	0.6
Sand	0.15	1.7	0.17	0.1	0.07	0.6
Organic	0.69	1.6	0.73	0.3	0.40	0.9
Unspecified	0.22	1.6	0.24	0.1	0.09	0.5

GM: geometric mean; GSD: geometric standard deviation; AM: arithmetic mean; SD: arithmetic standard deviation.

It may be noted that values for the groups 'all soils' and 'unspecified soil type' are generally in reasonable agreement, as they would be if the unspecified category in reality spans representatively over different soil types. Also, values reported for weapons fallout have been reported by Kirchner (2009), and these are in in most cases comparable with those for Chernobyl data. Differences would obviously reflect influences of contaminants that had not yet reached a soluble form at the time of assessment. Kirchner et al. (2009) seem to favour the assumption of a lognormal distribution with the above geometric mean and standard deviation. Minimum and maximum values should also be used as boundaries.

Values of D_s and v_s for other radioelements than caesium can be found by dividing the values in Table 1.5 for caesium by the 'retardation factor' (R = $1 + K_d \frac{\rho}{\varepsilon}$) relationship (i.e. the 'retardation factor' for the new element divided by that for caesium), applying appropriate values for all parameters (see above) for the soil type and element in question.

For the K_d values in equations 1 and 2, a wide range of data is available (also by soil type) from a relatively recent review by IAEA (2009). Table 1.6 shows the values for the 3 radioelements that would be thought to be of primary importance for external dose, but data for many more elements (Sr, U, Cd, Co, Ni, Zn, Ac, Ag, Am, As, Ba, Be, Bi, Br, Ca, Ce, Cl, Cm, Cr, Cu, Dy, Fe, Ga, H, Hf, Hg, Ho, In, Ir, K, La, Lu, Mg, Mn, Mo, Na, Nb, Np, P, Pa, Pb, Pd, Pm, Po, Pt, Pu, Ra, Rb, Rh, Sb, Sc, Se, Si, Sm, Sn, Ta, Tb, Tc, Te, Th, Tm, V, Y, and Zr) are readily available for use in the specified format in the report from IAEA, although generally based on much fewer data and often without soil type specific data or standard deviations. All values of K_d are assumed to be lognormally distributed based on the Central Limit Theorem, and the assumption of lognormal is generally supported by empirical evidence (Sheppard et al., 2009).

As for bulk soil density, this normally varies within a short range of 1.4-1.7 g/cm^3 for sandy soil, whereas it is typically 1.1-1.4 g/cm^3 for clay/loam soil (Brewer, 1964; Chesworth, 2008). Uniform distributions are reasonable over these rather small intervals. The relationship between soil bulk density and porosity is given by

$$\text{Soil Porosity} = 1 - (\text{Soil density} / \text{particle density}) \qquad (5)$$

(Blake & Hartge, 1986; Brady & Weil, 1996). In most soils the particle density can be assumed to be around 2.65 g/cm^3 (Brady & Weil, 1996). This is the density of quartz, which is the dominant mineral in most soils. For organic soils, soil porosity has on the basis of 180 soil samples been shown to have the dependence on soil organic C (SOC) shown in Fig 1.2 (Franzluebbers, 2011), whereas the bulk soil porosity depends on the soil organic C as shown in Fig. 1.3 (Hossain et al., 2015). These formulas might be included in ERMIN, so that the user can specify the values for organic soils directly from the soil organic C content, which is easily measurable by ignition (remembering the rule of thumb – the van Bemmelen factor - that organic matter contains 58 percent organic carbon (Périé & Ouimet, 2008)). However, according to Troeh & Thompson (2005), soils containing 12-18% SOC are generally those classified as organic soils. This corresponds to a bulk soil density of ca. 0.4-0.6 (perhaps assume 0.5) and a porosity of 0.69 (these values may be used for simplicity, as the carbon content can vary considerably in the same field from year to year, depending on e.g. soil fertilisation status).

Table 1.6 Results of a review of values of K_d for 3 important elements by soil type in different types of soil, based on hundreds (for Cs and I) of field assessments (in units of L/kg = cm^3/g).

Soil group	GM	GSD	AM	SD	Min	Max
Contaminant: Cs						
All soils	1.2E3	7	6.1E3	2.1E4	4.3	3.8E5
Clay/Loam	5.5E3	4	2.2E4	6.7E4	5.7E2	3.8E5
Sand	5.3E2	6	2.2E3	5.0E3	1.0E1	3.5E4
Organic	2.7E2	7	3.0E3	1.2E4	4.3	9.5E4
Unspecified	1.7E3	5	6.7E3	1.5E4	4.0E1	5.5E4
Contaminant: I						
All soils	5.4	6	2.5E1	7.0E1	1.0E-2	5.8E2
Clay/Loam	6.8	6	2.1E1	3.0E1	1.0	1.2E2
Sand	3.6	8	1.3E1	2.0E1	1.0E-2	1.3E2
Organic	3.6E1	4	9.3E1	1.8E2	8.5	5.8E2
Unspecified	2.6	6	2.0E1	7.oE1	1.0E-1	3.7E2
Contaminant: Ru						
All soils	2.7E2	8	4.7E3	1.7E4	5.0	6.6E4
Clay/Loam	5.0E2	2	6.0E2	3.6E2	2.0E2	9.9E2
Sand	3.6E1	6	7.7E1	9.0E1	5.0	6.6E4
Organic	-	-	6.6E4	-	-	-
Unspecified	1.4E2	3	2.3E2	2.1E2	3.4E1	4.9E2

GM: geometric mean; GSD: geometric standard deviation; AM: arithmetic mean; SD: arithmetic standard deviation.

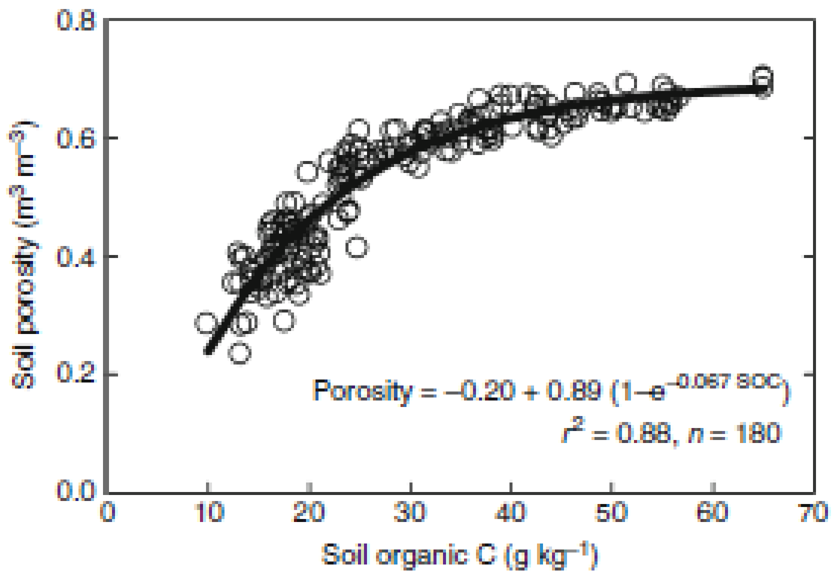

Figure 1.2. Soil porosity as a function of soil organic C. Experimental data and model fit as published by Franzluebbers (2011).

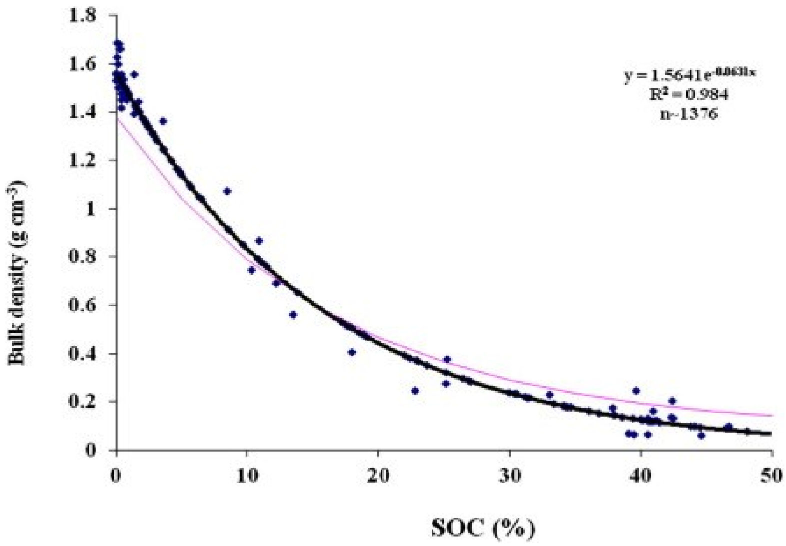

Figure 1.3. Bulk soil density as a function of soil organic C. Experimental data and model fit as published by Hossain et al. (2015).

It is clear that using this new soil migration data instead of the data so far used in ERMIN will make it possible to choose a relevant soil type on the basis of a simple and quick laboratory analysis of a local soil sample, and on that background estimate the future downward contaminant migration with much smaller uncertainty, than would have been the case, if the evaluation had not been based on any specification of soil type. Therefore, this work, which was carried out in the current project, has positive implications far beyond the analyses carried out within the project, when implemented in the standard ERMIN model.

1.2.3 Uncertainty in migration of contamination from trees

About half of the contamination on trees will according to literature (Roed, 1988) generally be removed with a short half-life of about 30 days (Nygren et al., 1994). An estimate of the standard deviation on this parameter would be about 15 days. Of the remaining 50 %, a small part of the order of 4 % (standard deviation of 4 %) will remain on the tree until it is felled (could in principle be hundreds of years), and the rest likely to have a half-life of some 1.7 years (Mamikhin & Klyashtorin, 2000), with an estimated standard deviation of about 1 year. For coniferous trees, the shedding of needles will occur with a half-life that depends on the exact species, but can be assumed to be of the order of 4 years, with a standard deviation of about 2 years (www.outdoorenvironmentsltd.com). For deciduous trees, it is assumed that they shed their leaves in the autumn. ERMIN models the shedding as a discrete event occurring at a certain time after deposition. As this would to some extent depend on the climate/weather and tree species, the time might be assumed to vary rather homogeneously within about 8 weeks of the autumn.

1.3 Occupancy

The key factor of importance in the public's occupancy of different positions in a contaminated inhabited (e.g., urban) environment is the fraction of time spent indoors, where they would to some extent be protected by the dwelling structures (e.g., walls) against any radiation coming from contamination in the outdoor environment. It is assumed that survey data obtained prior to the accident would still be reasonably applicable (certainly it would be likely to be the best available information in a transition phase), as areas where recovery is considered would not have very worryingly high levels of contamination that would necessitate a change in behaviour pattern, but of course the degree of social disruption and changes of habits would depend on how the situation is dealt with, including any public communication strategies. Naturally, this type of specific habit data is not available for each distinguished subset of the population (e.g., the people staying within a grid element of RODOS/ARGOS). This is recognised in ICRP publication 101 (2007a), defining the exposed individual in relation to practical optimisation. Here it is stated that if needed, such 'values may be derived from appropriate national or regional population data'.

A particularly useful data source in this context originates from the European EXPOLIS project (see, e.g., Jantunen et al. 1998; Rotko et al., 2000; Schweizer, 2004), where thousands of people in seven European cities (Athens, Basel, Grenoble, Helsinki, Milan, Oxford and Prague), were studied with respect to their time budgets, and the hours they spent in various microenvironments. Similar information has also been generated in various American studies (e.g., Adair & Spengler, 1989; Boudet & Zmirou, 1997; Echols et al., 1999; Ott, 1989), and recently in a small study in Australia (Physick et al., 2011). The purpose of the EXPOLIS study was to generate information that could be applied in environmental pollution exposure studies, since there was a general lack of such data for European countries. However also a single earlier American study (Ott, 1989) had reported results of time diaries for a number of European cities, also based on quite comprehensive datasets, though not as detailed with respect to characterisation of the involved population. Of particular interest in the present context is information on time fractions spent in the outdoor environment. For the time spent outside the indoor environment, distinction was made in both studies between the time spent in transit (walking, biking, bus, train, underground) and the time spent in more stationary outdoor positions. Table 1.7 summarises the *averaged* results of particular interest from these studies. It is worth noting that there is good correspondence between the two studies in the datasets for French and Czech cities, although the cities in these countries are actually not the same, and there is more than 10 years between the two studies. Also, there is correspondence between the fractions of time reported spent indoors at home in Osnabrück (Germany) and in a later, more comprehensive study of German homes (Brasche & Bischof, 2005). The results are also in reasonable agreement with results of newer, but smaller studies in Hertfordshire, UK, and Lille, France (Kornartit et al., 2010; Piechocki-Minguy et al., 2006).

Most of the time in transit was spent either walking or biking (45 %), or in cars (40 %), whereas less (14 %) of the transit time was spent in buses or trains, and very little (1 %) underground. There was only little variation in these figures between the different European cities. Obviously, people walking or biking are unshielded, and a modelling study (Lauridsen & Hedemann Jensen, 1982; Lauridsen & Hedemann Jensen, 1982a) has shown that the shielding provided by (empty) cars of different sizes typically only reduces the dose rate from a ^{137}Cs ground contamination by some 30 - 40 %. As these figures include the dose rate reduction from surface roughness compared to an infinite smooth source surface, the actual shielding effect of the car could well be considerably lower. Therefore, it seems reasonable as a (slightly) conservative measure to assume that nearly all time spent in transit is spent in unshielded positions. As can be seen, there is surprisingly little variation between the figures for different European cities in Table 1.7. The studies incorporate assessments made over different seasons of the year, and in significantly different European climates. The volunteers in the EXPOLIS study were all aged 19-60 years, whereas no information was given on the age distribution in the study reported by Ott (1989). A small Danish survey (Roed, 1990) reported a percentage of time spent indoors of 85 %, which seems in-line with the figures in the table. According to Table 1.7, the *average* (for all European cities) time spent indoors is 87 % (the figure should for modelling purposes be increased by one or two percent, accounting for stay in shielded positions while in transit).

Table 1.7. Averaged time fractions spent indoors, outdoors and in transit in various European cities. Figures from EXPOLIS (*) and Ott (1989) (#).

	Time fraction indoors	Time fraction outdoors§	Time fraction in transit
Helsinki (Finland)*	0.87	0.05	0.08
Athens (Greece)*	0.86	0.07	0.07
Basel (Switzerland)*	0.87	0.06	0.07
Grenoble (France)*	0.90	0.05	0.05
Milan (Italy)*	0.89	0.05	0.06
Prague (Czech Rep.)*	0.87	0.06	0.07
Oxford (UK)*	0.88	0.06	0.06
Kazanlik (Bulgaria)#	0.88	0.04	0.08
Osnabrück (Germany)#	0.86	0.05	0.09
Gyor (Hungary)#	0.85	0.07	0.08
Torun (Poland)#	0.91	0.02	0.07
Kragujevac (Serbia)#	0.89	0.03	0.08
Maribor (Slovenia)#	0.84	0.07	0.09
Olomouc (Czech Rep.)#	0.88	0.05	0.07
France (6 cities avg.)#	0.92	0.03	0.05
Average	0.87	0.05	0.07
Standard deviation	0.02	0.02	0.01

§ *Outdoor stay excluding transit*

It should be noted that all the applicable surveys were made for cities or towns above a certain size, and that rural populations might possibly spend more time outdoors. Also, for instance the proximity of residents and workplaces to resources, use of certain consumer products and appliance purchases could be important factors affecting the local behaviour (Loftness et al., 2007), and are not highlighted in these studies. Further, life style and life stage may vary, which can affect how people spend their time (Altergott & McCreedy, 1993). EXPOLIS data are represented with respect to age, marital status, working status, education, number of children, home environment, workplace building type, etc. Thus, both area specific and human-related parameters may be at play, but according to the EXPOLIS dataset, most of these influences are insignificant. There is considerable difference between time fractions spent outdoors during weekends and on workdays (Schweizer, 2004; Borrego et al., 2009), but this is of no interest in the present context, as longer dose integration time periods will even out any weekend effects. Differences between times spent outdoor in winter and summer (such data are available for the EXPOLIS study) are generally relatively limited (less than 50 % in the different studied cities), and not very interesting for long time dose integration. However, Kornartit et al. (2010) reported greater differences between times spent outdoors during summer and winter in Hertfordshire, UK (about 87 % difference). The European studies are limited to investigations of adults, but an American study reports on a survey of children (Geyh et al., 2000). Here 184 children between 6 and 12 years of age participated. On average, these spent 83 % of their time in locations that would be categorised as 'indoor' with respect to shielding. When comparing to other studies in similar American areas, it does not appear that children on average spend a lot more time outdoors than do adults (perhaps 20 % more).

Within each dataset for a given location, there is a considerable variation, notably since the number of hours that a person works outdoors varies considerably. This should be represented in the values used for decision support. In relation to deterministic calculations of dose to the representative person, the ICRP's (2007a) recommendation for the use of habit data is to use an 'average value for the more highly exposed group or 95th percentile of appropriate national or regional data'. Sufficiently detailed regional datasets to derive 95th percentiles are only available for the EXPOLIS study. Since increasing the amount of time spent *outdoors* will in all cases increase the external dose, 95th percentiles of the time fraction spent *outdoors* should be applied. Table 1.8 shows the relevant values extracted from EXPOLIS data for time spent outdoors and indoors (2 % have been added to the indoor budgets to account for the transit time spent in positions that are categorised as 'indoor' with respect to shielding; the rest of the transit time is here accounted for as 'outdoor'). As can be seen, also the 95th percentile varies only little between cities in different countries, and there does not seem to be a climatic correlation. However, the time fraction spent outdoors is here about twice as great as the average value.

To comply with ICRP recommendations regarding the definition of representative persons, it seems to be the values in Table 1.8 that should be recommended for deterministic dose calculations such as those performed in ARGOS and RODOS (due to the little variation between cities, standard values of 0.75 and 0.25 can be applied for respectively indoor and outdoor time fractions). A normal distribution should of course be assumed, and the standard deviations for the parameter 'time spent indoors' in each of the cities are also reported in Table 5 (Schweizer, 2004). Minimum and maximum would naturally be respectively 0 and 1.

Table 1.8. 95th percentile time fractions spent outdoors (and corresponding time fractions indoors) in various European cities. Figures from EXPOLIS.

	Time fraction indoors	Standard deviation	Time fraction outdoors
Helsinki	0.73	0.08	0.27
Athens	0.74	0.08	0.26
Basel	0.73	0.09	0.27
Grenoble	0.79	0.06	0.21
Milan	0.77	0.07	0.23
Prague	0.73	0.09	0.27
Oxford	0.75	0.07	0.25
Average	0.75	0.08	0.25

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D 9.20 Appendix 2.

ERMIN uncertainty

Final

Version 1.0

CONFIDENCE-WP4. Transition to long-term recovery, involving stakeholders in decision-making processes

Document Number: CONFIDENCE-WP4/T4.1.1-R02

Thomas Charnock. (PHE)

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2 Appendix ERMIN uncertainty

ERMIN is a tool in the RODOS (Ievdin et al, 2010) and ARGOS (PDC, 2018) decision support systems for analysis of recovery within inhabited areas. It models the long term contamination on urban surfaces and predicts public doses in different inhabited environments with and without different combinations of clean-up options. The purpose of this appendix is to evaluate uncertainty in the model in order to guide the appropriate use of ERMIN and to provide qualitative or quantitative statements about the uncertainty of the outputs.

ERMIN is a complicated model with many inputs and outputs (see Appendix 2.1), in order to keep this analysis constrained it is necessary to restrict the scenarios to which it is applied and to restrict the outputs considered. Appendix 2.2 describes the scenario restrictions in this analysis; Appendix 2.3 discusses the outputs considered. Appendix 2.4 identifies and categories the main sources of uncertainty that apply to ERMIN, particularly with regard to these scenarios and endpoints. Appendix 2.5 presents a preliminary sensitivity and uncertainty analysis using the information on parameter uncertainty compiled in Appendix 1. Finally, Appendix 2.6 summarises the conclusions.

2.1 ERMIN model description

The model input includes an estimate of the initial deposition of a set of radionuclides to a reference surface, which in most cases is lawn away from buildings. The deposition input may be obtained from an atmospheric dispersion and deposition model but by the transition phase, it can be expected to be derived from field monitoring and ideally from aerial monitoring that will provide the extensive coverage needed. In addition, the user selects one or more urban environments from the database to represent the urban area of interest. Finally, the user is able to select and combine management options from the ERMIN database and specify where and when they are applied in the area of interest.

The model uses a dataset of ratios derived empirically to make assumptions about the amount of initial deposition to other urban surfaces including, paved, trees, walls, roofs and interiors. Empirical weathering functions and parameters are applied to simulate the long term retention on the different urban surfaces and in the soil column.

ERMIN has a dataset of dose rates (UDL, united dose rate library) containing hundreds of factors that give gamma dose rate from a unit deposition of 1 Bq m^{-2} on 1 m^2 of each urban surface (e.g. roofs, walls, trees etc.) of each radionuclide to different locations (e.g. indoor ground floor, indoor 1st floor, cellar, outdoor behind building etc.) within different environments (e.g. terrace houses, semi - detached houses, multi-storey apartment building etc.). The UDL was compiled from previous in-depth studies that used Monte Carlo particle transport models to calculate dose rate at various locations within complex configurations of materials. It is used to estimate the long term doses in various locations indoors and outdoors.

Clean-up options are represented by modifying the surface activities, weathering rates and shielding properties with a dataset of empirical derived parameters.

2.2 Scenario considered

ERMIN was designed for reactor accidents but it could potentially be used in other situations such as transport accidents. In these situations, the uncertainties may be greater and the relative importance of sources of uncertainty and of endpoints will be different. However, this study primarily looks at the uncertainty when ERMIN is applied to reactor accidents. Furthermore, it is focuses on long term external doses that are assumed to be dominated by contributions from a deposition of ^{137}Cs in cationic form, as was the case with the Fukushima accident as well as in the more distant areas contaminated by the Chernobyl accident.

By the time the emergency phase has ended and the recovery phase has begun some uncertainties such as the initial deposition will have been reduced through field measurements and aerial surveys; a reduction in epistemological uncertainty. However, it is likely that consideration of recovery issues will begin well before the emergency phase is over and when uncertainty on initial deposition is high. At this time the focus is not on developing a strategy but on developing a situational awareness about potential magnitude of the duration and extent of disruption (e.g. relocation, restricted access and clean-up operations) in order to manage public expectations and begin to marshal the recovery response. In addition, there are early countermeasures that should be implemented without delay if they are deemed suitable at all, and some limited modelling may be required for these, particularly to get an overview of resource requirements. For example, if the deposition is dry the lawn mowing can be highly advantageous in reducing more permanent and problematic soil contamination if carried in a timely fashion.

At a later stage, the emphasis is on the evaluation of recovery measures for optimisation of intervention (including involvement of stakeholders). However, at this stage one can expect that measurements of surface contamination have been made reducing some of the epistemological uncertainty. This investigation focusses on the later phase starting at transition which is the term for the start of the recovery phase when field measurements of contamination are becoming available and when the priority is developing and implementing strategies to promote recovery.

2.3 Endpoints considered

ERMIN is a complicated model and produces a large number of different endpoints, including; predictions of surface contamination, concentration of resuspended radioactivity in air, surface dose rates, public doses, worker doses, waste amount , waste activity, and cost of clean-up options. The user is likely to focus on different endpoints at different stages of the recovery. The sources of uncertainty are likely to impact to a greater or lesser extent on different endpoints and therefore, to constrain the size of this investigation only the projected average normal living effective dose as a function of time from exposure to external radiation from deposited radioactivity, to the population or subset of the population living in a contaminated zone. This is an important endpoint because it can be directly compared with reference levels to estimate the “area affected” and the “duration of disruption” and it will be crucial in both early phase and at the transition and beyond.

“Duration of disruption” is an endpoint closely associated with the projected average normal living dose. However, it is subject to an additional large source of uncertainty on the criteria set by the decision makers and stakeholders. Therefore, this endpoint will be discussed along with the project average normal living dose but not examined quantitatively here.

The normal living dose that ERMIN calculates is not to an individual but represents an average dose to a population in a built up area. Within that population, there will be variations that arise from variability in the deposition, the inhabited environment (shielding, surface materials, weathering) and where the individuals spend time. In addition, there is physiological variability between individuals. Some of this variability is represented within the model; for example different urban and semi-urban environments have different shielding properties, but most of the variability is not represented and for most processes ERMIN uses average or representative values, for example;

average deposition to a reference surface, average weathering rates, average occupancy and standard adult physiological parameters (e.g. breathing rate).

In addition, the investigation looks at the projected average outdoor effective dose as this is less affected by the uncertainties in the environment configuration and in where people spend their time. Both the normal-living and outdoor doses can be further subdivided by urban surface to identify the surfaces that contribute most to the total dose and are thus suitable targets for clean-up.

The investigation focuses on external gamma doses from radioactivity deposited on urban surfaces. While ERMIN predicts beta doses in skin and effective doses from internal exposure to inhaled resuspended radioactivity, for reactor accident scenarios these are usually less important pathways and within ERMIN are subject to considerable additional uncertainties. In the case of beta doses, the most significant is judged to arise from stochastic uncertainty particularly in the unit doses rates from beta emitting radionuclides on different surfaces in different inhabited environments. For resuspension, the most significant probably arise from model uncertainty. The special cases of deposition during or onto snow are also not considered.

2.4 Sources of ERMIN uncertainties

As a first step, sources of uncertainty for the various components of ERMIN were identified and assigned to the various uncertainty categories:

- Stochastic uncertainty that relates to physical randomness.
- Judgmental uncertainties involved in the choice of parameters
- Epistemological uncertainty that relates to lack of knowledge.
- Computational uncertainty that relates to the computational choices when translating a model to computer code and running that model on specific hardware.
- Model uncertainty that relates to simplification of a model from the real world.
- Ambiguity, lack of clarity and endpoint uncertainty.
- Social and ethical uncertainty, uncertainty relating to value judgments.

With each source of uncertainty, a qualitative assessment of the magnitude of the impact of the uncertainty on the model endpoints was made. This assessment was made by the model developers based on experience of developing the model and the supporting libraries, and also running the model.

Table 2.1 to Table 2.5 list the sources of uncertainty identified for each category. Judgemental and stochastic uncertainties have been grouped together because they are often difficult to draw out, and social and ethical uncertainties have been omitted as beyond the scope of this investigation

Table 2.1 Sources of stochastic and judgmental uncertainty identified in ERMIN

Model component	Sources of uncertainty	Impact on residual dose uncertainty
Initial deposition to the reference surface from ADM – the input includes wet and total deposition of each radionuclide onto this surface for a number of periods for each ADM grid square.	Variability in total amount, dry/wet ratio, radionuclide composition, missing radionuclides, small scale patchiness of deposition not captured	Large: there is a linear relationship between total deposition and project dose. Small impact on relative surface contribution.
Initial deposition to the reference surface from measurements /aerial survey – the user delineates areas and specifies the total deposition of each radionuclide plus a deposition category (dry, wet, equal dray and wet, in snow or onto snow).	Errors in total amounts and radionuclide composition/missing radionuclide, Errors in interpolation between measurements Actual deposition conditions differ from broad deposition categories the user can specify.	Medium – it is expected that by the transition stage and beyond the definition of the radiological situation is becoming well established

Model component	Sources of uncertainty	Impact on residual dose uncertainty
		particularly if aerial surveying is available. Small impact on relative surface contribution.
Deposition scenario – regardless of the source of initial reference surface deposition the user must select a scenario from a choice of four this controls the portioning of the radioactivity in one of four particle groups, which in turn have different surface deposition ratios and surface retention parameter sets.	Actual partitioning different from the broad categories the user can select from. The particle groups do not fully capture the range of possible deposition and retention properties of the deposited radionuclides.	Small
Initial redistribution onto other urban surfaces – initial deposition onto urban surfaces is estimated by applying weather and particle dependent empirical ratios to the input deposition on a reference surface.	The broad surfaces categories in ERMIN actually comprise different materials, orientations, ages and conditions. Depositing particles can have different physicochemical properties that are not fully captured by the broad particle groups in ERMIN. Weather conditions at deposition may not be fully captured by the broad categories in PACE when initial deposition is from measurements.	Medium
Weathering/retention processes - radionuclides are weathered using particle group specific parameters.	The broad surfaces categories in ERMIN actually comprise different materials, orientations, ages and conditions. Depositing particles can have different physicochemical properties that are not fully captured by the broad particle groups in ERMIN. Long term weather conditions vary year to year (dry or wet years)	Small, assuming correct scenario selected (see deposition scenario)
Soil migration parameters - migration model requires two parameters, current default parameters fitted to ^{137}Cs Chernobyl observations and are applied to all particle group.	Does not account for different soil types, layers, particle groups, long term rainfall rates etc	Small in the short term (as migration is slow), potentially large in the long term.
Occupancy –a figure that represents average time indoors and outdoors.	Does not capture variability in target population. Does not capture time in different indoor environments and locations.	Medium for residual dose Potentially high for relative surface contribution when comparing interior surfaces to outdoor surfaces.
Urban environment – a number of idealised environments are represented in the ERMIN database as different proportion of broad urban surfaces, and unit doses rates to indoor and outdoor locations from radionuclides deposited on those surfaces.	Variability of real inhabited environments; construction materials, shielding, proportions of surface, spacing of buildings, locations of windows, proportions of paved, grass, soil, trees. Some of the gross variability is captured by the having different idealised environments.	Large, particularly in environments that are very different from the idealised and contain surfaces not included in ERMIN.
Clean-up option effectiveness parameters – options work in different ways and their effectiveness is described by different sets of parameters in the database	Surface variations, physicochemical properties, variation in weather prior to application, variation in application (timing, materials, equipment, personnel etc)	Small-medium (and only when clean-up options applied)
Tree processes – leaf emergence and fall described by various simple parameters	Tree species, changes in growing conditions year to year	Small-medium. Potentially large when considering the relative contribution from trees.

Table 2.2 Sources of epistemological uncertainty identified in ERMIN

Model component	Sources of uncertainty	Impact on residual dose uncertainty

Model component	Sources of uncertainty	Impact on residual dose uncertainty
User selected urban environment (and associate configuration) – the user selects one or more environments from the database based on their judgement about which ones best represented the real environment. The user can refine the selection by selecting a particular configuration for some of the environments.	Lack of knowledge about the real environment being assessed.	Small – it is likely the user will have sufficient knowledge to pick the most appropriate environment and break the area down into different zones to represent different environments. Facilities such as aerial imagery are readily available for most parts of the world that can assist this. (See also the stochastic uncertainty associated with environments in Table).
User selected deposition scenario – depending on a user selected scenario, the radionuclides are apportioned between four particle groups that have associated parameters sets for initial distribution, surface weathering and some countermeasures	Lack of knowledge about particle properties of the deposition or the conditions under which the particles were produced.	Small
User specified countermeasure strategy – the user specifies the a countermeasure in terms of different options applied at different locations at different terms	There may be a difference between the way the user assumes options will be applied and how they can actual be applied, in particular users may be over optimistic in how quickly resources can be marshalled and deployed	Small
Leaves on trees – in the simplistic ERMIN model for deciduous trees, the tree will either have leaves or not. If they have leaves then there will be a specified time before leaf fall.	The user may be unsure whether there are leaves on trees.	Medium

Table 2.3 Sources of computational uncertainty identified in ERMIN

Model component	Sources of uncertainty	Impact on residual dose uncertainty
ERMIN GRID size	Resampling from ADM grid to ERMIN grid Representation of environment zones, clean-up zones, deposition zones on a grid.	Small
Numerical integration	ERMIN numerical integrates the soil contamination to get integrated contamination by depth and time.	Small
Temporal steps	Internally there are some approximations that can cause small differences in outputs if different sets of output times are chosen. In calculation of the "duration of disruption", linear interpolation is used in the estimation between user specified times is used to estimate exact time that the dose falls below a specified level.	Small

Table 2.4 Sources of model uncertainty identified in ERMIN

Model component	Sources of uncertainty	Impact on residual dose uncertainty
Urban surfaces – ERMIN contains a set of surfaces chosen to represent adequately initial deposition, weathering processes and surfaces to which clean-up options might be applied in different inhabited environments, whilst not over burdening the model or the user.	Representation of differences in deposition with one surface (e.g. from one side of building to another) not possible. Missing surfaces (e.g. vehicle surfaces, glass). Differences in weathering rates because of differences in materials (e.g. roofing material, brick or wood walls) or differences in surface position (upper and lower parts of walls). Differences in grass length at time of deposition.	Medium – potential in large in areas with glass buildings
Weathering – radionuclides are weathered from urban surfaces generally by applying a single or double exponential	Represents a non-continuous process which is expected to be driven by individual rainfall events as continuous. Also see urban surfaces above.	Small, particularly over longer time scales. Potentially more significant if extreme weather occurs immediately after deposition; e,g, extreme rainfall.
Soil migration - ERMIN uses a convective/ dispersive	Deposition onto top of soil surface ignores	Small

model to represent downward migration through the soil model.	penetration into cracks in wet conditions. Migration ceases after CM applied to soil.	
Tree processes	There are two kinds of trees; deciduous and coniferous. Both are treated simply, for example there is no deposition on trees with no leaves, deciduous leaf fall is assumed to occur instantaneously, coniferous needles are assumed to drop linearly over a period.	Small but may be significant in environments with little or no grass/soil
Radioactive decay and ingrowth	ERMIN models the ingrowth of one daughter. Maybe inadequate if radionuclide has more than one significant daughter or is part of an important decay chain.	Small – for most of the radionuclides expected to be significant in a reactor accident this is not the case.

Table 2.5 Ambiguity, lack of clarity and endpoint uncertainty identified in ERMIN

Model component	Sources of uncertainty	Impact on residual dose uncertainty
Dose endpoints	e.g. It may not be clear to the user how the weighted averaging between several indoor and outdoor locations in different environments is undertaken	Small
Surfaces	The meaning of “other paved” surfaces. How the very broad category of “Interior surfaces” relates to wide range of possible interior surfaces (floors, ceiling, walls, carpets, furniture etc)	Small

2.5 ERMIN sensitivity analysis

This section further explores some of the sources of stochastic and judgmental uncertainty identified in Table 2.1 by means of a limited sensitivity analysis (SA) and uncertainty analysis (UA). Initial deposition has been identified as one of the most important source of parameter uncertainty for public average doses. However, it is omitted from the SA because it is well understood that, all other conditions remaining the same, the output doses predicted by ERMIN change linearly with the total level of initial deposition. Furthermore, the total level of initial deposition has no effect on the relative contribution of different surfaces.

ERMIN calculates doses and dose-rates using factors in the unit dose-rate library (UDL) that give the dose-rate from a unit deposition of 1 Bq m^{-2} on 1 m^2 of each urban surface (e.g. roofs, walls, trees etc.) of each radionuclide to different locations (e.g. indoor ground floor, indoor 1st floor, cellar, outdoor behind building etc.) within different environments (e.g. terrace houses, semi-detached houses, multi-storey apartment building etc.). The UDL was compiled from previous studies that used Monte Carlo particle transport models to calculate dose rate at various locations within complex configurations of materials. As identified in Table 2.1 this component is subject to considerable stochastic uncertainty from the infinite variability in possible configurations and materials. The sensitivity of these factors could not be investigated within the practical constraints of this project and was omitted from the SA. However, it should be recognised that the use of different idealised environments represents an attempt to capture some of the gross variability (see Figure 2.1). Therefore, in order to enhance the value of the SA, it was repeated for several of the ERMIN environments.

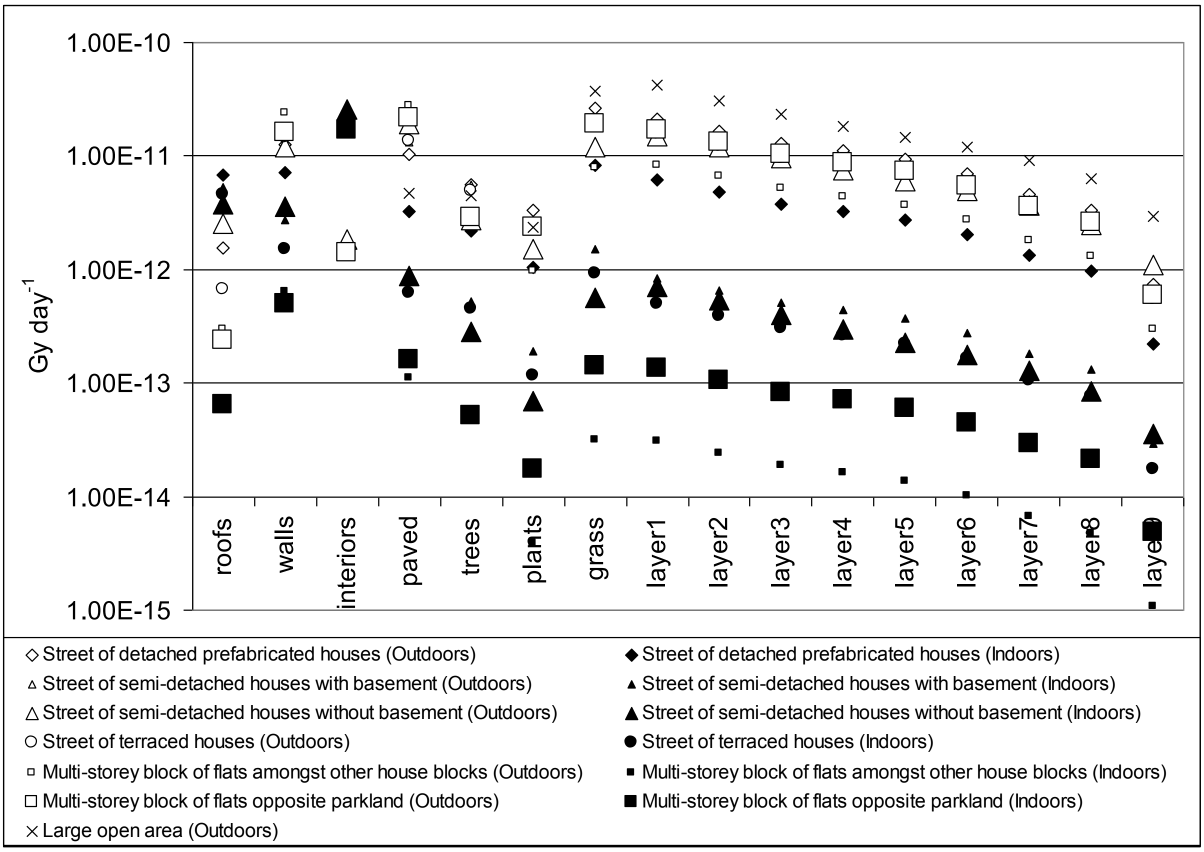

Figure 2.1 graph of the ^{137}Cs unit dose rates for indoors and outdoors in all the ERMIN environments. Unit doses rates from the UDL (Gy day^{-1} per Bq/m^2 per m^2 of surface) have been normalised by the area of the surface within the default configuration of each environment. Where an environment has more than one indoor or outdoor location the dose rate has been averaged.

The sources of stochastic and judgmental uncertainty included in the SA are:

- Occupancies
- Redistribution of initial deposition on to urban sources.
- Weathering
- Soil migration

The sensitivity analysis looked at a constant initial deposit of 10^6 Bq m^{-2} ^{137}Cs to the reference surface (lawn). This value gives an annual external dose in the first year of a few millisieverts. The default ERMIN deposition scenario ‘1’ was used; for Caesium this means that the whole deposition is considered by ERMIN to consist of soluble aerosols in cationic form. The analysis looked at both wet and dry deposition. The analysis is repeated where appropriate for different ERMIN environments including open area (infinite lawn), semi-detached houses, multi-storey blocks, terraced houses and prefabricated houses. The analysis assumed deposition in spring and the leaves will remain on deciduous trees for 158 days.

2.5.1 Base line runs

The principal outputs considered for the sensitivity analysis are the annual average normal living effective dose, the annual average residual outdoor effective dose and the relative contributions to those doses from different surfaces. Figure 2.2 gives plots of the predicted annual dose for different environments and broad occupancy assumptions (indoors, outdoors and normal living) and for both wet and dry deposition. As expected the high doses can be seen in the open area and the lowest doses are in the highly shielded multi-storey apartment block environment. Also as expected the highest doses are for the outdoor locations and the lowest for the indoor in each environment. Generally, wet deposition would be expected to give a higher dose because of enhanced deposition, however in this analysis the initial deposition is fixed at 10^6 Bq m^{-2} ^{137}Cs for both wet and dry deposition, and so it is not surprising that wet deposition gives lower doses in all environments except the open area where predicted doses are the same.

Figure 2.3 and Figure 2.4 illustrate the predicted contribution of different urban surfaces to annual dose under dry and wet deposition respectively. The biggest difference between the two figures is that doses from indoor surfaces contribute relatively much less under wet deposition than dry. Under dry deposition in the high shielded multi-storey environment (Figure 2.3b), interior surfaces are the dominant contributor to dose, whereas in the low shielded prefabricated building environment (Figure 2.3c) outdoor surfaces dominate.

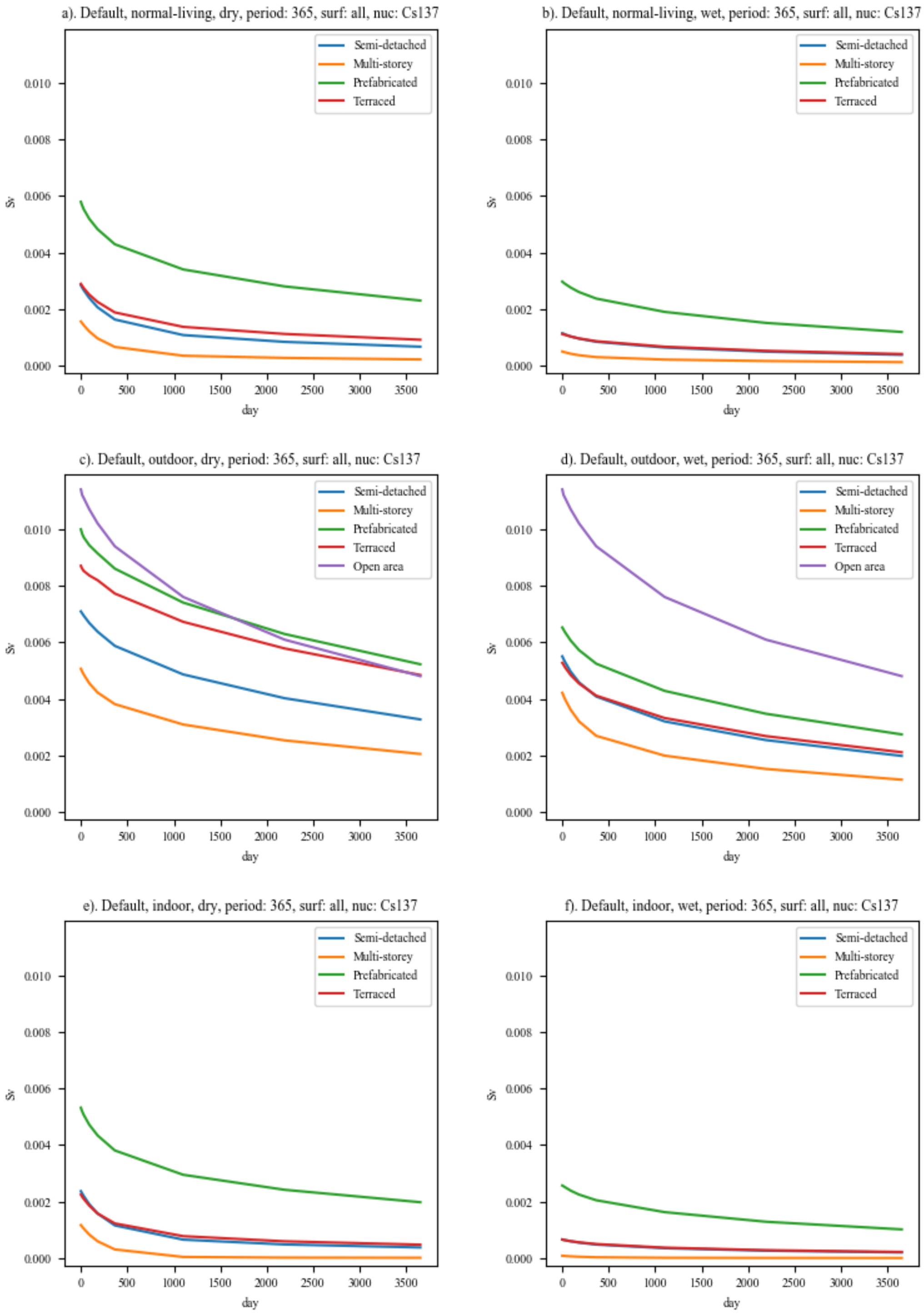

Figure 2.2 Predicted annual doses for different ERMIN environments, using default ERMIN parameters, with different occupancy assumptions (indoors, outdoors and normal living) for both dry and wet deposition.

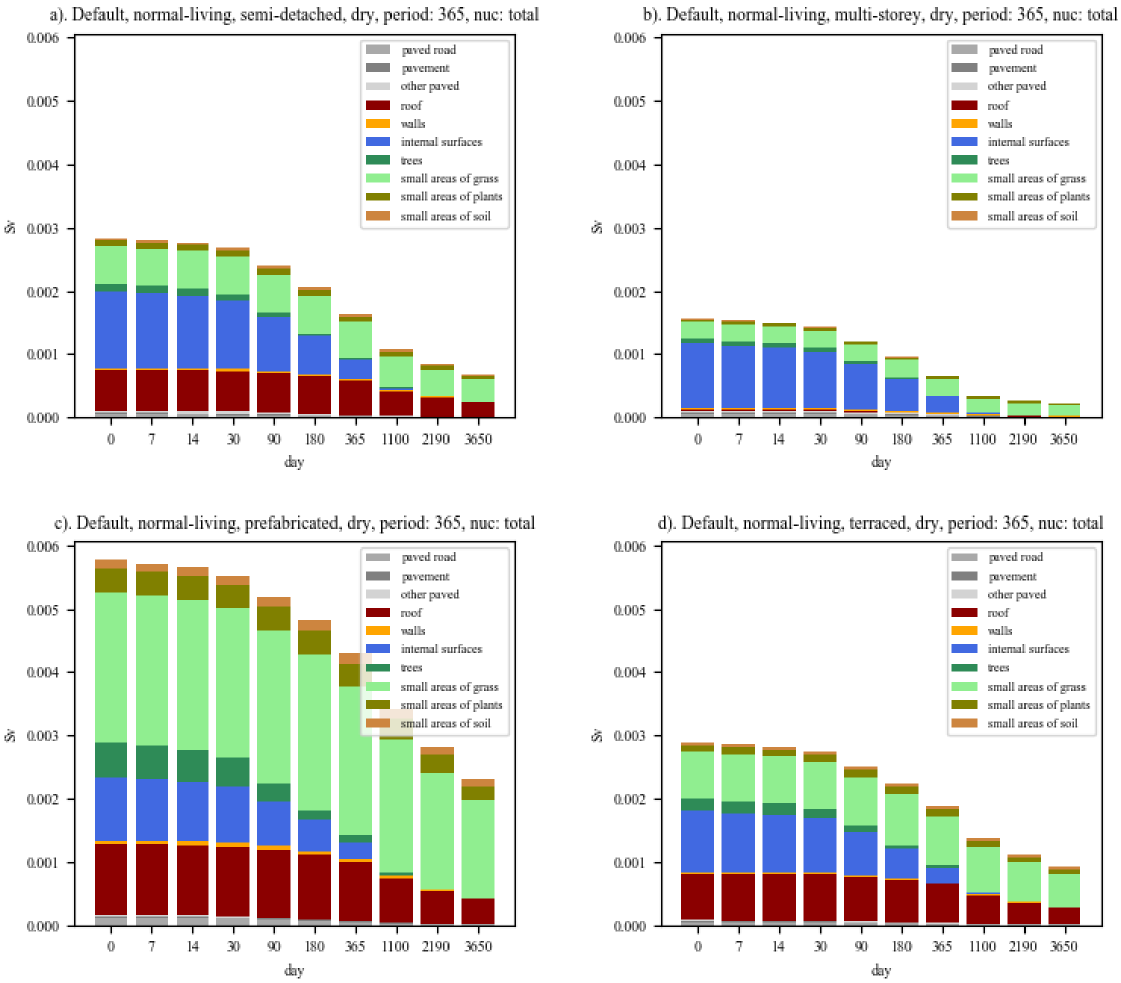

Figure 2.3. Predicted contribution to annual dose from different surfaces in four ERMIN environments assuming dry deposition.

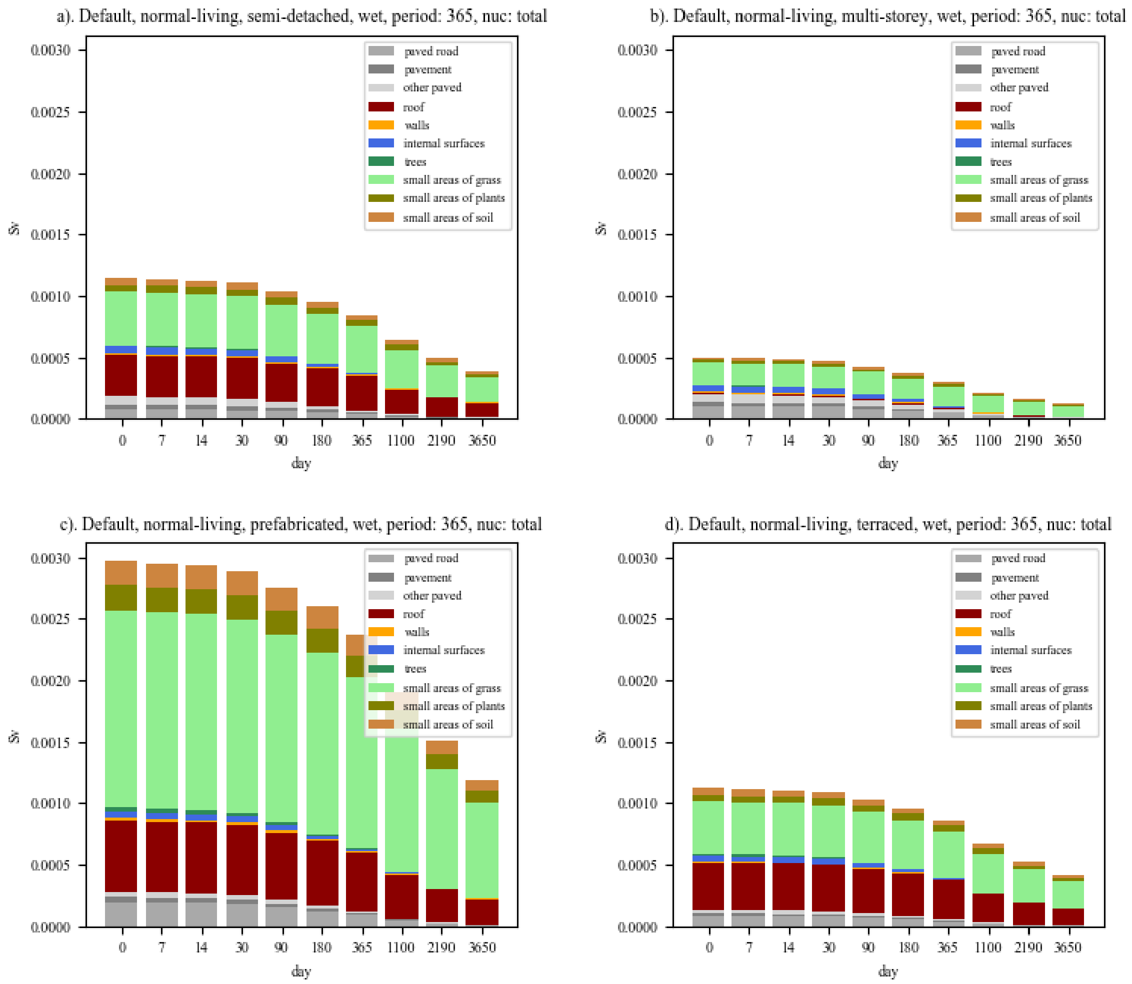

Figure 2.4. Predicted contribution to annual dose from different surfaces in four ERMIN environments assuming wet deposition.

2.5.2 Occupancy

ERMIN calculates the normal-living dose using a weighted average of the indoor dose and the outdoor dose. The weighting factor is the occupancy parameter; the fraction of time that an individual spends indoors. ERMIN was run for a deposit of ^{137}Cs under dry conditions in a number of different environments using a range of the occupancy values from Appendix 1 table 1.8 as well as some representing extreme behaviour.

Table 2.6 Occupancy values used for ERMIN sensitivity analysis

Behaviour	**Occupancy factor**
Permanently outdoors, extreme behaviour	0.0
Outdoor life style, extreme behaviour	0.5
95th percentile of occupancy (from EXPOLIS; Appendix 1 table 1.8), i.e. 95% of population spend more time indoors than this value implies.	0.75
Mean of occupancy (from EXPOLIS; Appendix 1 table 1.8)	0.87
Current ERMIN default	0.9
Permanently indoors, extreme behaviour	1.0

The plots show that, as expected, predicted total doses are less when occupancy factor). This is most pronounced in the high shielding environments and least in the low shielded (compare Figure 2.6 and Figure 2.5). In the low shielded the difference between the current ERMIN default occupancy factor (0.9), the mean of EXPOLIS values (0.87) and the 95th percentile (0.75) is small. However, in the high shielded environment, it is more pronounced but still less than 25% (compare Figure 2.6c and d).

The low shielded show the least variation in the predicted relative contribution of different surfaces. Whereas in the high shielded environment there are considerable differences, most notable is that even in the small range between the 95th percentile occupancy (0.75) and the current default (0.9) the contribution of grass surfaces changes from about 30% of predicted dose to about 10% (compare Figure 2.6c and e).

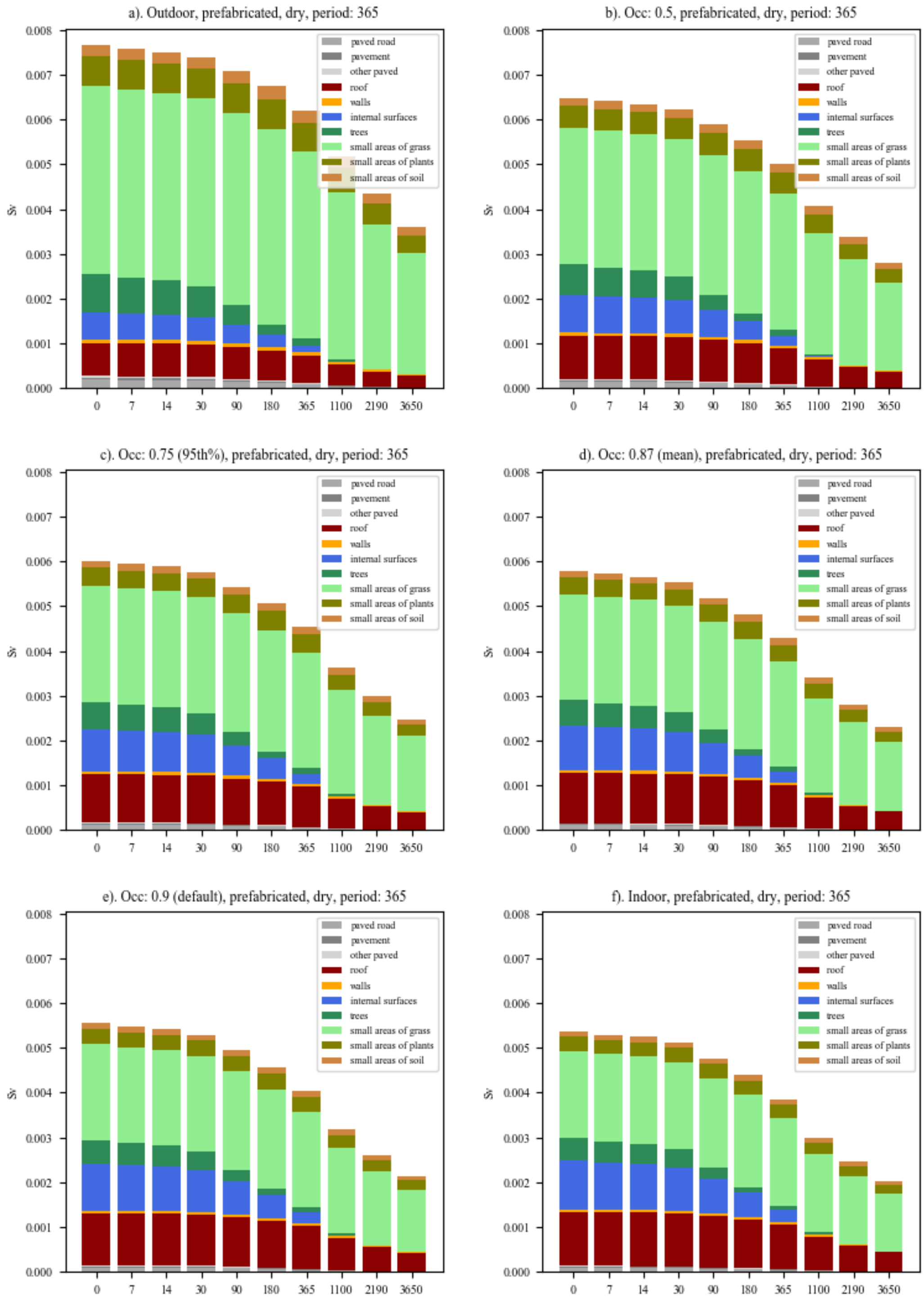

Figure 2.5. Predicted contribution to annual dose from different surfaces in prefabricated environment with different levels of occupancy.

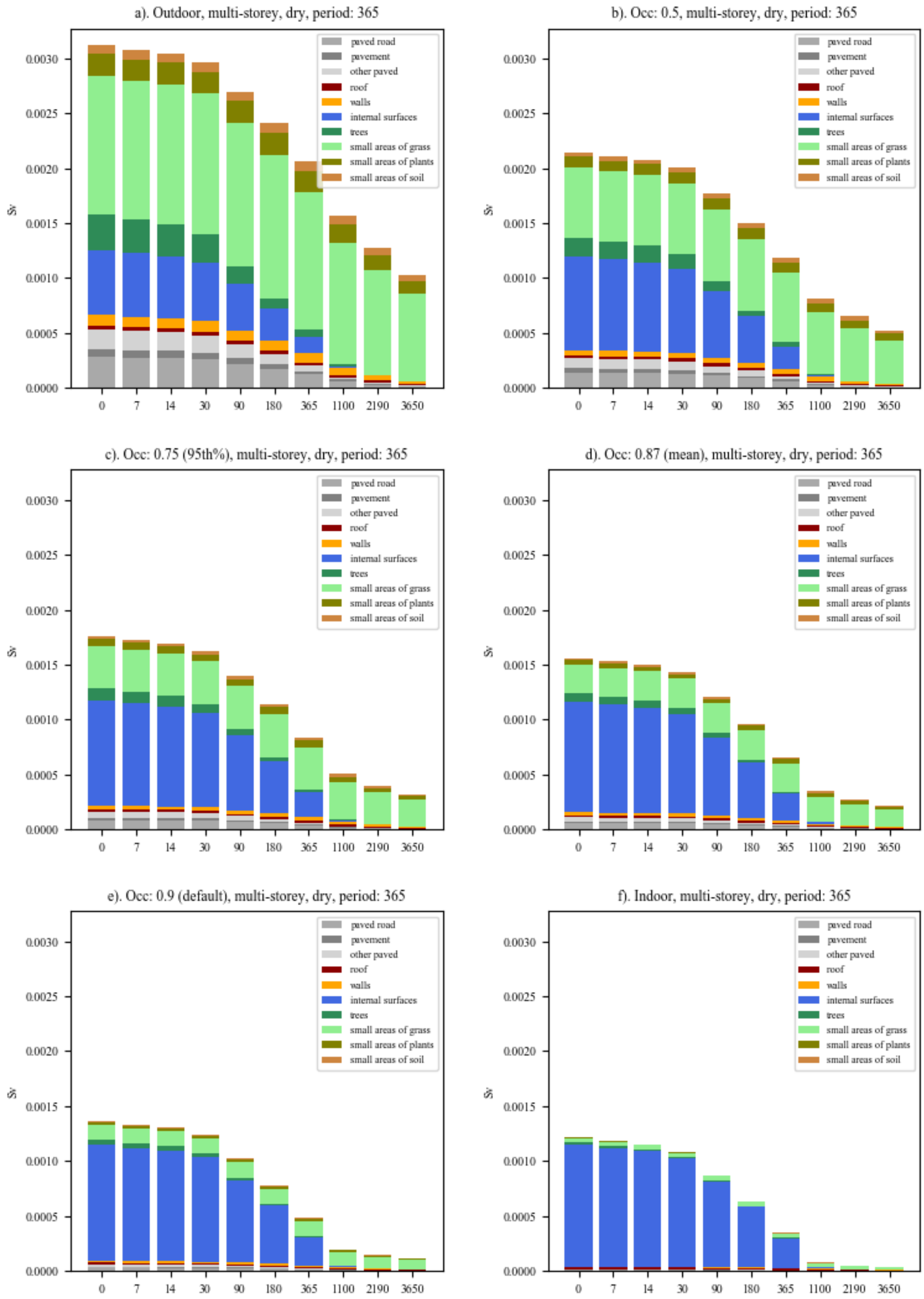

Figure 2.6. Predicted contribution to annual dose from different surfaces in multi-storey environment with different levels of occupancy.

The ERMIN unit doses library (UDL) contains the data that describe the dose rates from surfaces to various locations in each of the environments. The dose rates have been obtained from previous studies that use Monte Carlo codes to examine the shielding effects of different structures. Many of these studies provided sets of dose rates for receptors placed at different locations within the building, for example, the multi-storey building has receptors in the basement and on the ground floor, second floor and fourth (top) floor. Similarly, the semi-detached building has a ground floor and a first floor. ERMIN calculates a single indoor dose by using a weighted average of these locations. The default ERMIN averaging scheme weights the indoor locations equally and might represent a person moving around in the building uniformly. Figure 2.7, Figure 2.8 and Figure 2.9 compare the contributions of surface to predicted annual indoor dose of the default weighting scheme with other specific locations in different environments. Figure 2.7 shows the results for the prefabricated environment and it can be seen that even in this generally low shielded environment, the basement location is well shielded from radiation from external surfaces and so the dominant surface is internal. The default weighting and the ground floor and "below the roof" locations (attic) are similar for most surfaces, with the roof surface making a much bigger contribution for "below the roof" location.

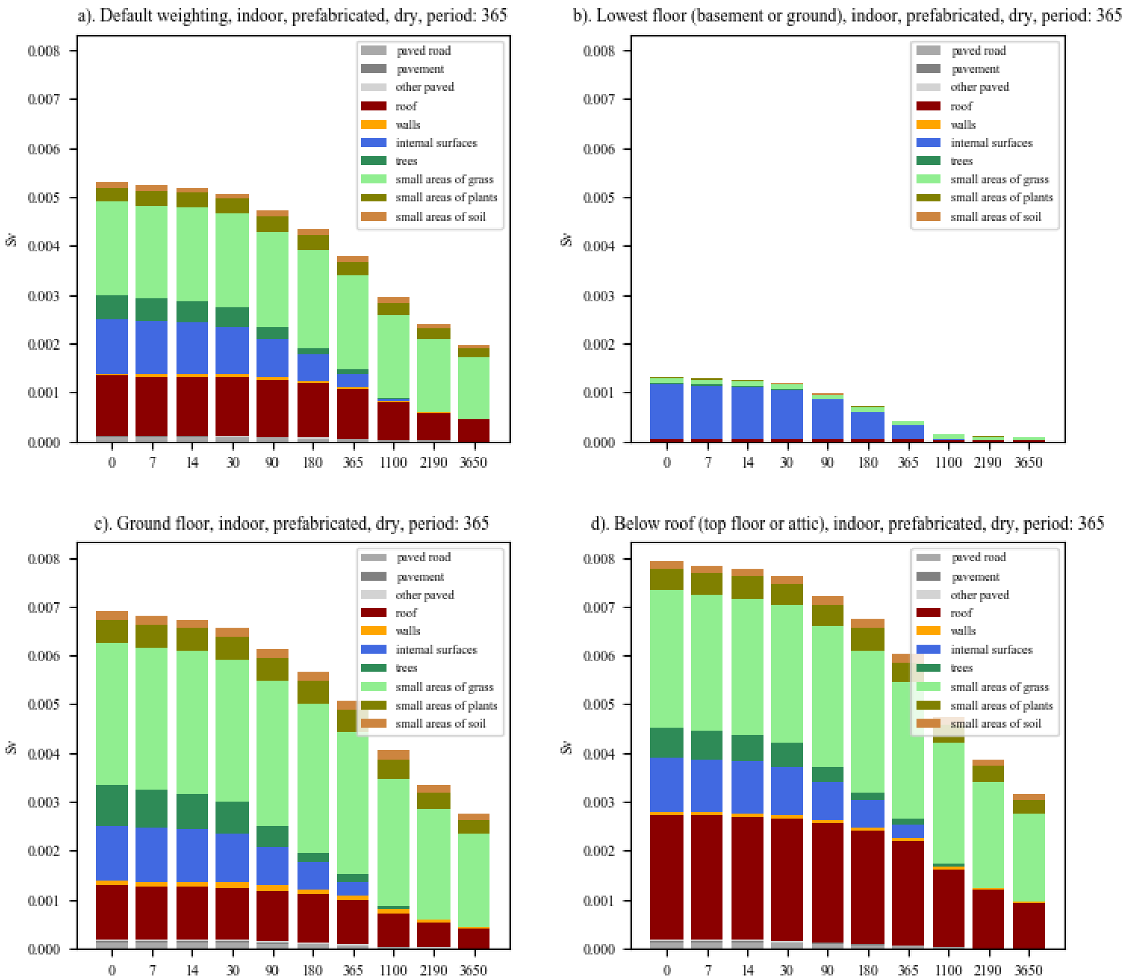

Figure 2.7. Predicted contribution to annual indoor dose from different surfaces in prefabricated environment assuming different indoor location averaging schemes.

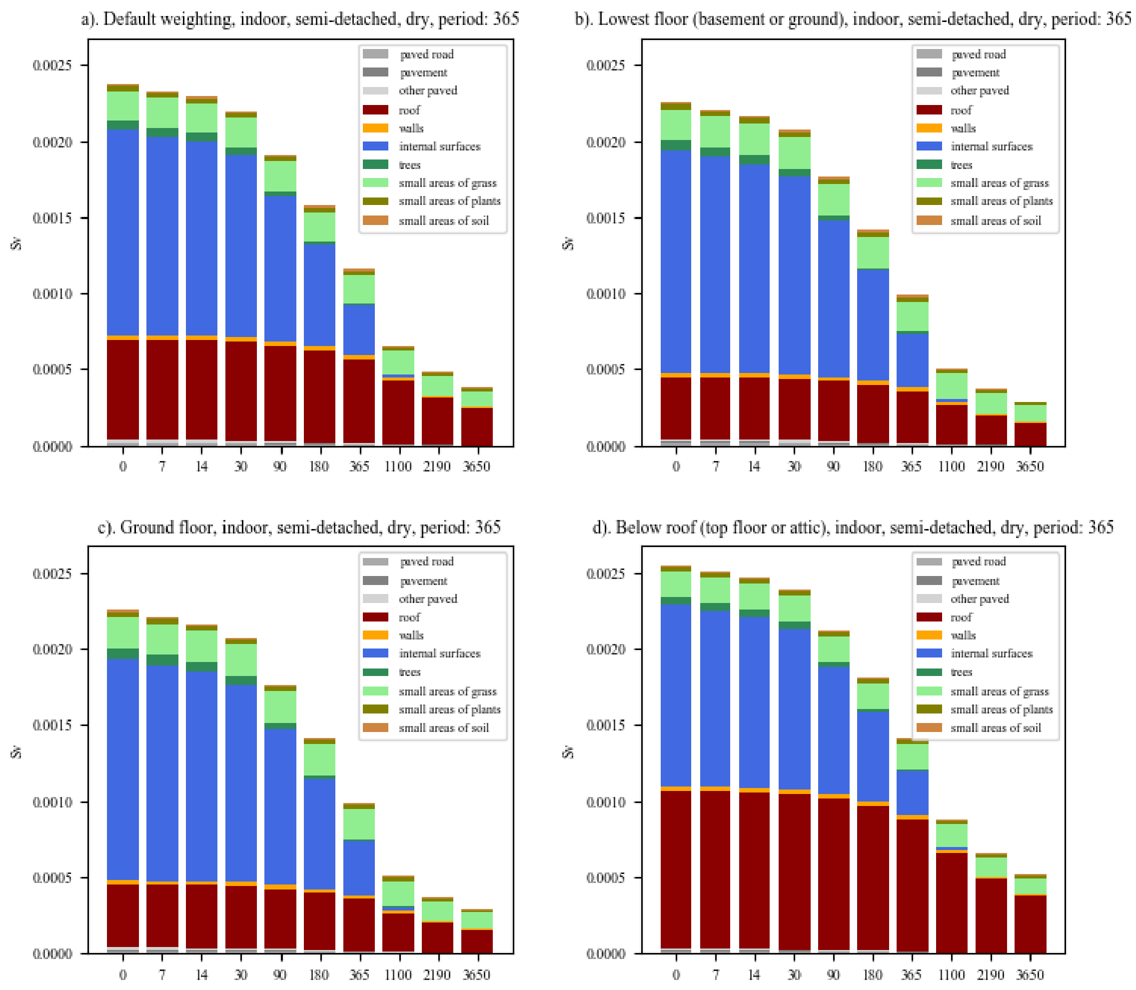

Figure 2.8. Predicted contribution to annual indoor dose from different surfaces in semi-detached house environment assuming different indoor location averaging schemes. NB this environment does not have a basement so the lowest flow is the ground floor and b. and c. and therefore the same. This environment also lacks an attic.

Figure 2.9 shows results for the multi-storey building environment. This shows a similar pattern to the other environments but because all locations are generally well shielded from radiation from outside the building, internal surfaces dominate the dose in all indoor locations, although the enhanced shielding in the basement and the enhanced contribution from the roof for the “below the roof” location can be seen. However when the default normal-living occupancy is assumed these effects are somewhat obscured, see Figure 2.10.

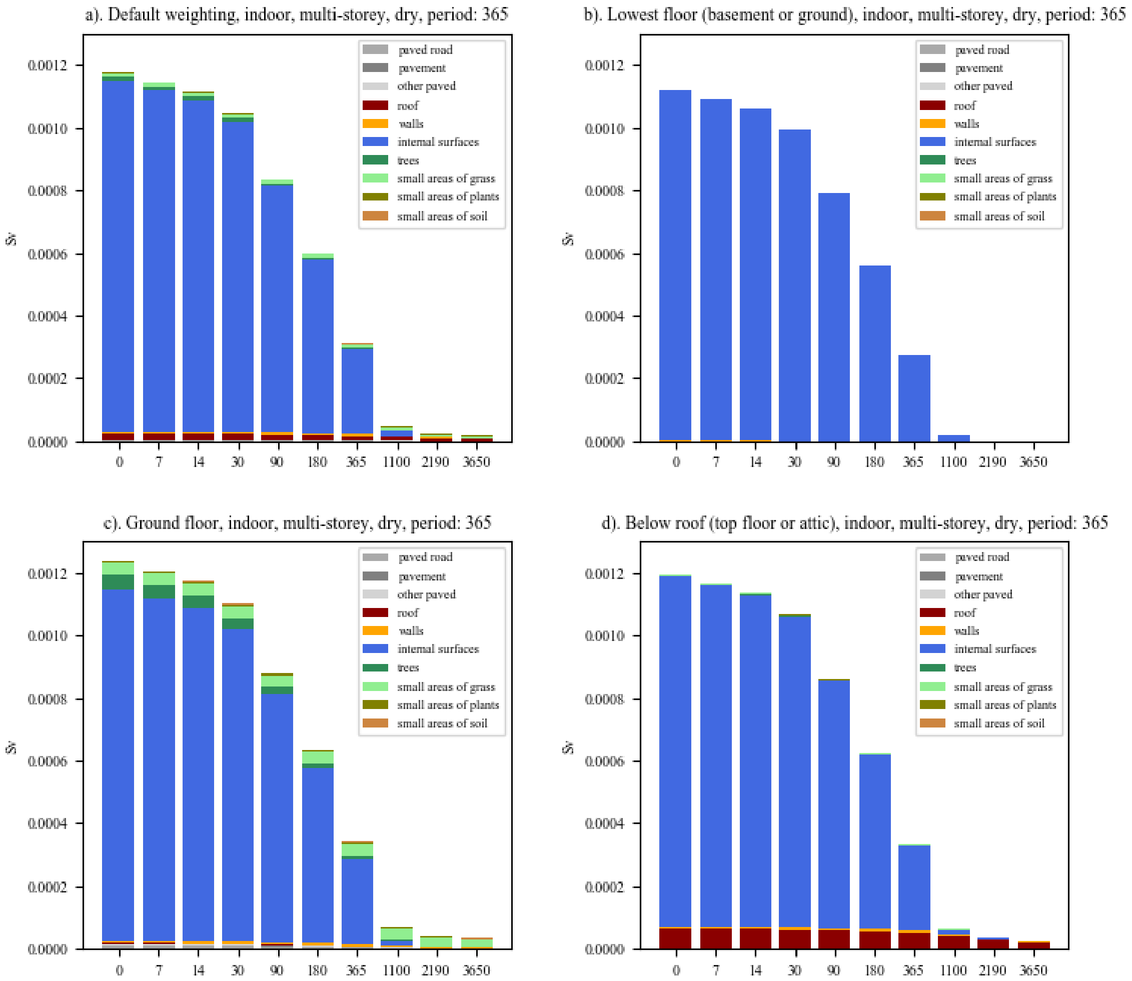

Figure 2.9. Predicted contribution to annual indoor dose from different surfaces in multi-storey building environment assuming different indoor location averaging schemes.

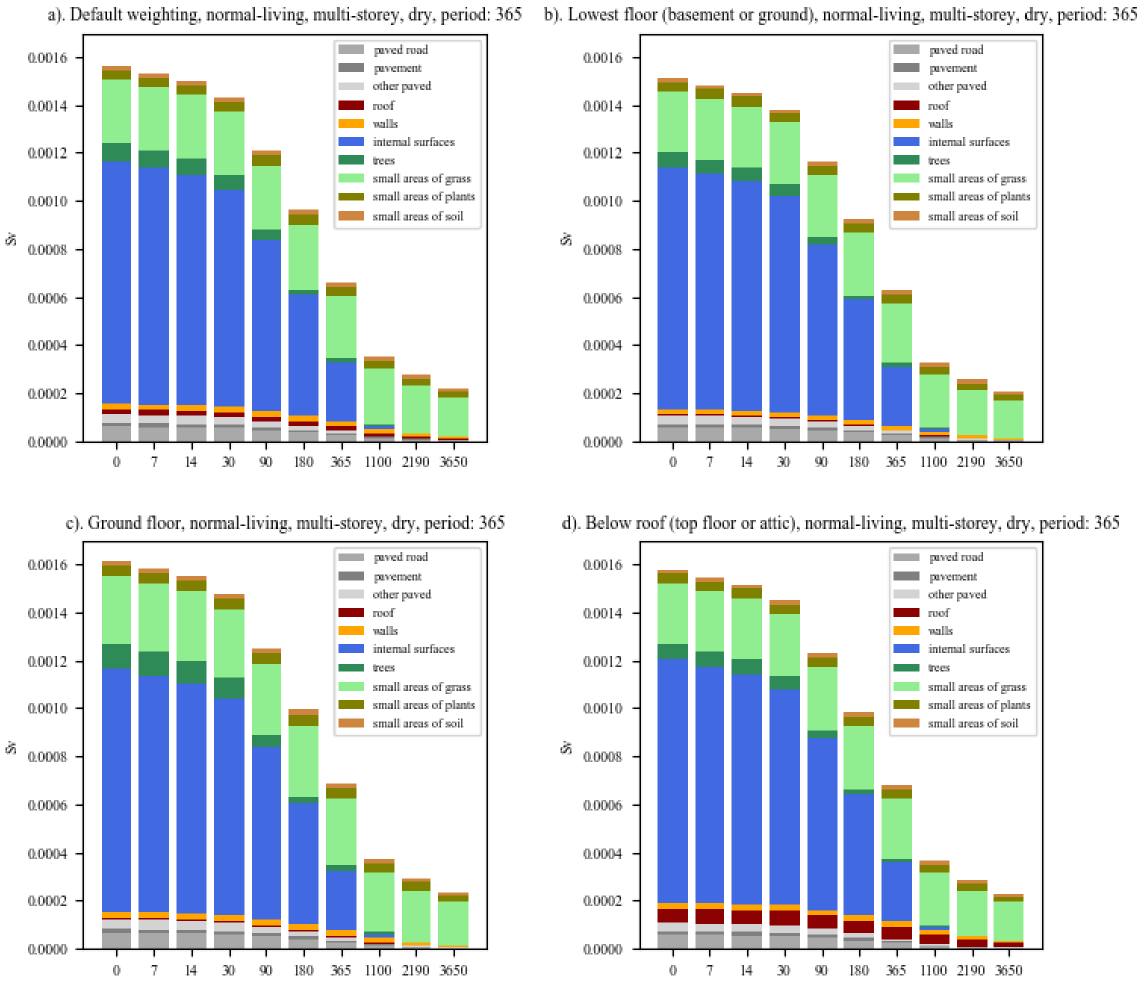

Figure 2.10. Predicted contribution to annual normal-living dose from different surfaces in multi-storey building environment assuming different indoor location averaging schemes.

2.5.3 Initial deposition on to urban surfaces

In the ERMIN model, the initial deposition onto the reference surface is used to estimate the initial deposition onto other urban surfaces included in the model using a dataset of surface, deposition and particle group dependent ratios.

2.5.3.1 Initial deposition to interiors

For all surfaces except internal surfaces, the initial deposition ratios in the ERMIN dataset have been obtained from field measurements and experiments. For internal surfaces, the deposition ratio is derived from a model. Appendix 1 Section 1.1.2 describes the model and suggests distributions for the input parameters. Figure 2.11 illustrates the results of a Monte Carlo analysis of the dry indoor deposition ratio for different room heights, and also assuming a uniform distribution of room heights from 2-5m. When it is wet there is an additional layer of uncertainty since indoor deposition is perforce dry and is therefore only a ratio to the dry component of outdoor deposition. Currently assumes that under wet conditions the ratio between dry and wet conditions is 1:19.

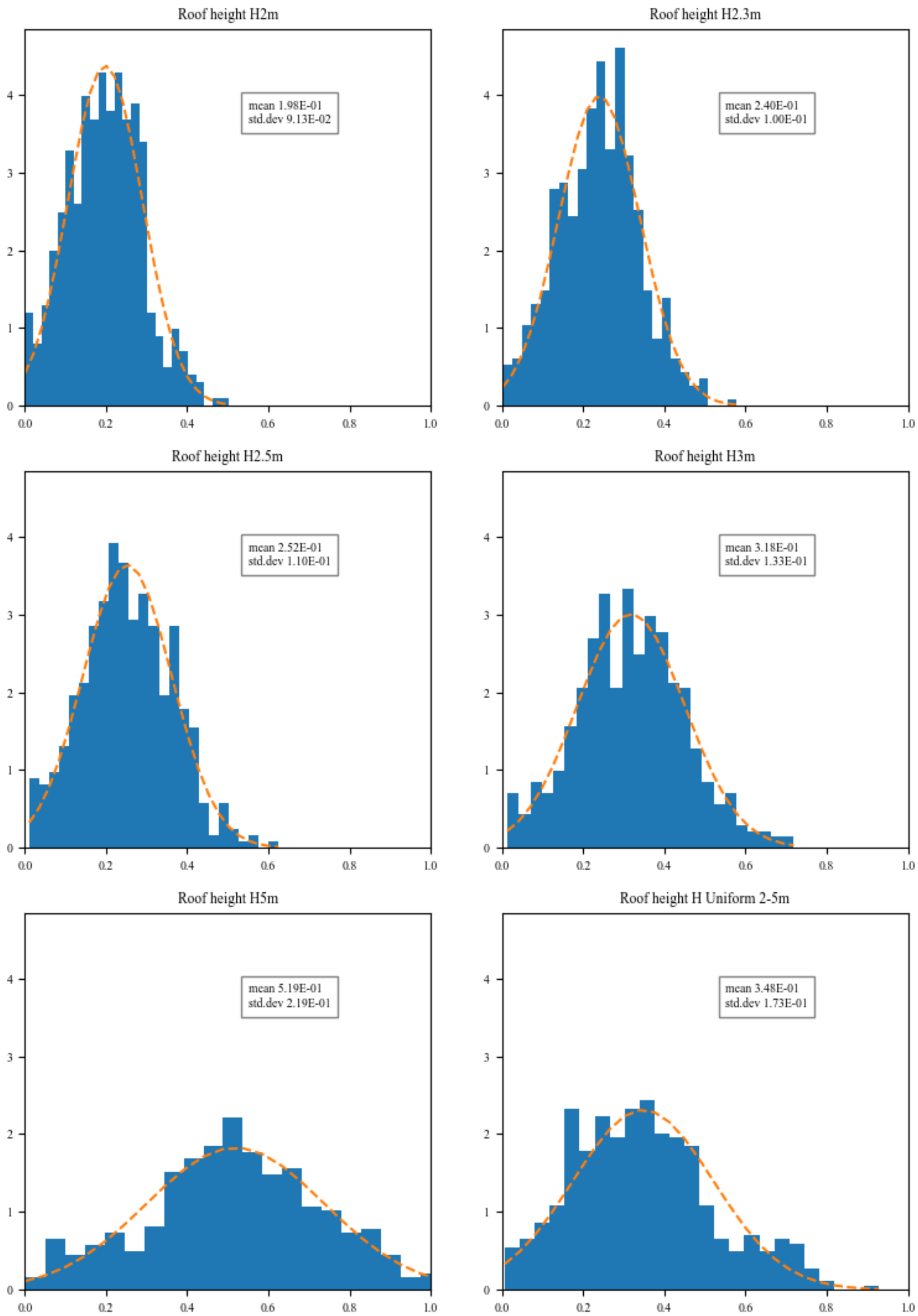

Figure 2.11. Predicted indoor dry deposition ratio for different room heights.

2.5.3.2 Initial deposition to roofs

While ERMIN has a single generic roof with a single set of deposition ratios, Appendix 1 Table 1.1 identifies several different roofing materials and gives parameters for the distributions of deposition of ratios for each material. Figure 2.12 shows box plots from a Monte Carlo analysis of these distributions. In addition to initial deposition, it might be expected that different roofing materials would have different shielding properties but this is not included in the analysis. Notwithstanding, the stochastic uncertainty in most situations is relatively low. In many urban situations there is likely to be a variety of roofing materials so there is probably little advantage in requiring the user to specify a roof material.

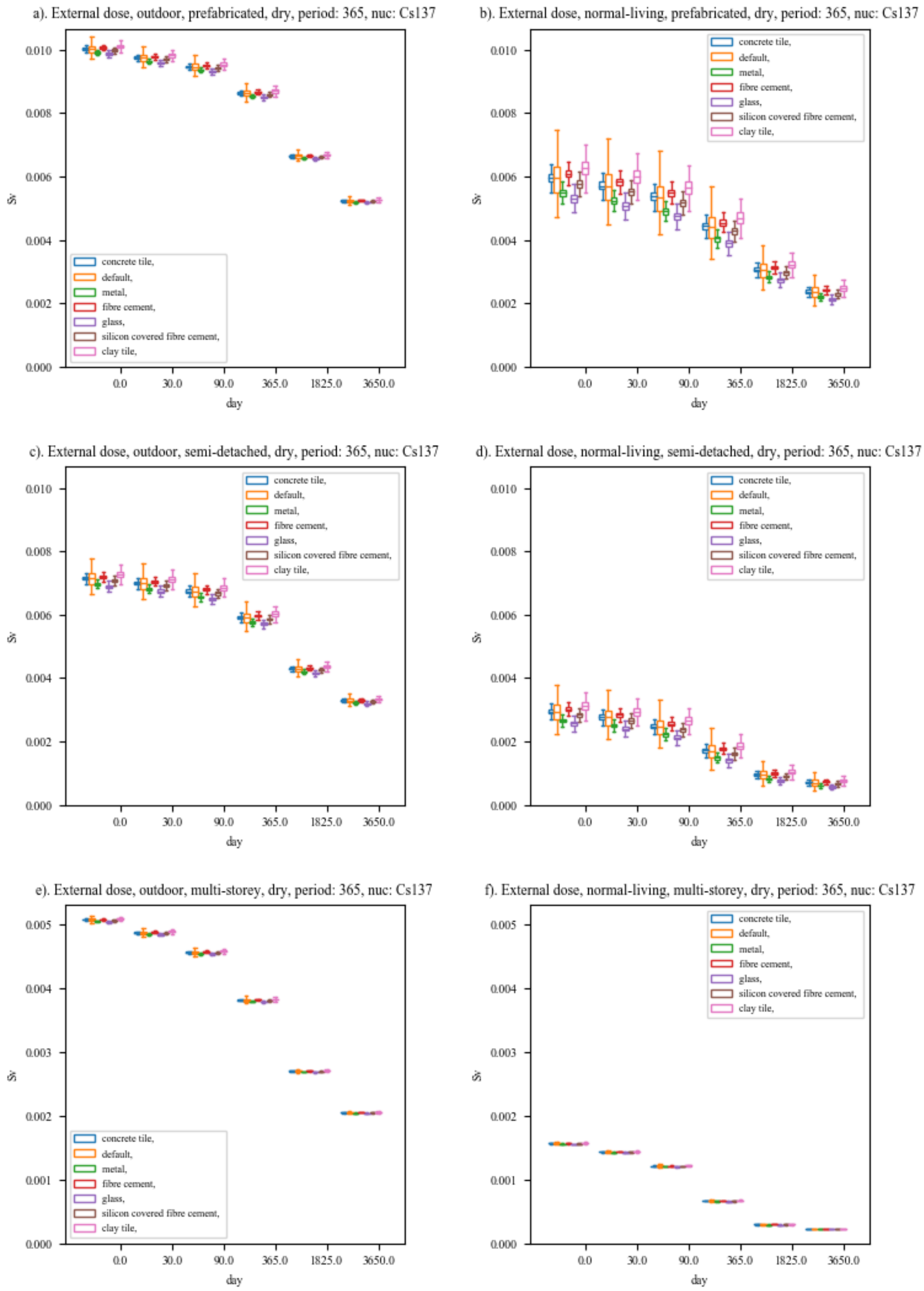

Figure 2.12. Box plots of a Monte Carlo analysis of predicted annual dose for various environments and locations following dry deposition from all surfaces. In each series, the distributions for dry deposition ratios parameters for roofs of a particular surface were sampled whilst all other surfaces were kept at the default

value. (The box for each marker indicates the inter quartile range (IQR), the range from quartile 1 (Q1) to quartile 3 (Q3), therefore 50% of results are contained in the box, the line is the 2nd quartile or the median. The 'whiskers' are defined as multiplies of the IQR, the lower whisker is Q1 - 1.5IQR and the upper whisker is Q3 + 1.5IQR).

2.5.3.3 Initial deposition to all surfaces

Figure 2.13, shows the results of a Monte Carlo analysis of ERMIN in which the distribution of the ratio under dry conditions for each surface was sampled in turn keeping the ratios for other surfaces fixed at the default ERMIN value. In a final Monte Carlo analysis, the distributions of the ratios for all surfaces were sampled together assuming no correlation. Since these are ratios to initial deposition on grass, grass itself is not included in the analysis. For this analysis, the roofs were assumed to be concrete tiles and for calculation of indoor deposition ratios rooms were assumed to be 2.5m.

In all the outdoor locations, trees appears to be the most significant surface; of course if the event took place in winter when there are no leaves on deciduous tree or it was an environment with few or no trees then this would not be the case. In more built up areas, the paved is also significant for outdoor locations.

When normal living assumptions are assumed, the uncertainty on the indoor ratio is by far the most significant in the more shielded environments such as the multi-storey building, and it should be noted that this assessment only incorporates some of the uncertainty; for example, the impact of modern ventilation systems is not included. However, this uncertainty should not be overstated either; the indoor surface in these situations becomes significant because of the shielding offered by the building to deposition on other surfaces. Under wet conditions with enhancement outdoor deposition one would expect the uncertainty on the indoor deposition to be less significant.

Uncertainty roofs and trees have significance in the low shielding prefabricated environment. It should be noted that the significance of trees persists for several years; this does not imply that the radioactivity remains on the trees; in fact, ERMIN assumes that most of this activity is transferred to the soil below.

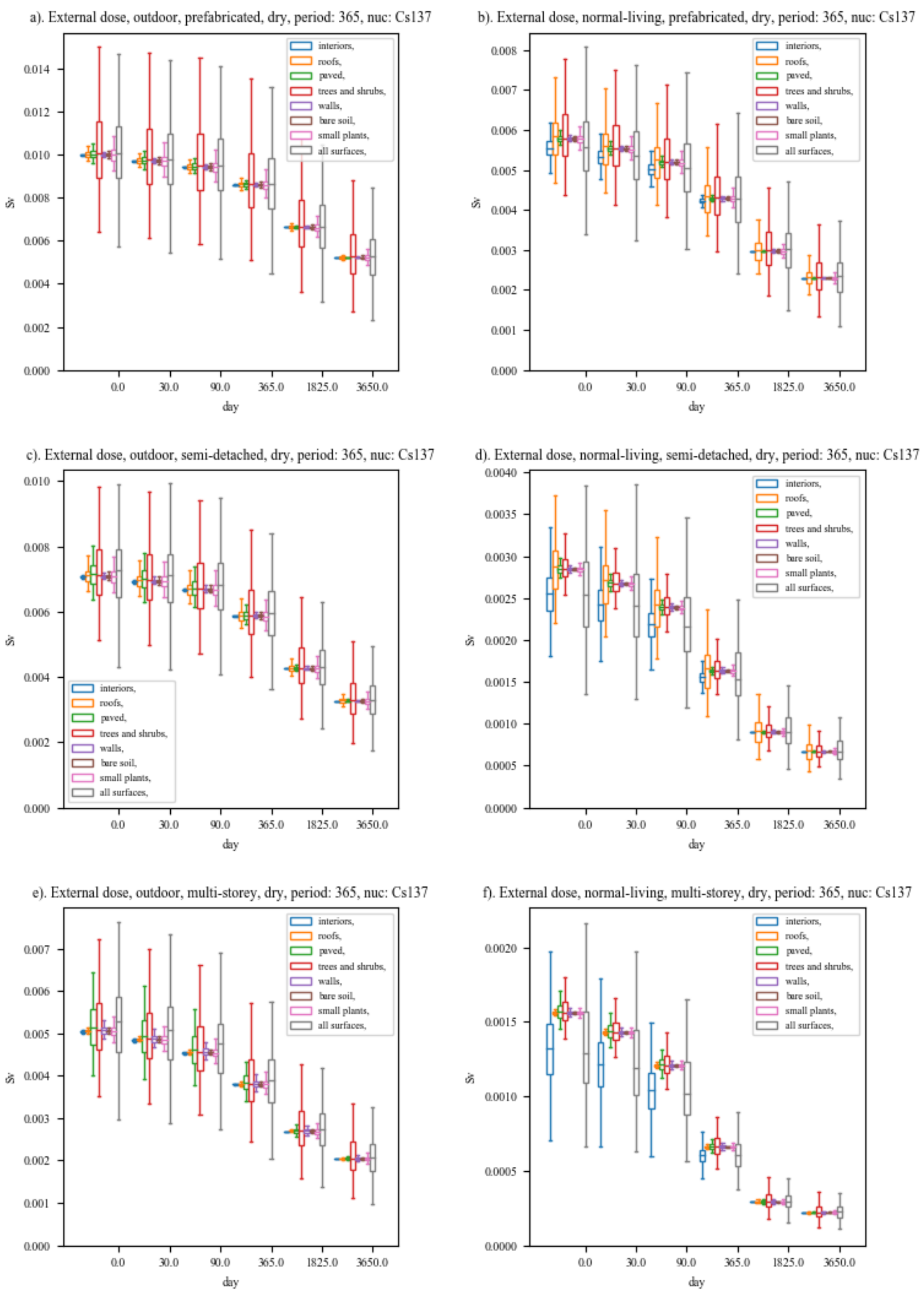

Figure 2.13. Box plots of a Monte Carlo analysis of predicted annual dose for various environments and locations following dry deposition. In each series, the dry deposition ratio for a particular surface was sampled whilst all other surfaces were kept at the default value. In the "All surfaces" series all ratio distributions were sampled. (The box for each marker indicates the inter quartile range (IQR), the range from quartile 1 (Q1) to quartile 3 (Q3), therefore 50% of results are contained in the box, the line is the 2nd quartile or the median. The 'whiskers are defined as multiples of the IQR, the lower whisker is Q1 - 1.5IQR and the upper whisker is Q3 + 1.5IQR).

2.5.4 Surface retention

For most surfaces, retention is ERMIN models the surface retention by either a one or two term exponential:

$$C(t) = C(0)\left[A_1 e^{\left(-\frac{\ln(2)}{T_1}t\right)} + A_2 e^{\left(-\frac{\ln(2)}{T_2}t\right)} \right] e^{\left(-\frac{\ln(2)}{T_{1/2}}t\right)}$$

For a two term exponential, A_1 represents a fraction of the deposition that weathers quickly and A_2 represents the remaining fraction that weathers slowly. A one term exponential has an A_1 value of 1 and A_2 value of 0 and all deposition weathers at the same rate. T_1 and T_2 are the retention half-lives of material in the A_1 and A_2 components. $T_{1/2}$ is the half-life of the radionuclide. C(0) and C(t) are the surface concentrations at time 0 and t.

Migration down the soil column is modelled using a one dimensional convective-dispersive, local equilibrium mass transport model (Bunzl et al, 2000). For small plants and for grass the retention on the leaves is modelled by a one term exponential and when that radioactivity reaches the ground by the convective-dispersive model. Uncertainty on soil migration is discussed in Section 2.5.4.1.

For trees, the retention on the branches and trunks is modelled using a two term exponential expression. However, there is a third component (A3) that represents the contamination on leaves. Retention on tree leaves is assumed to relate only to leaf fall. For deciduous trees, the leaf loss is assumed to occur instantaneously at some time after deposition. For coniferous trees, the pine needles are assumed to fall and be replaced at a constant rate and so retention is a linear function. For both fallen deciduous leaves and coniferous pine needles the activity is held within the leaves on the ground surface until the leaves are removed or the leaves a mixed into the soil during a soil mixing countermeasure.

2.5.4.1 Soil migration

ERMIN uses a convective dispersive model for soil column migration. It is parameterised with two parameters; V_s which describes the downward velocity and D_s which describes the spread.

2.5.4.1.1 Caesium

The current ERMIN default values and the geometric mean (GM) values for V_s and D_s for “clay/loam”, “Sand” and “Organic” for Caesium (See Appendix 1 table 1.5) were used to generate plots of annual external dose and soil layer contamination for comparative purposes. The values used are summarised in Table 2.7.

Table 2.7 ERMIN default and GM values of Vs and Ds for different soil types.

Soil type	D_s cm^2 per year	V_s cm per year
Current ERMIN defaults	0.6	0.15
Clay/Loam	0.20	0.06
Sand	0.11	0.15
Organic	0.94	0.69

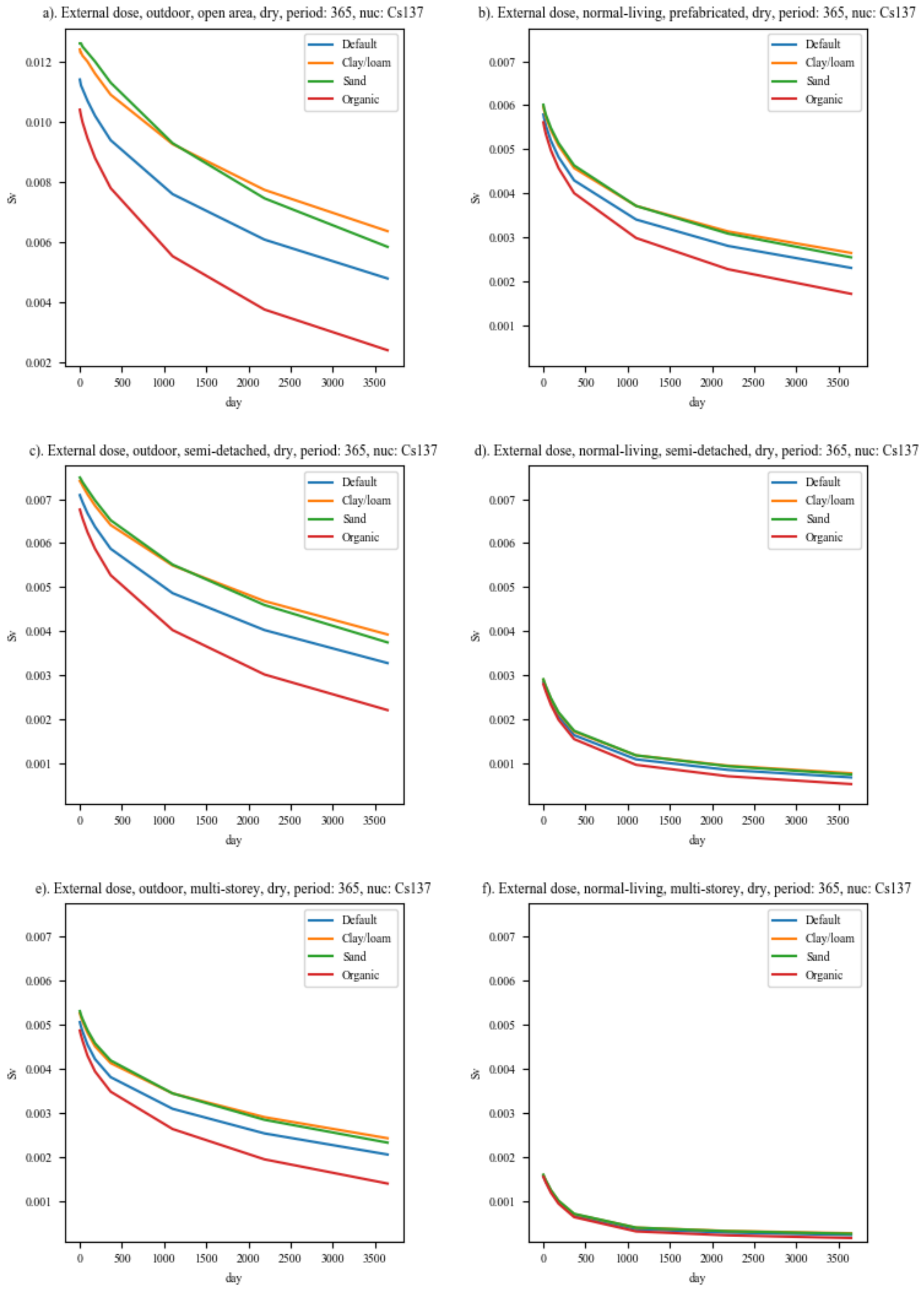

Figure 2.14. Predicted annual dose in various ERMIN environments, soil migration model parameterised for Caesium migration in different soil types.

Figure 2.14 shows the predicted annual doses in various ERMIN environments assuming different soil types. The largest spread of lines are seen in those situations where soil surface contribute the most to the total dose, hence they are largest outdoors in the open area and smallest assuming normal living in the multi-storey environment. The ERMIN default values produce dose predictions that sit in the middle of the range.

Figure 2.15 shows the predicted soil profile of radioactivity for different soils at various times. Organic soils which have the highest mean V_s and D_s parameters for Caesium show a corresponding much faster movement through the soil profile, whereas clay/loam soils, which have the lowest mean V_s and second lowest mean D_s, has the slowest movement.

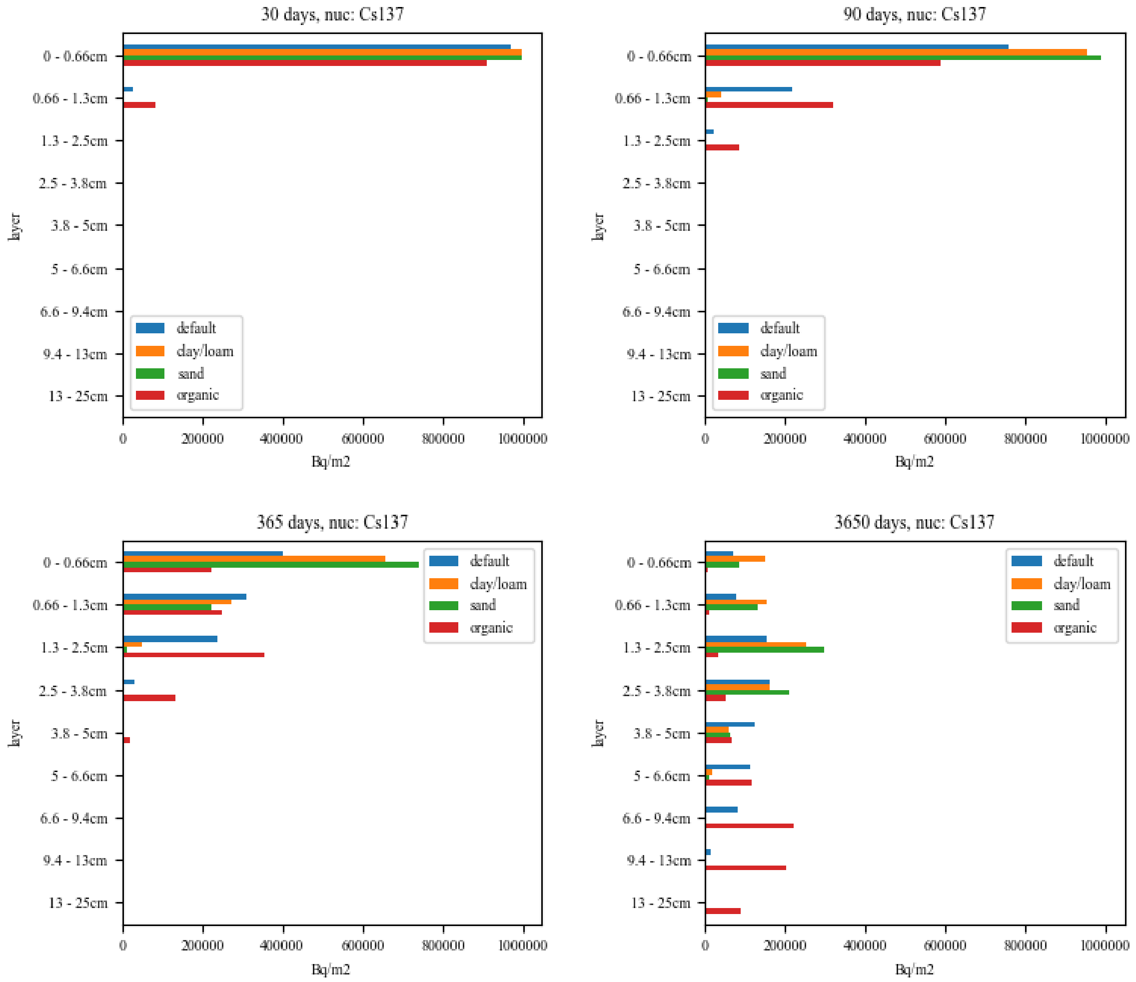

Figure 2.15. Predicted caesium radioactivity in the soil profile at different times following deposition. NB the predictions are for the open area environment where there are no trees and therefore no transfer from trees via leaf-fall.

As a further exercise, the maximum and minimum values for V_s and D_s were extracted from Appendix 1 table 1.5 to define a maximum parameter space. Figure 2.16 shows the predicted annual dose, and Figure 2.17 shows the predicted soil radioactivity profiles.

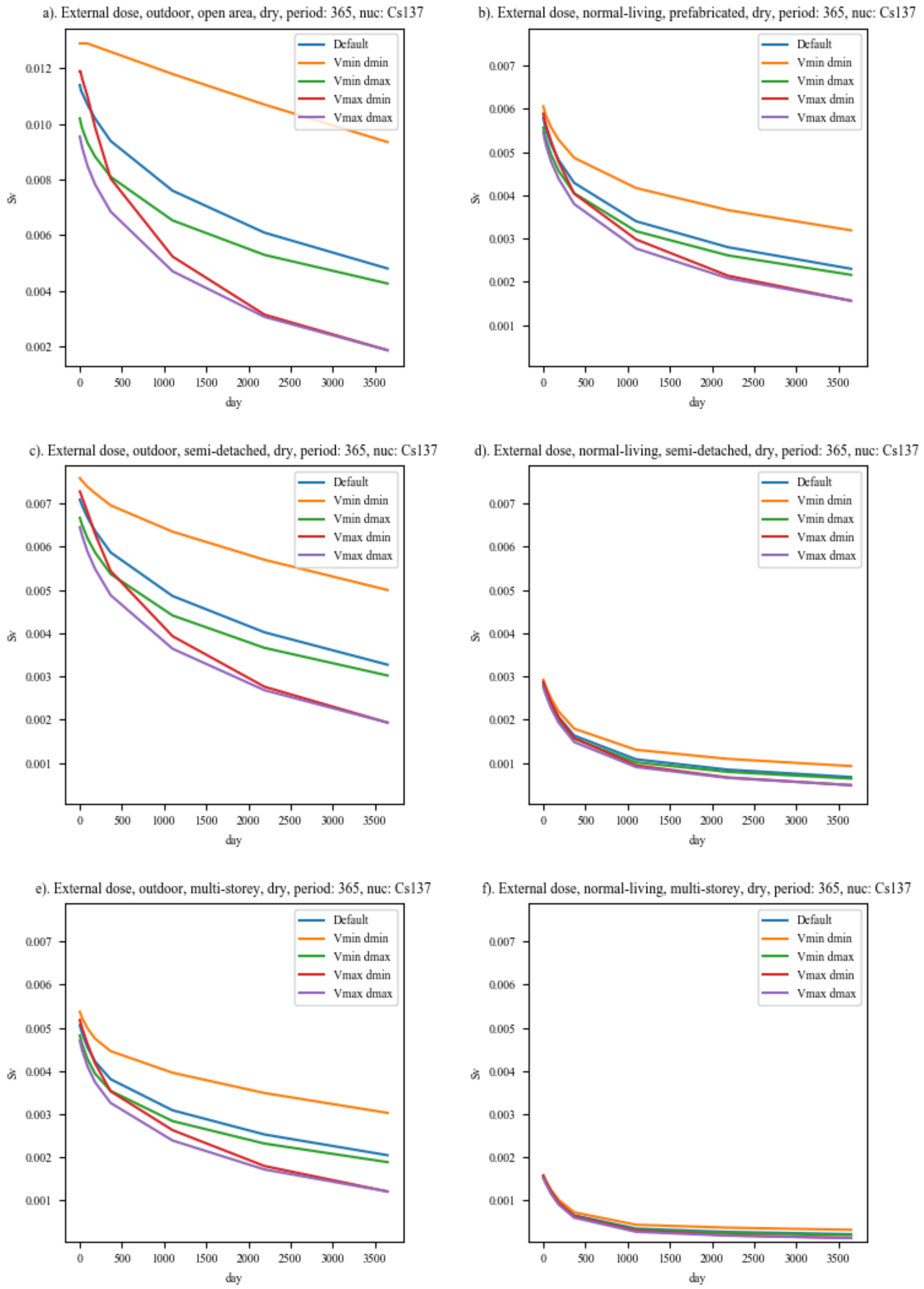

Figure 2.16. Predicted annual dose in various ERMIN environments with the maximum and minimum values for Vs and Ds to define the parameter space for the soil migration model.

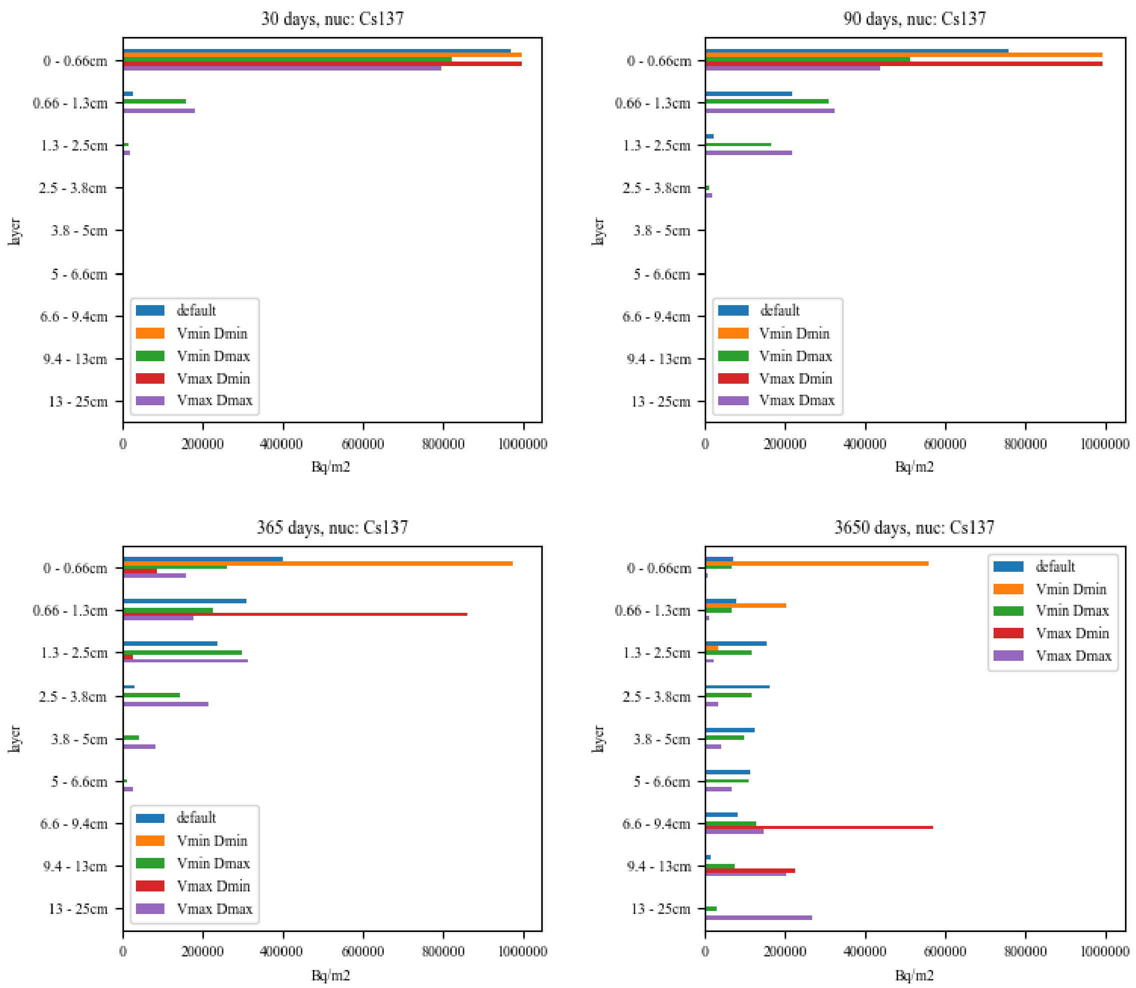

Figure 2.17. Predicted radioactivity in the soil profile at different times following deposition using the maximum and minimum values for Vs and Ds to define the parameter space for the soil migration model. NB the predictions are for the open area environment where there are no trees and therefore no transfer from trees to soil via leaf-fall.

Appendix 1 also provides information about the distributions of the V_s and D_s parameters which were randomly sampled in a Monte Carlo run of ERMIN for each of the soil types identified in Appendix 1; "all soils", "clay/loam", "Sand" and "Organic". The assumption was made that D_s has a lognormal distribution and Vs has a normal distribution. The parameters are given in Table 2.8 and Table 2.9.

Table 2.8 The arithmetic mean (AM) and standard deviation (SD) for the Ds parameter extracted from Appendix 1, Table 1.5 along with the derived parameters for the log normal distribution (μ, σ) used to describe the log-normal distribution of D_s.

Caesium D_s (cm^2 per year)				
Soil type	**AM**	**SD**	**μ**	**σ**
All soils	0.37	0.4	-1.38132	0.879855
Clay	0.36	0.3	-1.28532871	0.726192072
Sand	0.16	0.2	-2.30307	0.970043
Organic	1.07	0.7	-0.11047	0.596879

Table 2.9 The arithmetic mean (AM) and standard deviation (SD) for the V_s parameter extracted from Appendix 1, Table 1.5 used to describe the assumed normal distribution of V_s.

Caesium V_s (cm per year)		
Soil type	**AM**	**SD**
All soils	0.27	0.2
Clay/load	0.24	0.3
Sand	0.17	0.1
Organic	0.73	0.3

Figure 2.18 gives box plots of the predicted annual dose for various environments and occupancy combinations. In all plots, the spread increases with time. The spread is largest in situations in which soil surfaces make the largest relative contribution to dose, hence it is largest outdoors in the open area environment (Figure 2.18a) and smallest with normal living in the multi-storey environment (Figure 2.18f). There is considerable overlap between the tails of the plots for all combinations and all times, however (ignoring the 'all soils category') the boxes often do not overlap and this is particularly true for the organic soil type box which never overlaps with the boxes for other soils.

It should be noted that the tails represent an especially extreme case since they require that all the soil around the receptor that contributes doses (within a radius of a few tens of metres) can be parameterised with the same extreme V_s and D_s values. In reality, the properties of the soil are likely to be variable within this radius and the overall effect is to give an average dose more appropriately represented with values of V_s and D_s closer to the mean.

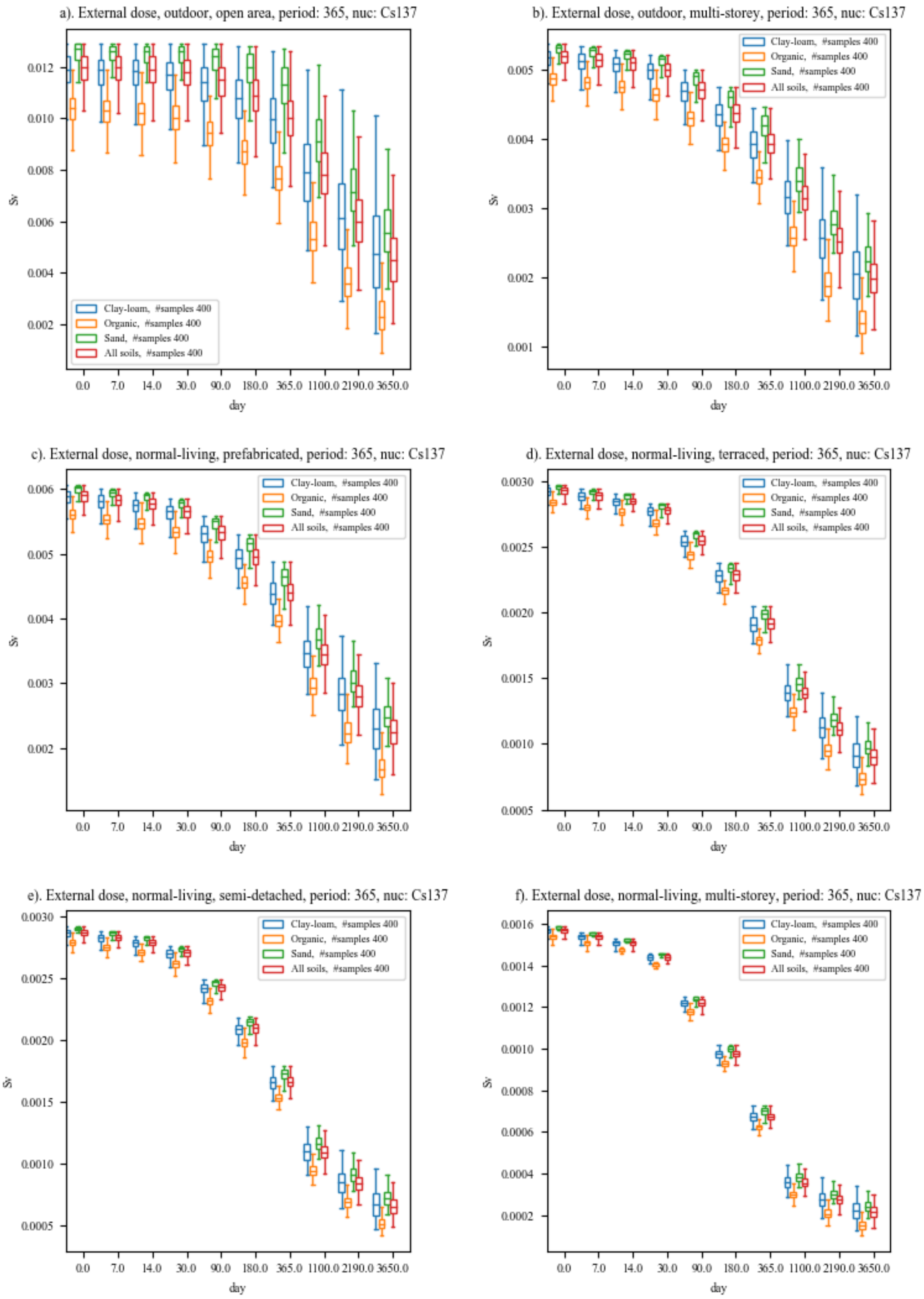

Figure 2.18 Box plots of a Monte Carlo analysis of predicted annual dose for various ERMIN environments with different soil types. (The box for each marker indicates the inter quartile range (IQR), the range from quartile 1 (Q1) to quartile 3 (Q3), therefore 50% of results are contained in the box, the line is the 2nd quartile or the median. The 'whiskers are defined as multiples of the IQR, the lower whisker is Q1 - 1.5IQR and the upper whisker is Q3 + 1.5IQR).

2.5.4.1.2 Iodine

Appendix 1 gives an approach to convert Caesium specific V_s and D_s values to other elements by scaling with a 'retardation factor'.

The GM values for V_s and D_s for "clay/loam", "Sand" and "Organic" soils, for Caesium in Appendix 1 Table 1.5 were converted to V_s and D_s for Iodine using the retardation factor based on the GM value for Kd taken from Appendix 1 Table 1.6. The derived parameters are given in Table 2.10 and were used to generate plots of annual external dose and soil layer contamination for comparative purposes. A deposition of 10^6 Bqm^{-2} of the iodine isotope ^{129}I was modelled. ^{129}I was chosen for its long half-life although it is not expected to be the most important isotope of iodine in most real accidents. ERMIN scenario '1' considers that Iodine, like Caesium, is deposited entirely as soluble aerosols.

The Kd values for Iodine are as usual relatively much smaller than for Caesium leading to relatively much higher values for D_s and V_s for Iodine compared with Caesium and compared with the current default values. In the soil profiles given in Figure 2.19 it can be seen that the activity moves relatively much more quickly through the soil profile for all soil types.

Furthermore the largest Kd for caesium is for Clay/loam soils (5.5E3) whereas for Iodine the largest Kd is for Organic soil (3.6E1). This means for that the Caesium radioactivity moves quickest for organic soils whereas for Iodine the radioactivity moves the slowest.

Table 2.10 Current ERMIN defaults values for Vs and Ds and values derived for Iodine for different soil types from the equivalent Caesium values.

Soil type	D_s cm^2 per year	V_s cm per year
Current ERMIN default	0.6	0.15
Clay/Loam	152.3	45.7
Sand	15.1	20.6
Organic	6.8	5.01

Figure 2.20 shows the predicted annual dose using parameters for the different soil types and the current ERMIN defaults. The effect of the much larger D_s and V_s values for Iodine compared to the defaults is particularly significant in the outdoor/open area situation (Figure 2.20a), with the predicted annual dose when using with the Iodine specific parameters falling close to zero within the first year. However, for the normal living situations (Figure 2.20b to f) the effect is much less significant. There are two reasons for this that can be seen by comparing Figure 2.21 with Figure 2.22 which give equivalent plots for the current ERMIN default soil parameters and parameters for iodine in clay soil respectively; firstly in these built environments the soil surfaces contribute a smaller fraction to the total dose because of the presence of other contributing surfaces and because of shielding properties, and secondly these environments contain trees and there is a transfer of radionuclides to the soil surface from trees via leaf fall (in ERMIN the radioactivity transferred this way is assumed to stay permanently in the leaf litter and not migrate down the soil column). It is this second process that explains why Figure 2.21c and Figure 2.22c are so similar.

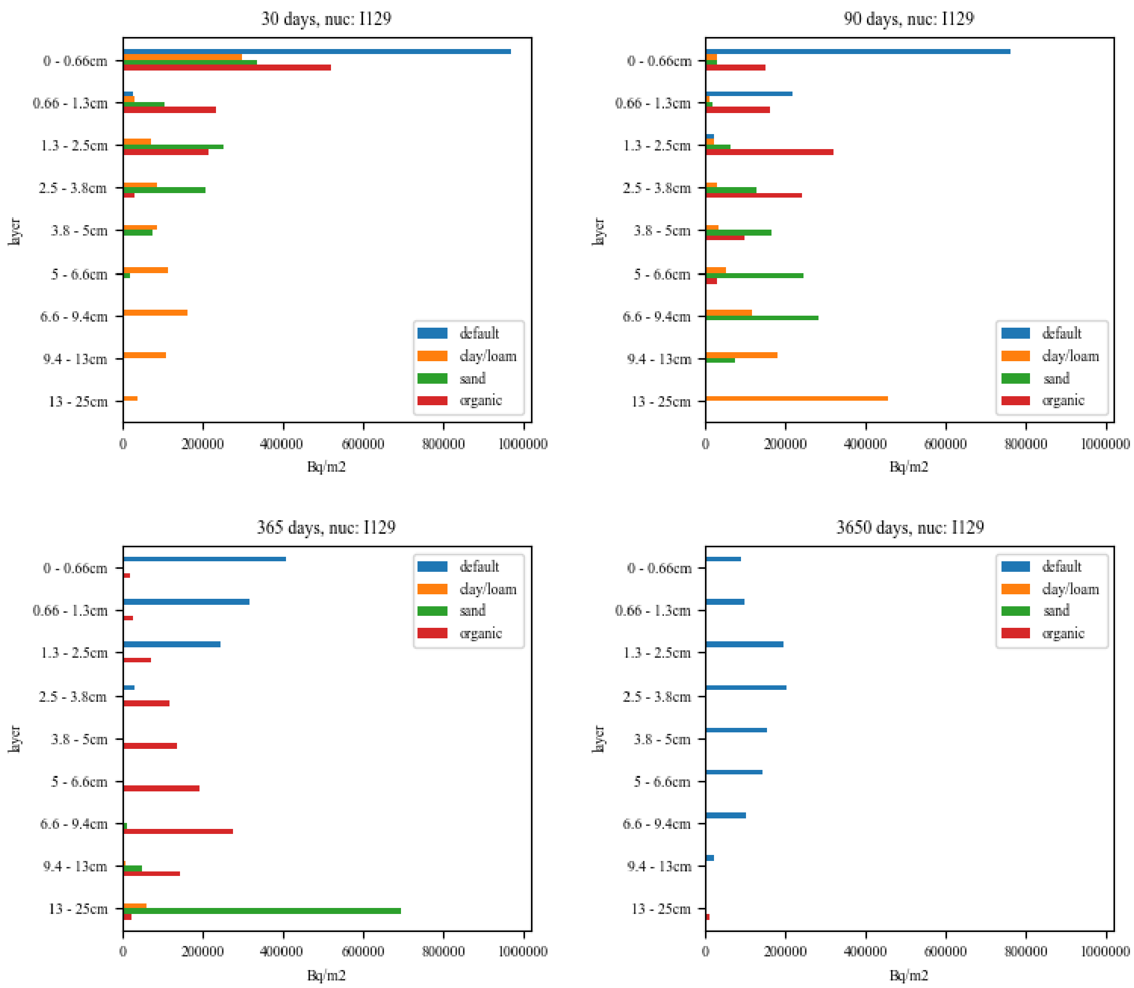

Figure 2.19. Predicted iodine radioactivity in the soil profile at different times following deposition. NB the predictions are for the open area environment where there are no trees and therefore no transfer from trees via leaf-fall.

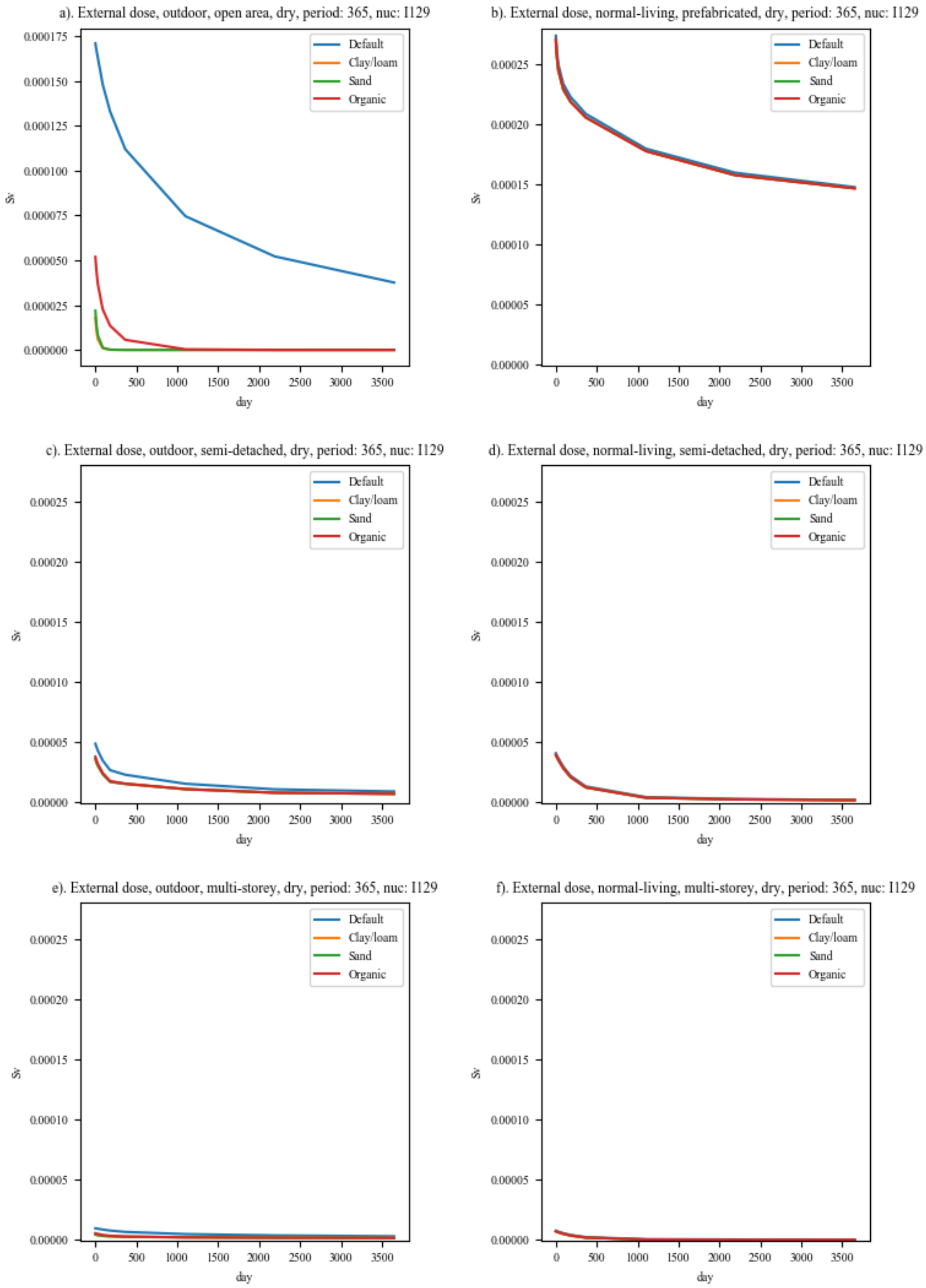

Figure 2.20. Predicted annual dose in various ERMIN environments with the soil migration model parameterised for iodine for different soil types

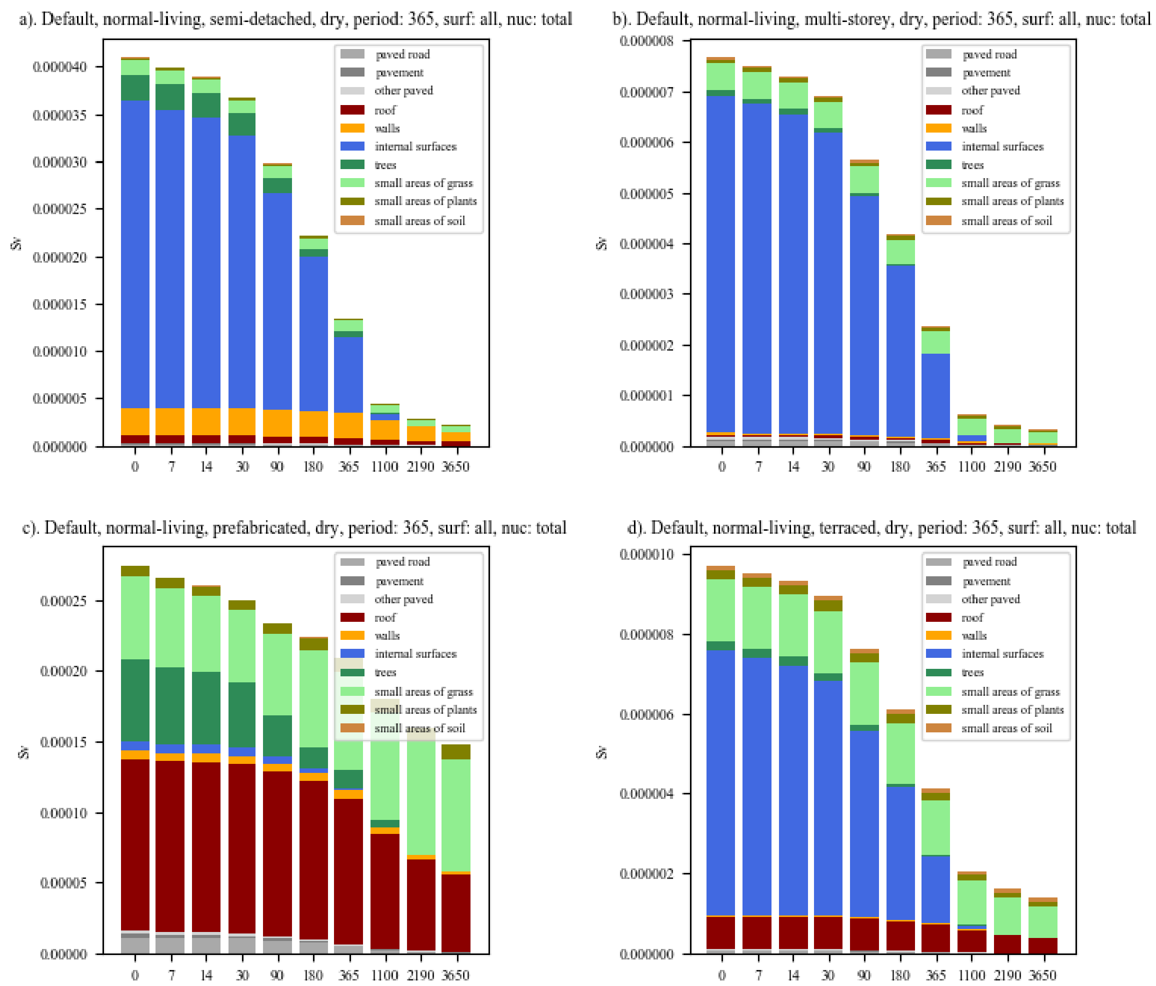

Figure 2.21. Predicted contribution to annual dose from different surfaces in four ERMIN environments assuming dry deposition if ^{129}I and Default soil parameters.

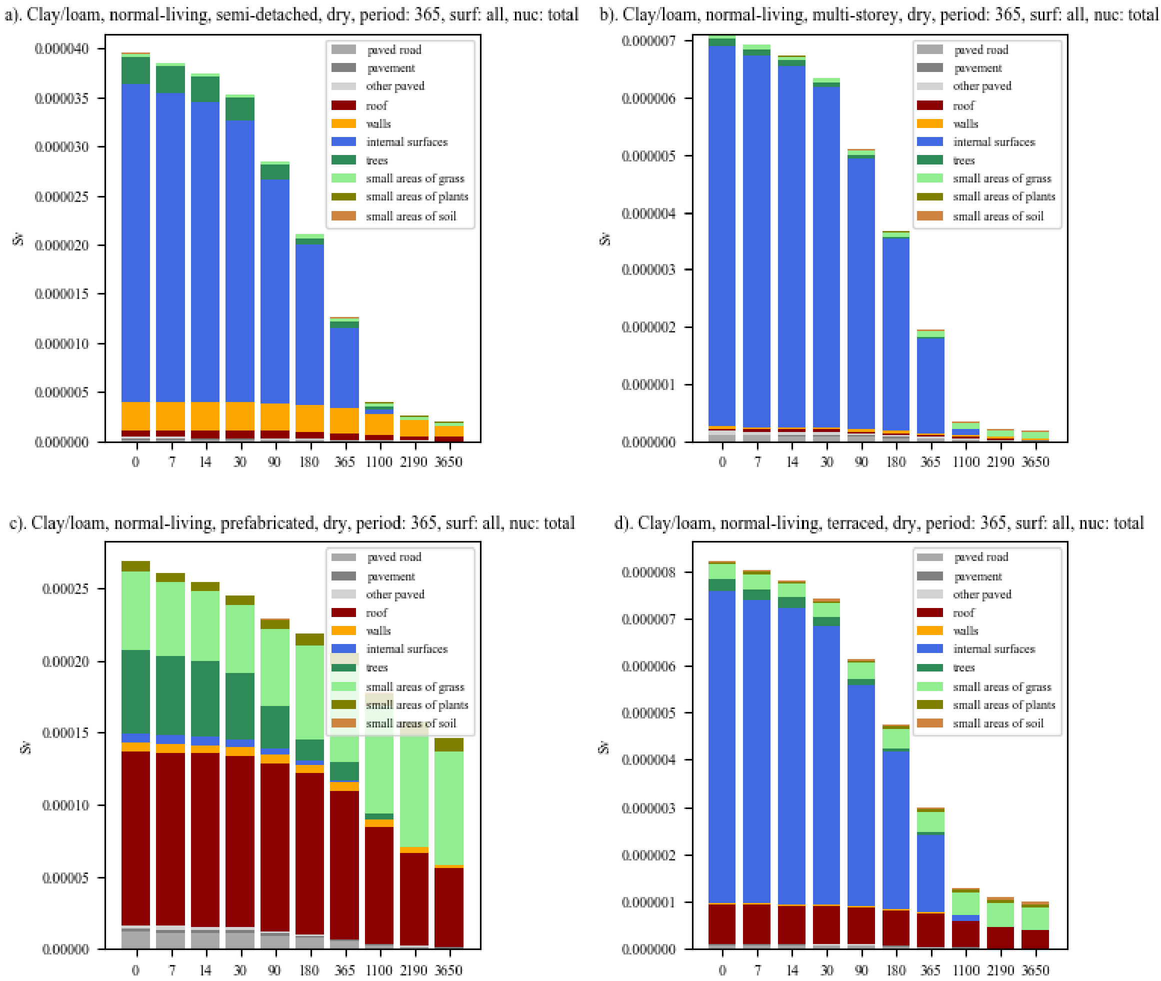

Figure 2.22. Predicted contribution to annual dose from different surfaces in four ERMIN environments assuming dry deposition if ^{129}I and soil parameters set to the mean values for clay/loam soils.

2.5.4.1.3 Ruthenium

Specific Ruthenium V_s and D_s parameters were derived from the geometric mean (GM) values for V_s and D_s for Caesium (Appendix 1 Table 1.5), for each of the soil types; “clay/loam”, “Sand” and “Organic”. The derivation was performed with retardation factors calculated from the GM value for Kd for each soil taken from Appendix 1 Table 1.6 (NB the arithmetic mean Kd for organic soils was used because no GM value was available). The resulting parameters are given in Table 2.11 and were used to generate plots of annual external dose (Figure 2.23) and soil layer contamination for comparative purposes (Figure 2.24).

A deposition of 10^6 Bqm^{-2} of isotope ^{106}Ru was modelled. ^{106}Ru was chosen because it has the longest half-life of all the Ruthenium isotopes in the ERMIN database, however that half-life is only 1 year which is smaller than the 10 year scope of the graphs.

Strictly, the V_s and D_s parameters apply to the component of Ruthenium that is deposited as soluble aerosols. The modelling predictions presented here assume that all the deposition is as soluble aerosols, however, in Scenario ‘1’ only 50% of the deposition is in this form with the remainder bound to insoluble fuel fragments. Scenario ‘3’ best matches the predictions here with 95%of the deposition as soluble aerosols.

The Kd values for clay/loam soils and for sand soil for Ruthenium are about an order of magnitude lower than the corresponding values for Caesium. The Kd value for organic soils (6.6E4, an AM value) is very much higher than the corresponding Caesium Kd. The results is that for sand and clay/loam soils the D_s and V_s values are larger than for Caesium and the current ERMIN defaults and the Ruthenium activity moves more quickly through the soils. For organic soils, the D_s and V_s values are very much less and the ruthenium radioactivity predictably moves much more slowly. However, these features are difficult to pick-out in the graphs which are dominated by the effect of the relatively short half-life of ^{106}Ru.

Table 2.11 Representative values for Vs and Ds derived for Ruthenium from equivalent Caesium values.

Soil type	D_s cm^2 per year	V_s cm per year
Defaults	0.6	0.15
Clay/Loam	2.1983	0.6595
Sand	1.6083	2.1931
Organic	0.0039	0.0028

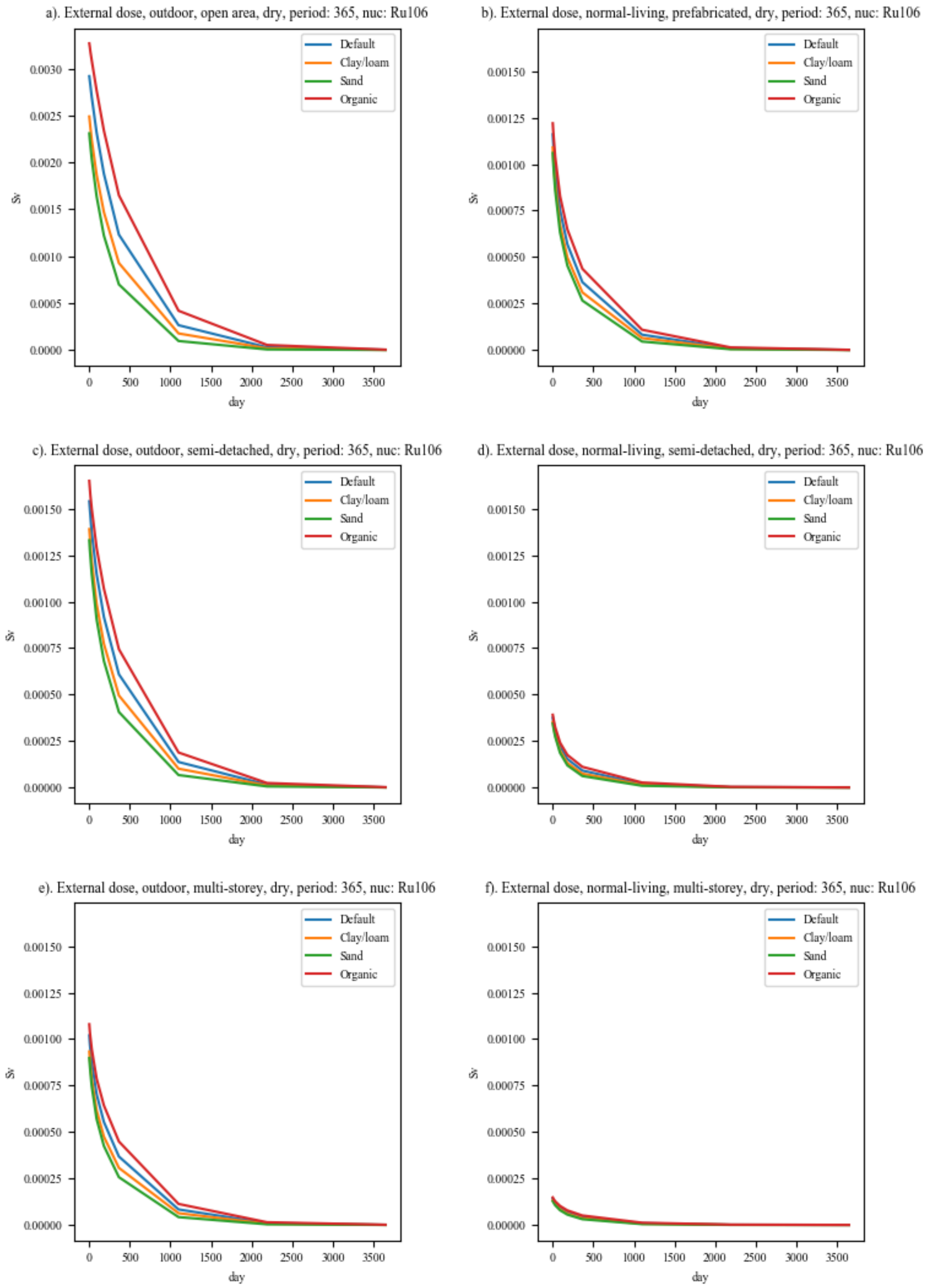

Figure 2.23. Predicted annual dose in various ERMIN environments with the soil migration model parameterised for Ruthenium for different soil types

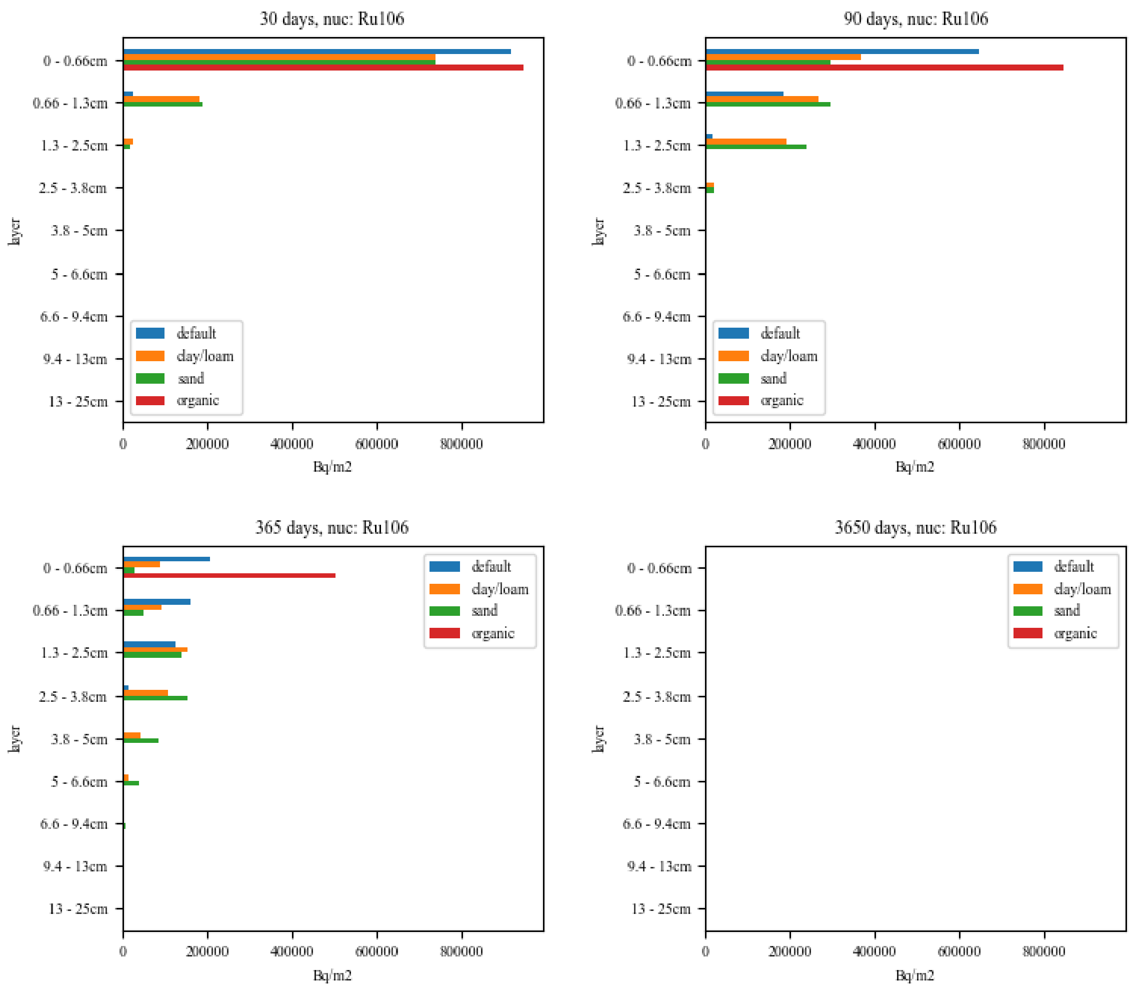

Figure 2.24 Predicted ^{106}Ru radioactivity in the soil profile at different times following deposition. NB the predictions are for the open area environment where there are no trees and therefore no transfer from trees via leaf-fall.

2.5.4.2 Retention on Roofs

While ERMIN has a single generic roof with a single set of parameters, Appendix 1 identifies several different roofing materials and gives parameters for the distributions of each. The parameter for which roof material really makes a difference is the initial retention on the surface after the first significant precipitation. Uncertainties are also given in Appendix 1 for the long term natural removal parameters.

2.5.4.3 Retention All Surfaces

Figure 2.25 and Figure 2.26 show the results of a Monte Carlo analysis of ERMIN in which the distributions of the retention parameters for each surface were sampled in turn, while keeping the parameters for other surfaces fixed at the default ERMIN value. In a final Monte Carlo analysis, the distributions of the parameters for all surfaces were sampled together assuming no correlation. For grass and small plants changing the retention on the leaves has little effect on the long term residual dose because the retention on the leaves is relatively quick compared to the subsequent migration down the soil column and therefore varying it makes little different to the predictions of residual dose as the radioactivity is still present at the soil surface long after it has moved from the leaves.

Therefore, for this analysis, the soil column parameters where also sampled using the distributions defined for ‘clay/loam’ as described in Section 2.5.4.1.

Deciduous trees are problematic in this analysis since the model of leaf fall in ERMIN is very simplistic; essentially it is considered at be an instantaneous event. The date of the accident is always well known and that means that there is a high level of certainty about whether there are leaves on the trees and a good level of certainty about when they might fall (albeit with a degree of modelling uncertainty ad ambiguity when representing protracted leaf fall as an instantaneous event). For this analysis the accident is assumed to take place in spring and the baseline run (see Section 2.5.1) assume the time to leaf fall is 158 days. In order to examine this source of uncertainty, a plausible but arbitrary distribution was applied to the time to leaf fall; it was assumed to be distributed uniformly between 138 and 178 days. Because the leaves remain on the surface below, it can be expected that the effect of the uncertainty surround leaf fall will not have a large impact on the total residual dose.

Following dry deposition (see Figure 2.25); in outdoor locations in the prefabricated environment and the semi- detached environment, grass appears to be the most significant source of retention parameter uncertainty. For the multi-storey environment with a relatively small area of grass; the parameter uncertainty both for retention on roads and grass are equally significant at the early times, whilst the parameter uncertainty for retention on grass and the underlying soil dominates later on.

Following dry deposition and assuming normal loving assumptions (see Figure 2.25); it is the retention on interior surfaces that dominates the parameter uncertainty in all environments at early times, with grass and roof surface showing more significance later.

Following wet deposition (see Figure 2.26); in outdoor locations in the prefabricated environment and the semi- detached environment, grass and the underlying soil exhibited the most significant retention parameter uncertainty. For the multi-storey environment with a relatively small area of grass, the parameter uncertainty for retention on roads dominates at the early times, whilst the parameter uncertainty for retention on grass and the underlying soil dominates later on.

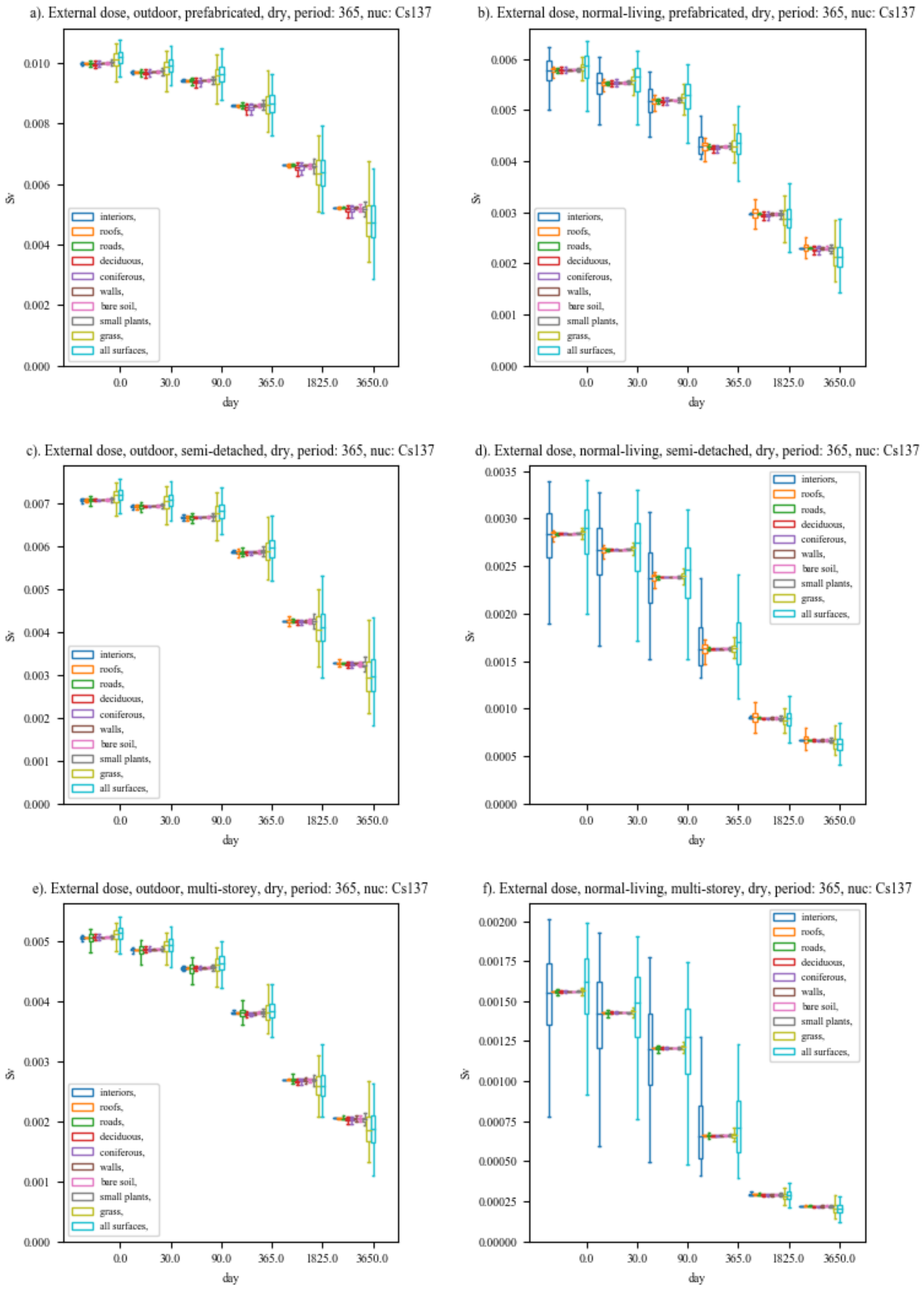

Figure 2.25. Box plots of a Monte Carlo analysis of predicted annual dose for various environments and locations following dry deposition from all surfaces. In each series the retention parameters for a particular surface was sampled whilst all other surfaces were kept at the default value. In the “All surfaces” series all retention parameters were sampled. The box for each marker indicates the inter quartile range (IQR), the range from quartile 1 (Q1) to quartile 3 (Q3), therefore 50% of results are contained in the box, the line is the 2nd quartile or the median. The ‘whiskers are defined as multiples of the IQR, the lower whisker is Q1 - 1.5IQR and the upper whisker is Q3 + 1.5IQR.

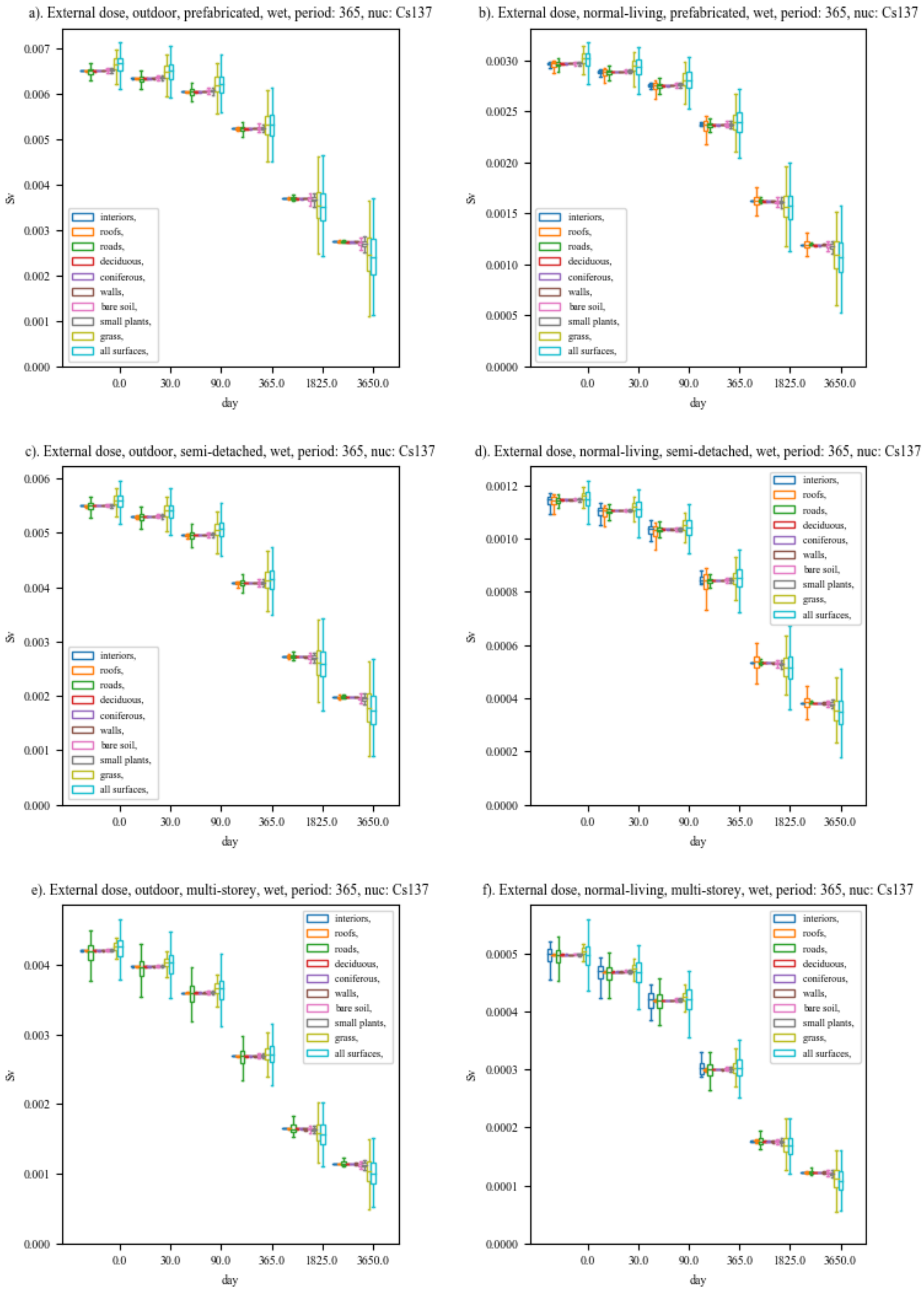

Figure 2.26. Box plots of a Monte Carlo analysis of predicted annual dose for various environments and locations following wet deposition from all surfaces. In each series the retention parameters for a particular surface was sampled whilst all other surfaces were kept at the default value. In the "All surfaces" series all retention parameters were sampled. (The box for each marker indicates the inter quartile range (IQR), the range from quartile 1 (Q1) to quartile 3 (Q3), therefore 50% of results are contained in the box, the line is the 2nd quartile or the median. The 'whiskers are defined as multiples of the IQR, the lower whisker is Q1 - 1.5IQR and the upper whisker is Q3 + 1.5IQR).

2.5.5 Aggregated stochastic and judgmental uncertainty

This section explores the combined effects of some of the judgemental and stochastic uncertainties discussed in previous sections.

For the illustration and lacking better occupancy data a plausible but arbitrary and simplistic occupancy distribution was assumed as given in Table 2.12. For initial deposition all the parameter uncertainties defined for the “all surfaces” series in Figure 2.13 were included. For retention all the parameter uncertainties defined for the “all surfaces” series in Figure 2.26 were included.

Table 2.12 Notional occupancy distribution for illustration

Proportion	Occupancy	Proportion	Occupancy
1%	0.5	25%	0.9
4%	0.75	9%	0.95
15%	0.8	1%	0.99
45%	0.87		

No correlations are considered and uncertainties missing from this analysis include:

- No uncertainty in initial reference surface deposition, radionuclide mix, or particle properties. Initial deposition to reference surface is dry and fixed at 10^6 Bqm^{-2} ^{137}Cs in a cationic form.
- Variations within an environment type, although different broad environments are considered.
- A dry situation was assumed but under wet conditions there would be additional uncertainty on the ratio of wet to dry; as indicated in Appendix 2.5.3.1, this impacts particularly on the interior ratio.

Figure 2.27 shows the boxplots from this analysis. Generally, it is the uncertainty on the initial deposition that is most significant most situations. Occupancy obviously is not a factor when considering only outdoor dose, but even under normal-living assumptions and notwithstanding that the occupancy distribution used for this analysis is simplified and notional; occupancy does not appear to be a significant source of uncertainty. Retention parameter uncertainty has most significance in the more shielded buildings and this is likely to be the uncertainty on the interior deposition as indicated by the discussion in Appendix 2.5.4.2.

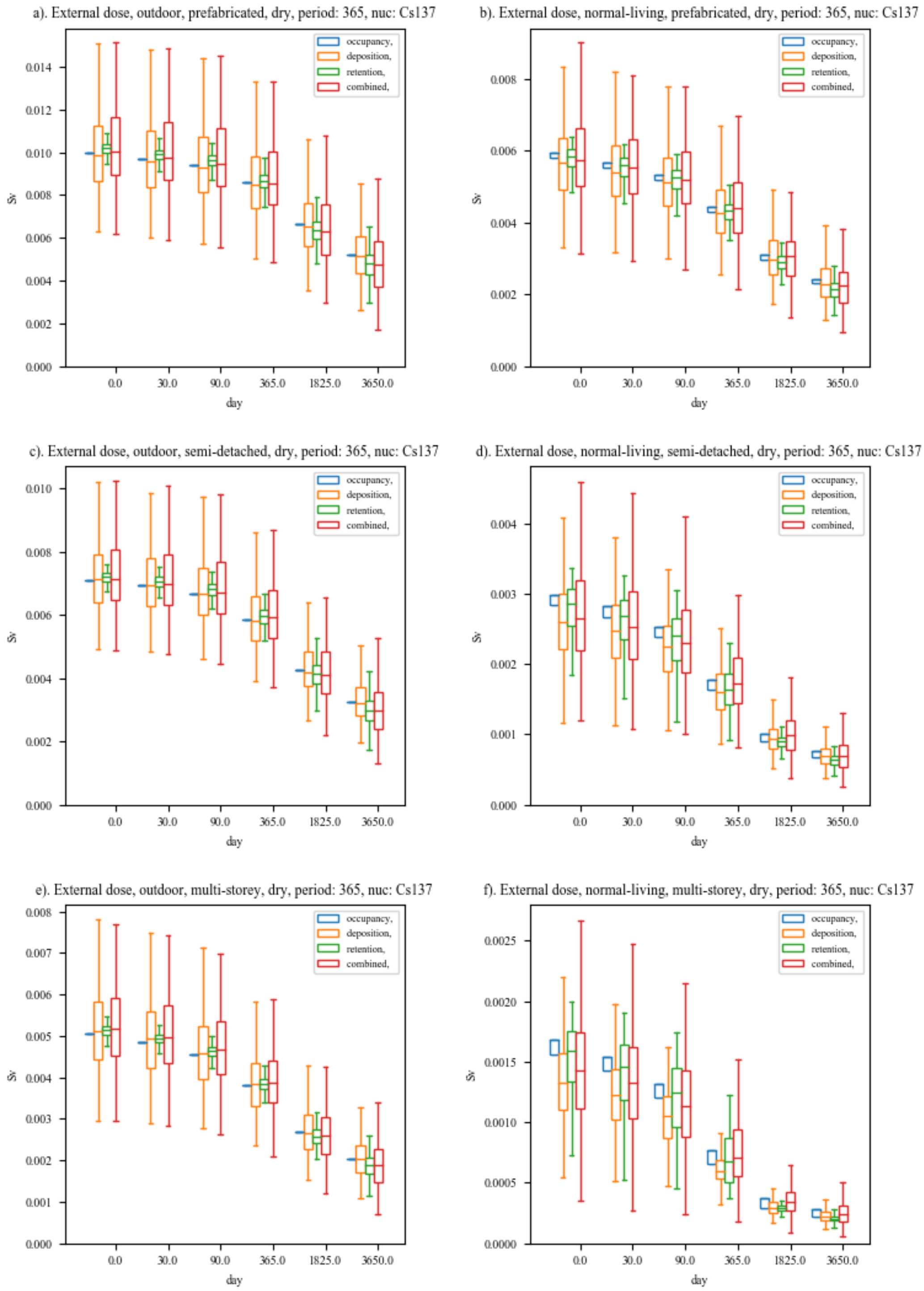

Figure 2.27. Box plots of a Monte Carlo analysis of predicted annual dose for various environments and locations following dry deposition from all surfaces. The “occupancy” series was generated by sampling a simplified notional distribution, the deposition series by sampling the distributions of all surface ratios, the “retention” series by sampling of all the retention parameters distributions, and the “combined” series was generated by sampling all the distributions for occupancy, initial deposition and retention. (The box for each marker indicates the inter quartile range (IQR), the range from quartile 1 (Q1) to quartile 3 (Q3), therefore 50% of results are contained in the box, the line is the 2nd quartile or the median. The ‘whiskers are defined as multiplies of the IQR, the lower whisker is Q1 - 1.5IQR and the upper whisker is Q3 + 1.5IQR).

The importance of the uncertainty in initial deposition is an important finding as in a real incident this is an uncertainty that could be reduced by direct monitoring of surfaces, particularly external hard surfaces such as paved, walls and roofs. Even on these surfaces, such monitoring is challenging as the ratios ERMIN requires represent average values and not point values that would obtained from monitoring; direct monitoring of trees or interior surfaces would be particularly difficult.

2.5.6 Management options

The analysis has so far not included any clean-up options. ERMIN represents different types of options including:

- removal of radioactivity from surfaces, e.g. brushing roofs or hosing roads,
- removal of surface with the radioactivity on it, e.g. removing turf,
- mixing of radioactivity within the soil profile, e.g. ploughing or digging soil,
- shielding radioactivity, adding a soil layer,
- tying material to a surface either temporally or permanently, and
- options that are a combination of the above, e.g. removing turf and adding soil.

Since it is not possible to consider all combinations of options, the analysis considers the options included in the HARMONE 'model'. In addition, the option "vacuum sweeping roads" was added to encompass the paved surface which is not included in the HARMONE strategies because of the low contribution from paved surface to total residual dose. The options are listed in Table 2.13.

Table 2.13 Management options considered

Surface	Option	Application start	HARMONE strategy	Alternative strategy
Roof	Roof brushing	Late	x	x
Internal surfaces	Vacuuming	Late	x	
Internal surfaces	Washing	Late		x
Grass	Grass cutting	Early	x	
Small plants	Plant removal	Early	x	
Grass and Small plants	Rotovating	Late	x	
Tree and shrub	Removal	Late	x	x
Grass	Top soil removal/replacement	Late		x
Paved roads	Vacuum sweeping	Early		

In ERMIN, the action of the different types of option is described by sets of parameters. For example, removal of radioactivity is described by particle group and element dependent decontamination factors (DF), as well as further parameters that describe how that DF varies with time following deposition, whereas a soil mixing technique is defined by a matrix that describes how layers of soil are redistributed into new layers.

While these option effectiveness parameters are subject to both stochastic and judgemental uncertainties, it is clear that many of the sources of uncertainty are the same as those already explored in the previous sections, in particular surface retention uncertainty. In addition, model uncertainty has a role; for example, in ERMIN options are assumed to be applied instantaneously whereas they will be applied at different places at different times.

The significance of these two sources of uncertainty on the uncertainty of the effectiveness of a clean-up option can be demonstrated by looking at grass-cutting. This can be done because in the ERMIN model grass cutting is modelled as a 'surface removal' technique, the surface being grass leaves, and the retention of radionuclide on that surface is modelled explicitly. For these kinds of techniques, ERMIN does not use a DF parameter from the database. This differs from the way other techniques such as roof-brushing are modelled; roof-brushing is a ‘radioactivity removal’ technique, i.e. radioactivity is removed from the surface but the surface is left in place. Whilst retention on the roof surface is explicitly modelled, the component of that radioactivity that is removed by brushing is not and instead a DF from the database is applied to represent its effectiveness.

Figure 2.28 shows a Monte Carlo analysis of ERMIN in which a DF value for grass cutting was calculated from grass surface contamination before and after grass cutting under dry conditions. In this analysis the only the parameters for retention on grass and a plausible but arbitrary distribution for time of application were sampled. The maximum DF of around 6 (which equates to approximate 83% removal of radioactivity) compares well with the quoted value of 10 (90% removal) given in the EURANOS handbook (Nisbet et al, 2010). The minimum value given in the handbook is 2 (50% removal), whereas the analysis predicts that in some combinations of retention and timing there is no effect, i.e. a DF of 1 (0% removal), but the handbook is assuming the technique will be applied under reasonably favourable circumstances. This suggests that most of the uncertainty for grass cutting is already accounted for in the retention and timing parameters.

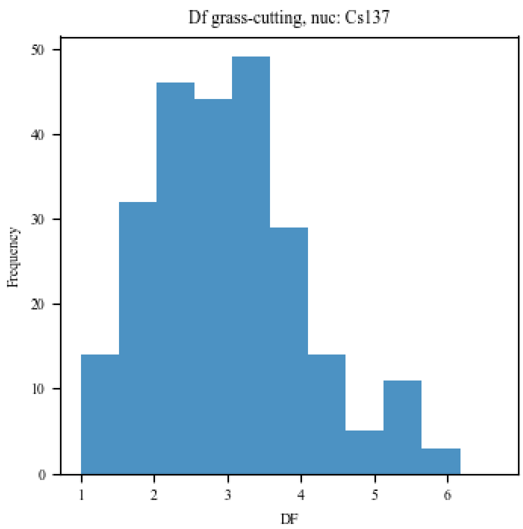

Figure 2.28 Frequency of calculated DF for grass cutting from Monte Carlo analysis in ERMIN in which the distributions of surface retention parameters for the grass and the soil below were repeated sampled along with a simple, arbitrary but plausible distribution of timing of application.

The EURANOS handbook also gives ranges for other techniques, for example, the DF range for Roof Brushing is stated as being between 2 and 7. However, in looking at the uncertainty on the projections of residual dose when options are applied, it may not be appropriate to incorporating such a range into the analysis along with the other sources of uncertainty (particularly retention parameter uncertainty) risks exaggerating the total uncertainty since, as demonstrated with grass-cutting much of the uncertainty in the quoted DFs may well derive from those other uncertainties.

Therefore, the subsequent analysis of all the individual options was undertaken on the basis that much of the uncertainty is captured by the timing and the retention parameters already explored. The uncertainty of start time was simulated with a plausible but arbitrary and simplistic binned distribution; for relatively simple options that can be applied early the time distribution is assumed to be: 25% applied at 4 days, 50% at 7 days and 25% at 10days. For more intensive options that will be applied later the time distribution is 25% at 2 weeks, 50% at 3 weeks and 25% at 4 weeks. ERMIN also assumes that an option will be successful; however, there are situations when an option can fail for example grass cannot be cut if it was already cut the day before, this source of uncertainty is not included.

Figure 2.29 shows the results of the uncertainty analysis looking at the residual dose reduction factor when options are applied separately. The sources of uncertainty are from occupancy, initial deposition, and retention parameters as included in the “combined” series in Figure 2.27. In addition, uncertainty on the application time has been simulated. Even without explicitly incorporating success as a variable, it is clear that the uncertainty on many options includes the possibility that they are ineffective. The biggest uncertainties when normal-living assumptions are applied are for the effectiveness of those options that apply to indoor surfaces; this is as expected because this is the surface from which the relative contribution has been found to be the most uncertain (see Appendix 2.5.3.3 and 2.5.4.2). The options with the least uncertainties tend to reflect the relative unimportance of the surface in contributing to total dose. Top soil removal and replacement is the most effective option in all environments when considering both outdoor dose and normal-living, however the uncertainty on its effectiveness is larger for normal living doses. This probably doesn’t reflect uncertainty in the absolutely contribution from the grass surface so much as the relative contribution of the grass surface which in turn will depend on how much or how little is contributed from internal surfaces.

Figure 2.30 shows boxplots of the predicted residual dose when individual clean-up options are applied as packages or strategies. Both strategies are very effective at reducing dose. The notable feature is that whilst when considering outdoor dose it is clear that the “Alt HARMONE” scenario is the more effective. This is probably due to the inclusion of “top soil removal and replacement”, an effective but intensive option that produces large quantities of waste. It is less clear if normal living assumptions are accounted for.

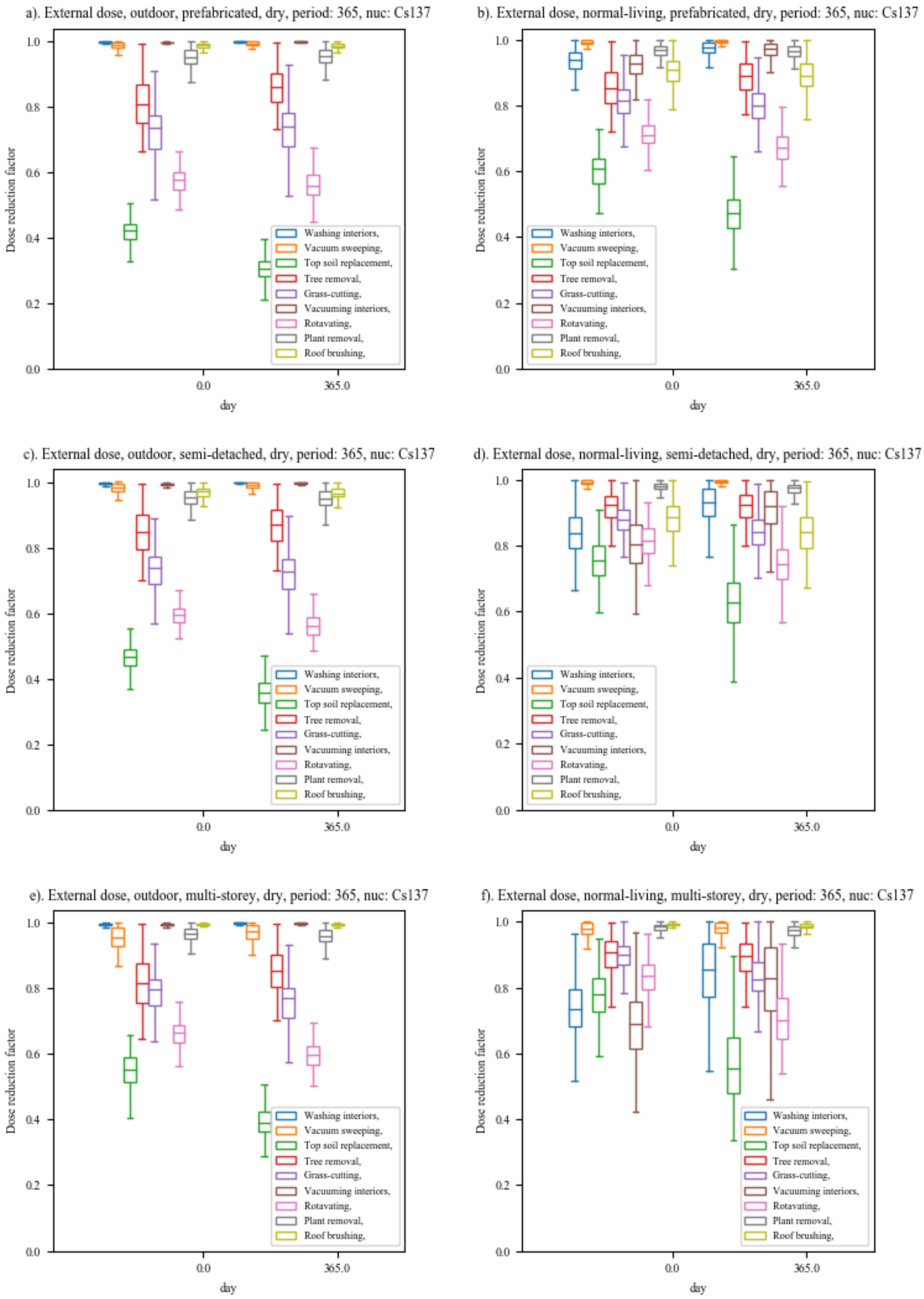

Figure 2.29. Box plots of a Monte Carlo analysis of predicted dose reduction factor on annual dose for various environments and locations following dry deposition from all surfaces. Each series represents an option applied using the default parameters from the ERMIN database and sampling model parameter and timing parameter distributions. The doses given are the 1st year dose staring from time 0 and the second year starting from time 365 days. (The box for each marker indicates the inter quartile range (IQR), the range from quartile 1 (Q1) to quartile 3 (Q3), therefore 50% of results are contained in the box, the line is the 2nd quartile or the median. The ‘whiskers are defined as multiples of the IQR, the lower whisker is Q1 - 1.5IQR and the upper whisker is Q3 + 1.5IQR).

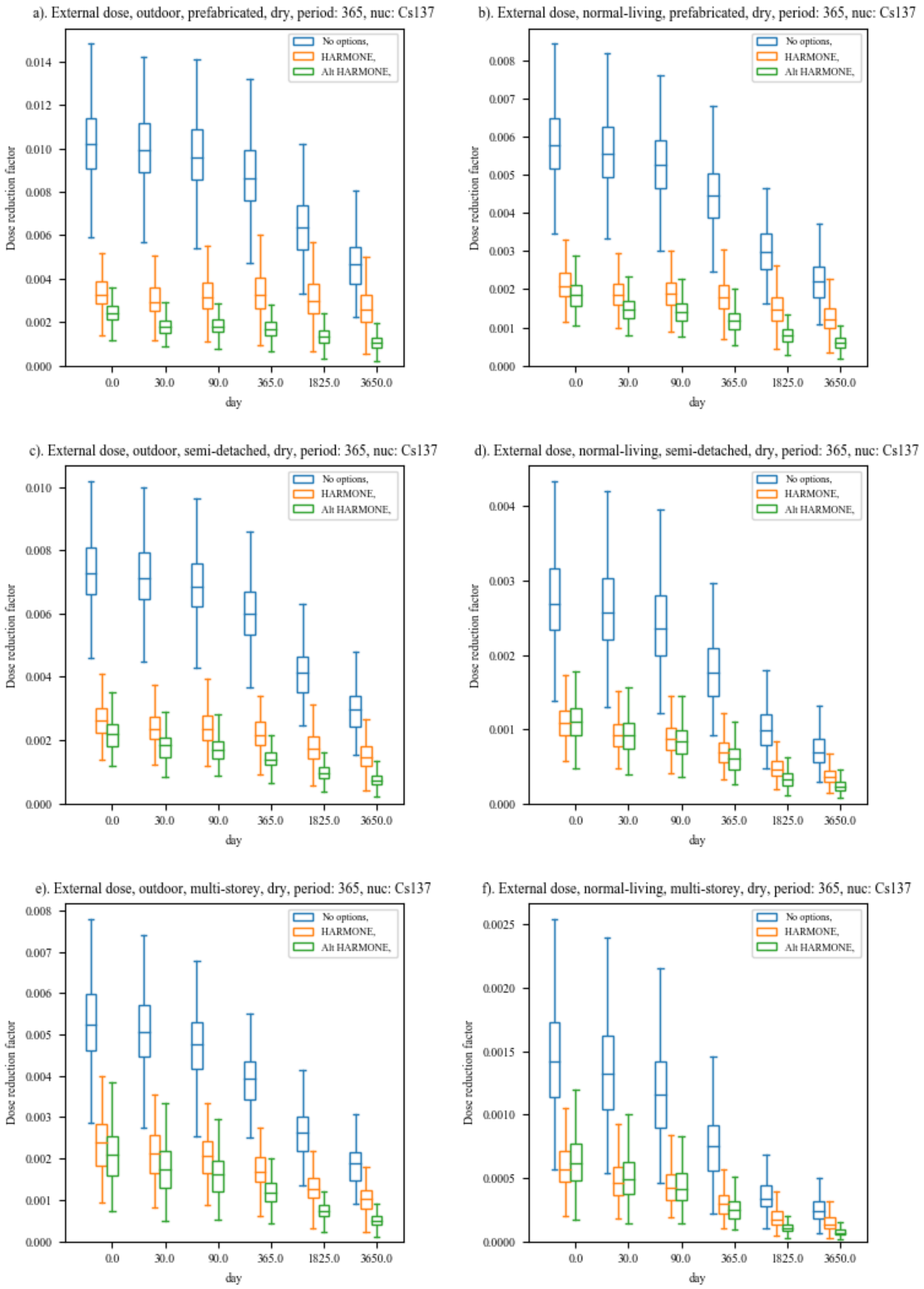

Figure 2.30. Box plots of a Monte Carlo analysis of predicted annual dose for various environments and locations following dry deposition with no options applied and with two possible strategies applied. (The box for each marker indicates the inter quartile range (IQR), the range from quartile 1 (Q1) to quartile 3 (Q3), therefore 50% of results are contained in the box, the line is the 2nd quartile or the median. The 'whiskers are defined as multiples of the IQR, the lower whisker is Q1 - 1.5IQR and the upper whisker is Q3 + 1.5IQR).

2.6 Conclusions

When considering stochastic and judgmental uncertainty, stochastic is arguably less important because people can be expected to move around. There are likely to be small areas of soil, certain roofs or building interiors which receive and retain radioactivity in greater amounts than the average parameters of ERMIN would suggest, similarly there will be surfaces that receive and retain much less. The analyses and plots presented above all contain the implicit assumption that the population does not move from these high and low dose-rate areas. Clearly, people do move and such stochastic variation is likely to be smoothed out. Judgemental uncertainty is therefore more important; the correct choice of the average parameters that ERMIN needs to predict residual doses that represent this smoothing. For example, incorporating the functionality to use different average soil migration parameters depending on soil type is arguably a worthwhile development for ERMIN, whereas a full uncertainty analysis that incorporates values from the extremes of the distributions for these soil type specific parameters may well just create spurious uncertainty.

With those caveats in mind, and also understanding that some sources of uncertainty have not been included, a tentative conclusion is that stochastic and judgemental uncertainty whilst in some situations may be significant gives uncertainty on residual dose predictions that is less and usually much less than an order of magnitude. The largest spread seen in Figure 2.27f encompassed a range of doses that varied by a factor of 8 or 9 without accounting smoothing; and this was dry deposition in a high shielded environment, a situation in which indoor deposition, identified as one of the most uncertain components in ERMIN, is likely to be a dominant contributor to dose. It therefore could be argued therefore that the fact the stochastic uncertainty of environment shielding properties could not be examined here is not a weakness, because what is important is not the stochastic variations within an environment but that the idealised environments represent a robust average for the set of real environments they might represents, unfortunately this is not something that is easy to demonstrate.

From this analysis the components of ERMIN that could be improved in order to reduce uncertainty are:

- For soil migration, allow different average migration parameters to be used for different soil types, and element specific parameters when the deposition is in soluble form (also in non-soluble form although this aspect was not explored in this analysis).
- Indoor deposition ratio has been identified as an important source of uncertainty. Further work would be useful to reduce the uncertainty on this parameter, with the aim of eventually allowing the user to pick between different classes of building for example. If uncertainty analysis was built into an operational version of ERMIN then it would be advantageous to fully implement the model of ingress into ERMIN to allow meaningful parameters such as ventilation, room height and indoor deposition to be directly folded into the analysis.
- It would be advantageous to have a wider range of idealised environments for the ERMIN user to choose from. For example, the multi-storey apartment block environment in ERMIN is often chosen to represent urban areas, in the absence of anything better. However, the presence and importance of grass and to a lesser extent trees in this environment demonstrates that it is still a somewhat suburban environment (see Figure 2.3b and Figure 2.4b).

A final question is should uncertainty analysis be included in an operational version of ERMIN? Before this can be done two other challenges must be overcome:

- How to present uncertainty to the user in a way that is useful, doesn't overstate or understate the uncertainty and acknowledges the sources of uncertainty not included in the analysis.
- How to quantify the uncertainty in the initial deposition to the reference surface so that it can be propagated through ERMIN. As indicated in Appendix 2 Table 2.1, initial deposition to the reference surface is expected to be one of the largest sources of uncertainty. However, it is unclear particularly at transition and particularly for very urban areas how large the uncertainties are and whether they are larger, comparable or smaller to the model subsequent uncertainties of ERMIN investigated here.

2.7 References

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