UK Health Security Agency

Verification and validation of models used in radiological assessment tools PACE and PC CREAM 08

Contents

Abstract

The Radiation Assessments Department (RAD) has responsibility to provide radiological protection advice to the UK Government and public and where necessary to carry out radiological impact assessments. An important area of work for RAD is the assessment of the radiological consequences of potential and planned releases of radioactive material into the environment. In support of its assessment capability, RAD has developed 2 important tools: PACE which is used to assess the consequences of an accidental release of radioactive material to the atmosphere, and PC-CREAM 08 which is used to assess the consequences of planned routine discharges to the atmosphere, rivers and seas.

These tools include a number of models that simulate the dispersion of radionuclides through the environment and their accumulation in the biosphere. The models are typically used for prospective assessments of future radiation exposures, but they can also be used for retrospective assessments to supplement measurements, for example, where measurements are incomplete or below limits of detection. An important consideration when using a model is to ensure it is fit for purpose and has been verified and validated.

This report briefly describes the models used in PACE and PC-CREAM 08 and summarises the verification and validation studies that have been undertaken.

Quality assurance

This work was undertaken under the Radiation Assessments Department’s Quality Management System, which has been approved by Lloyd's Register Quality Assurance to the Quality Management Standard ISO 9001:2015, Approval Number ISO 9001 - 00002655.

1. Introduction

An important aspect of the work of the Radiation Assessments Department (RAD) is the assessment of the radiological consequences of potential and planned releases of radioactive material into the environment. RAD has developed 2 software programs to assist with such assessments: PACE (Probabilistic Accident Consequences Evaluation) (22) which is used to assess the consequences of an accidental release of radioactive material to the atmosphere, and PC-CREAM 08 (Consequences of Releases to the Environment Assessment Methodology) (103) which is used to assess the consequences of planned routine discharges to the atmosphere, rivers and seas. These assessment tools incorporate models which can be used to simulate the dispersion of radioactivity in the environment. The models may be used for prospective assessments, for example, for calculating doses arising from future exposures, or for retrospective assessments when monitoring data are incomplete or below limits of detection. Models are used to predict the spatial and temporal distribution of radionuclides in the environment, their uptake by biota and the doses to humans that arise as a consequence of internal exposures from ingestion or inhalation and external exposure to radionuclides. Models are a mathematical representation of the real-world processes involved in the dispersion of radionuclides under different environmental conditions and it is important that they have been verified and validated to ensure they are fit for purpose.

Verification is the process of demonstrating that the mathematical model being used is a faithful implementation of the conceptual model. It should include tests to ensure that the mathematical equations involved have been solved correctly and that appropriate input data for model parameter values has been used.

Validation is the process of demonstrating that a model is an adequate representation of the real world. This is done by comparing model predictions with measurements and with other verified and validated models. In practice, model validation may not be straightforward because a lack of measurement data may mean that model validation is limited to certain environmental conditions or certain parts of the model. Nevertheless, the validation process is important and may identify the limits of applicability of the model which is extremely useful information for the user.

This report illustrates the models currently used in PACE and PC-CREAM 08, gives a brief description of their purpose and capabilities, and summarises the verification and validation studies that have been undertaken. The report focuses on studies carried out since the verification and validation review of the environmental models developed by RAD, which is described in report NRPB-R300 (98), but earlier studies are considered where appropriate.

2. The PACE software tool

PACE (Probabilistic Accident Consequence Evaluation) is a software tool for calculating the consequences of a short-term release of radionuclides to the atmosphere (19, 21). It is a level-3 Probabilistic Safety Analysis (PSA) or Probabilistic Risk Analysis (PRA) code. Level-1 PSA codes evaluate the probability of reactor core damage occurring, level-2 PSA codes evaluate the probability of containment failing, and level-3 PSA codes evaluate the probability of off-site consequences occurring following an atmospheric release after core damage and containment failure. Offsite consequences include individual doses from different exposure pathways, stochastic and deterministic health effects arising from the exposure, requirement for protective actions and economic costs arising from health effects and from disruption to the public, agriculture, industry, business and tourism.

PACE models the transfer of radionuclides through the environment, the subsequent dose distributions in the population, the impact of protective actions which might be introduced to reduce the doses, the health effects in the population and the economic costs of the health effects and protective actions. It operates by simulating the release under many different meteorological conditions and in this way estimates the ranges and percentiles of the endpoints, for example, the minimum and maximum distance of evacuation and percentiles of numbers of fatalities. The tool was developed under the ISO9001:2015 certificated quality management system operating in the Radiation Assessments Department and verification of the software package itself has focused primarily on extensive software testing and peer review while the environmental transfer models included in the programme have been subject to their own separate verification and validation as summarised below.

PACE incorporates 2 atmospheric dispersion models: a simple Gaussian plume model called Adept, described in section 4.1.2, and the more sophisticated NAME (Numerical Atmospheric-dispersion Modelling Environment) model which has been developed by the Met Office. NAME is a Lagrangian particle-trajectory model designed to predict the atmospheric dispersion and deposition of gases and particulates. A description of this model and details of its verification and validation are given in section 4.2.

Predicted activity concentrations in air are used to calculate doses from inhalation of the dispersing plume and external exposure to gamma radiation from the plume. Predicted deposition of material onto the ground is used as an input to the calculation of dose from ingestion of foods, exposure to gamma emitters deposited on the ground and inhalation of resuspended material.

Transfer of deposited radionuclides through the terrestrial food chain is estimated using data sets from the FARMLAND (Food Activity from Radionuclide Movement on LAND) food chain model (section 5.1) for spike releases occurring in January and June.

Exposure to gamma emitters from material deposited on the ground, vegetation and buildings is estimated using a dose factor library originally developed for the COSYMA level-3 PSA code (72). The data set is a compilation of information from different sources, and includes doses calculated using 3 different models. The values for the radionuclides which typically make the largest contributions to deposited external gamma dose from accidental releases from nuclear fission reactors (^{103}Ru, ^{106}Ru, ^{131}I, ^{132}Te, ^{134}Cs, ^{137}Cs, and ^{140}Ba) were calculated with the EXPURT (Exposure from Urban Radionuclide Transfer) model (section 6.3), which is a relatively sophisticated model that explicitly represents weathering on, and exposure from, a range of surfaces found in residential areas. Dose factors for other radionuclides were calculated using a simpler model which assumes that the dose in the area where people live can be represented by that over an open field (18, 41). The doses are calculated allowing for material to migrate down the soil column. This approximate description, together with a suitable shielding factor, gives an adequate estimate of the dose from these radionuclides which are less important for this application (13). The data set also includes dose factors for radionuclides which are likely to be important in releases from accidents at fusion reactors; the doses for these radionuclides were derived from a model developed by the Institut für Strahlenschutz, Gesellschaft für Strahlen- und Umweltforschung (57).

The long-term integrated radioactivity concentrations in air from resuspension of deposited radioactivity are estimated using the resuspension factor approach recommended in report PHE-CRCE-047 (109) and described in section 6.5.

PACE uses factors to estimate the reduction of doses from deposition and resuspension achieved by clean-up operations; the factors were calculated using the European Inhabited Area (ERMIN) model described in section 6.2. Finally, the economic consequences of an accidental release are modelled using Cost of Consequences Offsite-2 (COCO-2) model (see section 9).

3. The PC-CREAM 08 software tool

PC-CREAM 08 (Consequences of Releases to the Environment Assessment Methodology) is a software tool that comprises a suite of models and data for performing radiological impact assessments of routine, continuous discharges of radioactivity to the environment (103). The program can calculate both individual and collective doses from discharges to atmosphere, sea or rivers depending on the location of the site. PC-CREAM 08 is composed of a main assessment application, called Assessor, which combines activity concentrations in food and environmental media with habit data and dose coefficients to calculate doses. It also includes a number of supporting models which allow the user to calculate environmental activity concentrations should the dispersion conditions change from the defaults and use these as input to the main assessment application if required. PC-CREAM 08 was developed under the ISO9001:2015 certificated quality management system to which RAD is accredited and verification has focused primarily on extensive software testing, comparison with the previous version and peer review. The environmental transfer models included in the suite have been subject to their own separate verification and validation as discussed in later sections.

The calculation of the dispersion of radionuclides in the atmosphere is performed using an implementation of the R91 model (24), called Plume, appropriate for continuous releases. Model outputs include activity concentrations in air, external gamma dose rates from the plume and deposition rates onto the ground. A description of this model and details of its validation are given in sections 4.1.1 and 4.1.3.4, respectively. The transfer of deposited radionuclides through the food chain is modelled using FARMLAND, see section 5.1, and the output values are used to calculate ingestion doses. In addition, calculations of exposures from radionuclides deposited on the ground and their subsequent resuspension are carried out using the models GRANIS (Gamma Radiation Above Nuclides In Soil) (section 6.1) and a resuspension model called RESUS which is based on the model described in section 6.5.

Models are also included in PC CREAM 08 to simulate the dispersion of radioactivity in both fresh and sea water bodies. These models can be used to predict activity concentration in river and seawater, suspended and bed sediments, seafood and freshwater fish. A description of these models and details of their validation are given in section 7.1 and section 7.2.

Finally, the models used to calculate collective doses from global circulation of certain radionuclides are briefly described in section 8, although no new verification or validation studies of these models have been carried out since the previous model verification and validation review (98).

4. Atmospheric dispersion and deposition

4.1 Implementations of the R91 model

The R91 model is so called because it refers to the Gaussian plume diffusion model described in the first report of the UK Atmospheric Dispersion Modelling Working Group, NRPB-R91 (24). This report contains the basic formulation of the model and its application, while subsequent reports describe how effects such as plume depletion, plume rise, the impact of buildings and coastal effects could be taken into account (64, 65, 66, 67, 68, 70). The R91 model has been implemented in several programs developed by the Radiation Assessments Department, such as Plume and Adept.

4.1.1 Plume

The Plume model included in the PC CREAM 08 assessment system (103) is designed to predict the dispersion of continuous releases of radionuclides to the atmosphere. It takes into account the meteorological conditions at a single location (typically in the vicinity of the location of a release), and models processes such as radioactive decay, and wet and dry deposition to calculate activity concentrations in air, deposition rates and doses from external exposure to gamma rays from the plume. Deposition is described by means of a source depletion model (65). Dry deposition is modelled using deposition velocities, while wet deposition is modelled using washout coefficients (1, 67). As a basic formulation of the Gaussian plume model the R91 model is applicable only when the atmospheric stability can be assumed to remain constant during the release period. This assumption is not valid when calculating annual average activity concentrations and therefore Plume was modified according to the recommendations given in NRPB-R91 (24) on how to model the dispersion when the meteorological conditions change. Briefly, the continuous release is assumed to occur during different but constant meteorological conditions and the frequencies with which these meteorological conditions occur during the year are used to evaluate the distribution of activity in sectors around the release point, while it is assumed that the horizontal distribution of activity is uniform over each sector. In Plume, meteorological conditions are allocated to one of 6 Pasquill Stability Categories (A to F). Modelling assumptions for Plume are summarised in Table 1.

4.1.2 Adept

The main features of the Adept model included in the PACE accident consequence assessment tool (19) are the same as those of Plume, although Adept accounts for meteorology changing as a function of time. This is achieved in a simple way by assuming that the single site meteorology affecting radioactive material released in a single hour persists for the entire duration of the atmospheric dispersion of the radioactive material, while radioactive material released in adjacent hourly periods are associated with a different set of meteorological conditions. Modelling assumptions for Adept are summarised in Table 1.

Table 1. Summary of modelling assumptions for Plume and Adept

Feature	Plume	Adept
Plume rise	Only considers release height but an effective release height can be applied. The effective release height remains constant throughout a single model run considering a range of meteorological conditions.	Only considers release height but an effective release height can be applied.
Particle size distribution	Single particle size of 1 μm or less and gas.	Single particle size of 1 μm or less and gas.
Description of the atmospheric stability	Three schemes are available to categorise stability: Doury, Hosker-Smith (A to F) or Pasquill Stability Category (PSC) (A to F). The boundary layer height assumed for each stability category can be modified.	Pasquill Stability Category (PSC), categories A to F.
Wind speed	Wind speed is implicit in the stability scheme selected but the default value can be altered (that is, for PSC D a default wind speed of 5 m s^{-1} is assumed but this can be altered).	Wind speed is considered independently of the PSC.
Wind direction	Wind direction is considered in terms of the frequency with which the wind blows into different (sectors) directions over the course of a year. The number of sectors, and hence wind directions, around the site are typically 8, 12 or 18.	Either a single wind direction can be considered over the entire release, or a time varying wind direction can be considered (as described in the main text).
Description of the precipitation	Rainfall is only assumed to occur during PSC C and D or the 3 Doury normal categories and then for just a fraction, typically 10%, of the time that these categories occur. The rainfall rate is assumed to be 1 mm h^{-1}.	Considered in units of mm h^{-1}.

Feature	Plume	Adept
Temporal description of the meteorology	Modelled as a set of independent continuous releases, each occurring during constant meteorological conditions. The frequencies with which these meteorological conditions occur in different wind directions are used to evaluate the distribution of activity in sectors that surround the release point.	Meteorology varies hourly, but the meteorological conditions affecting the dispersion of any one hour release remain constant for the entire duration of the dispersion of that component of the plume.
Spatial description of the meteorology	Single definition of meteorology applies to the entire spatial domain.	Single definition of meteorology applies to the entire spatial domain.
Surface roughness	Six different values can be selected ranging from 0.01 m to 4 m.	A single value of 0.3 m is assumed.
Dry deposition	Dry deposition velocities applied.	Dry deposition velocities applied.
Wet deposition	A default value of the washout coefficient of 1 10^{-4} s^{-1} is included to be consistent with a rainfall rate of 1 mm h^{-1} and for 1 μm particles. The washout coefficient can be modified.	Washout coefficients as a function of rainfall rate applied. Washout coefficients can be modified, including application of a specific value for elemental iodine.
Plume depletion	Yes, by way of a modified source strength.	Yes, by way of a modified source strength.
Radioactive decay during atmospheric dispersion.	Yes.	No.

4.1.3 Verification and validation

Significant verification of the atmospheric dispersion models based on the R91 model has been performed by the Radiation Assessments Department to check that the implementation of the models is in agreement with their description in the methodology documentation. The work to verify these models was carried out under RAD's formal Quality Management System and as such was subject to careful quality assurance in terms of version control and testing. Although comprehensively documented, the verification tests were in general carried out as part of a number of different projects and the results are not publicly available.

Verification and validation of Gaussian plume models in general and of the different implementations of the R91 model in Plume and Adept are summarised below.

4.1.3.1 Gaussian models

Validation of Gaussian plume models was considered by (26) and subsequently by the Working Group on Atmospheric Dispersion Modelling (ADMWG) (69) in terms of the uncertainty associated with model predictions. These studies concluded that annual average activity concentrations in air predicted by Gaussian plume models within a few kilometres of the site and in flat terrain are generally within a factor of 2 of observed values (or measurements). The uncertainty increases with increasing distance from the site (10 to 100 km) and when reducing the averaging period from annual to monthly or seasonal, and in such situations activity concentrations are likely to be predicted within a factor of 4.

4.1.3.2 The R91 model

A comparison of an application of the R91 model with the UK Met Office's NAME model (see section 4.2) in the context of an accidental release scenario is described in report HPA-CRCE-029 (7). For a short duration release in neutral atmospheric stability conditions, a relatively low release height and flat terrain, disparities (in time integrated activity concentrations in air) of approximately a factor of 3 at a few kilometres from the release and a factor of 2 at a few tens of kilometres from the release were observed. However, for unstable and stable conditions, low wind speeds and large release heights the disparities were significantly larger. The descriptions of the cross-wind and vertical spread of the plume and the wind-driven advection of the plume were identified as the primary causes of the observed differences between R91 and NAME model output.

Intercomparisons of the R91 and ADMS (a Gaussian plume model developed by Cambridge Environmental Research Consultants) models are described in a number of publications (17, 42, 43, 73). Hill and others (1999) indicated very little difference between the predictions of the R91 and ADMS models when compared to field measurements in predominantly neutral atmospheric stability conditions. Hill and others (2004) compared R91 model predictions with hourly and daily ^{85}Kr time averaged air concentration measurements (collated over almost a 4-year period) from routine releases at BNFL Sellafield. In the field, release durations and distances between the release point and the receptors were variable and the terrain was moderately undulating. They reported that approximately 75% of observations were within a

factor of 10, approximately 50% of observations were within a factor of 5 and approximately 30% of observations were within a factor of 2 of model results. Carruthers and others (1996) focused on the validation of the ADMS model but in the process also assessed the relative performance of the R91 model. Model validation was performed against light detection and ranging (LIDAR) data for isolated stacks in flat terrain, across a range of neutral and unstable atmospheric stability conditions, for receptors between a few hundred metres to 3 kilometres downwind, stack heights ranging from 120 to 260 metres, a surface roughness of 0.2 and 0.5 metres, and accounting for plume rise. This study substantiated the conclusions of the HPA-CRCE-029 report (7) that the R91 model under-predicts the standard deviations of the cross-wind and vertical Gaussian plume profiles (σ_y and σ_z, respectively). Carruthers and others (1996) found that for releases with differing site, source and emission characteristics, values of σ_y and σ_z estimated by the R91 model were within a factor of 1.6 and 2.2 of observations, respectively. Jones and others (1995) found that the relative difference between R91 and ADMS model estimated time integrated activity concentrations in air varied by approximately a factor of 3.0 and 1.2 at 1 km and 40 km downwind, respectively, that is, consistent with the differences observed by Bedwell and others (2011) in a comparison of R91 versus NAME. Jones and others (1995) assumed Pasquill stability category D conditions, a roughness length of 0.3 m, a ground level release, a 30-minute release duration and a source diameter of 1 metre.

Hall and others (2000) contains a review of dispersion model intercomparison studies using ISC (a historic atmospheric dispersion model developed by the US Environmental Protection Agency), the R91 model, AERMOD (an atmospheric dispersion model developed by the US Environmental Protection Agency more recently than ISC) and ADMS. One of those studies (86) compared models analogous to the R91 model with ISC models, which are nominally based on the same fundamental dispersion characteristics, and identified agreement in calculated air concentrations within a factor of 2.

The Working Group on Atmospheric Dispersion Modelling (ADMWG) produced a report (69) summarising the applicability of the R91 model and validation work for Gaussian plume models in general. Barker (1979) reported that maximum air concentrations from a release of approximately one hour's duration can generally be predicted to within a factor of 2 or 3 of observed values; whether the model over or underpredicted depended on the atmospheric stability conditions. Crawford (1978) concluded that over flat terrain, steady meteorological conditions, and within 10 km of the release point, the hourly air concentrations from accidental releases are likely to be predicted within a factor of 10. Miller and Hively (1987), in a review of validation studies for Gaussian plume atmospheric dispersion models, provided ratios of 2 to 4 of predicted to observed annual average air concentrations, applicable to a range of release heights. Kretzschmar and others (1984) performed an intercomparison of Gaussian plume models and noted that, for short-term releases, ground level activity concentrations in air estimated by the R91 model were intermediate in terms of the ensemble of twelve combinations of model and atmospheric stability schemes. Furthermore, numerous validation studies have been performed on the generic Gaussian plume modelling approach using the Kincaid and Prairie Grass tracer experiments, for example Korsakissok and Mallet (2009).

4.1.3.3 Adept

In-house model verification has been performed on the R91 modelling approach implemented in Adept, but none of this work has been published. As part of the quality assurance for PACE, and therefore the Adept model implemented within PACE, automated testing procedures have been developed. Thus, when changes are made to the model code, comprehensive checks can be performed to ensure no inadvertent effects (resulting from such changes) have been introduced.

4.1.3.4 Plume

NRPB-R300 (98) references work carried out to evaluate the risks of leukaemia and other cancers in Seascale (101). The annual average activity concentrations in air and annual deposits of radionuclides in Seascale from all sources (including natural, medical, weapons testing fallout and radioactive discharges) were calculated using the ESCLOUD model, of which Plume is essentially a simplified PC version, and compared with observed values. The predicted values were generally higher than the observed values by a factor of 2, with the greatest difference being a factor of approximately 5.

A comparison of predictions from Plume and the NAME model (see section 4.2) was carried out by Lutman and others (2004) for a long duration release. For the scenarios considered, differences between the 2 models were found to be generally small, typically within a factor of 2 for activity concentrations in air and a factor of 5 for deposited activity concentrations, compared to the expected precision of the models.

Comprehensive Plume model verification has been performed in-house, including comparisons between the versions of the model implemented in PC CREAM 98 and PC-CREAM 08. These results compared favourably, with only very minor differences being identified, mainly due to the use of different software compilers.

4.2 NAME

4.2.1 Model description

The Numerical Atmospheric-dispersion Modelling Environment (NAME) is a Lagrangian particle-trajectory model designed to predict the atmospheric dispersion and deposition of gases and particulates (63). The model was originally developed in 1986. Version 6.5 of the NAME model was implemented in version 3.3.4 of PACE, however, NAME and PACE model versions are continually evolving. The mean flow or advection of a particle is determined by the flow information, primarily the wind velocity, detailed in the required meteorological data. Diffusion is described by random walk (Monte Carlo) processes, determined by the turbulent velocity. Each model particle carries a mass or activity of one or more pollutant species and evolves by various physical and chemical processes during its lifespan. A box-averaging scheme is used to derive activity concentrations in air from particle activities. The dry deposition scheme in NAME uses a deposition velocity, whereby the flux of a pollutant to the ground is proportional to the

concentration in air and deposition velocity. The wet deposition scheme in NAME uses scavenging coefficients that are a function of precipitation rate, type of precipitation and type of deposition process. For radiological releases, NAME incorporates radioactive decay and also provides estimates of external dose from the radioactive plume (6).

4.2.2 Verification and validation

Sections 4.2.2.1, 4.2.2.2 and 4.2.2.3 focus on NAME model validation, whilst section 4.2.2.4 discusses NAME model verification.

4.2.2.1 Atmospheric dispersion modelling for radiological scenarios

The NAME model was validated in a number of studies against measurements taken following nuclear accidents.

NAME was one of 5 atmospheric dispersion models used in a study (30) that compared model estimates of activity concentration in air and activity deposited on the ground with field measurements taken following the accident at the Fukushima Daiichi nuclear power plant. Time series of activity concentrations in air for ^{137}Cs and ^{131}I at a single location approximately 110 km from the plant were considered. When compared using statistical performance measures, NAME performed as well as, if not slightly better than, most of the other models. The NAME model successfully captured the timing and magnitude of most of the peak passages of the plume but struggled to suitably describe the smaller activity concentrations in air in the periods between the peaks, possibly as a result of not accounting for resuspension. Measurements of deposited activity concentrations of ^{137}Cs were significantly more comprehensive in number and geographical coverage. NAME outperformed all models in terms of the statistical metric fractional bias; specifically, relative to the cumulative magnitude of the observed and predicted deposited activity concentrations, the difference between the observed and predicted deposited activity concentrations was relatively small. These smaller over- and under-predictions compared to other models, as illustrated by the fractional bias, were at least in part a result of the meteorological analyses (that is, the way meteorological fields are processed and the way atmospheric processes are parameterized within the atmospheric dispersion model), which were generally better resolved in NAME. The fractional bias, irrespective of the (4 types of) meteorology considered, was always found to be positive in value, indicative of the model underestimating the respective observations. Whilst NAME performed strongly in respect of fractional bias, NAME performed less well in respect of the other 4 statistical measures considered: the correlation coefficient, the figure-of-merit in space, the Kolmogorov-Smirnov parameter, and the normalised mean square error. Hence, for estimates of deposited activity concentrations, relative to other models, NAME ranked midway. Leadbetter and others (2015) expanded upon this work, noting that NAME model predictions of deposited ^{137}Cs activity concentrations showed mixed agreement with observations across eastern Japan, with correlation coefficients ranging from 0.44 to 0.80 (with a perfect linear relationship represented by the value 1).

Nelson and others (2006) used NAME to calculate best estimates of time-averaged and time-integrated air concentrations of radioactivity attributed to releases following an accident involving a fire in a nuclear reactor at Windscale Works, Sellafield, in October 1957. NAME successfully recreated the general movement of the radioactive plume and was in broad agreement with previous analyses. Discrepancies between modelled and measured plume arrival time and activity concentrations in air for certain locations were primarily associated with errors in the calculation of the wind fields used by the model. Johnson and others (2007) reduced the uncertainty in the plume's behaviour by using the latest available Numerical Weather Prediction Model reanalysis of meteorological data from the European Centre for Medium Range Weather Forecasts (ERA-40) coupled with current best estimates of the radioactive emissions profile. In general, an improvement in model comparisons against measurements was found. More specifically there was good agreement with the timing of plume arrival at locations in the UK, however the modelled times of arrival at mainland European sites were later than the actual timings. There was also close agreement between observed and predicted activity concentrations in air of ^{131}I, especially at the UK monitoring stations. Observed errors were thought to be due to model inaccuracies and uncertainty in the input data (notably errors in the meteorological fields and source profile).

4.2.2.2 Atmospheric dispersion modelling for non-radiological scenarios

The most comprehensive NAME validation study to date has been against the Kincaid data set, using a process based on the Model Validation Kit methodology developed under the Harmonisation initiative (62). The Kincaid data set contains measurements of ground level air concentrations from an elevated, buoyant plume release from the Kincaid power plant in the US (61). The validation study used the puff scheme in preference to particle scheme, as this is designed to be used for short-range applications. This study demonstrated that the performance of NAME was comparable with other leading short-range atmospheric dispersion models (for example, ADMS). The results showed a small over-prediction in the mean (measured) air concentrations, but the spread in the predicted concentrations was in good agreement with the observed spread.

NAME has also been validated against the European Tracer Experiment (ETEX). The experiment aimed to produce more reliable long-range data (notably air concentrations at 150 locations over a period of 60 hours) to validate atmospheric dispersion models. NAME successfully predicted the overall spread and timing of the plume arrival across Europe, but over predicted the observed air concentrations (96). This was in common with most other models which have been compared to the ETEX data set but was in contrast to other NAME validation studies (comparing NAME with other data sets), indicating either no significant bias or a tendency to under predict air concentrations.

Non-radiological case studies have also been regularly performed for comparison against observational data, notably following volcanic eruptions. Devenish and others (2012), Cooke and others (2014), Webster and others (2012), Heard and others (2012), Johnson and others (2012) and Marenco and others (2011) all made some form of comparison between NAME model estimates and observations generated using ground based LIDAR and sun photometers,

infrared satellite technology and research aircraft instrumentation, primarily following the eruptions in Iceland of Eyjafjallajökull in 2010 and Grímsvötn in 2011. The studies showed a number of parallels with radiological incidents in terms of modelling difficulties, such as complexity in determining a representative source term, including particle size distribution.

Further modelling parallels (and challenges) include a tendency for significant vertical spread of the release and dispersion over relatively large distances. Heard and others (2012) reported that NAME results compared favourably with available observations in terms of both geographical distribution and magnitude for all 3 cases (eruptions of Sarychev, Kasatochi and Eyjafjallajökull over the period 2008 to 2010) considered. In addition, Marenco and others (2011) concluded that ash prediction maps developed using output from NAME showed reasonable agreement with the overall magnitude of the observed concentrations, but noted that in some cases there were positional errors in the predicted plume location, due to uncertainties in the eruption source details, driving meteorology, and in the model itself.

Webster and others (2012) reported that predicted peak ash concentrations lay within about one and a half orders of magnitude of the observed peak concentrations. However a significant improvement in the agreement between modelled and observed values was seen if a buffer zone, accounting for positional errors in the predicted ash cloud, was used. Cooke and others (2014) noted that NAME effectively modelled the vertically sheared spread of the plume. Air concentrations estimated using NAME have also been compared with satellite imagery of the Etna (47) and Hekla (48) eruptions in 2002 and 2000, respectively, and these comparisons demonstrated a representative positioning of the plume as a function of space and time.

Webster and Thomson (2002) describe a plume rise scheme, based upon conservation equations of mass, momentum and heat, implemented in NAME. The performance of the scheme was assessed against data from the Kincaid field experiment, where NAME was found to be comparable with other models (including ADMS). It was also noted that the updated scheme added value to the model and significantly outperformed the previous plume rise scheme. Webster and others (2006) evaluated NAME model runs against data from the Buncefield Oil Depot incident. Estimates of the heat release rates from the fire were used as input to NAME's plume rise scheme. Comparisons with satellite images showed that NAME did not accurately capture the extent of the vertical spread of the plume, primarily due to insufficient plume rise. Potential reasons included the complex nature of the source, given that in NAME a single simple plume was modelled; lofting of the plume from absorption of solar radiation by the black carbon and the release of latent heat from the condensation of water vapour, both of which were not taken into account by NAME's plume rise scheme; and potential inaccuracies in the input meteorology, in particular the atmospheric temperature profile.

Hort (2004) performed NAME runs to investigate the behaviour of the deposition routines. Mesoscale meteorological data was used with a single 6-hour point release of sulphur dioxide over the South East of England, and a tracer release, in the same location. It was concluded from these limited runs that the NAME deposition schemes were working at least as well as previous versions of the model.

Hort and Athanassiadou (2005) investigated the effect of varying the spatial resolution of Numerical Weather Prediction (NWP) data on the predictions of NAME. Three resolutions were considered: global (60 km), mesoscale (12 km), and high resolution (4 km). The investigation found that increasing the resolution of the NWP data affected the flow and dispersion in a number of ways. Firstly, a significantly different plume direction and path was observed in the local flow. It was also found that a higher resolution of NWP data led to widening of the time integrated air concentration plots and enabled 'hot spots', often caused by local topographic effects, to be resolved. When considering wet deposition, Hort and Athanassiadou (2005) reported that the highest resolution allowed much more structure to be observed in the NWP precipitation fields, while the lower resolution simulations spread out the precipitation. Consequently, the area over which deposition is predicted to occur by the NAME model was much larger when using lower-resolution NWP data. Davis and Dacre (2009) used NAME to simulate a tracer release from the ETEX experiment and found that increasing the temporal and spatial resolution of the meteorological input led to improvements in the model predictions of the plume location and concentrations, particularly for the period 24 to 48 hours after the start of the release.

4.2.2.3 Cloud gamma modelling

A validation study on the method for the calculation of cloud gamma dose rates implemented in NAME (version 5.3) was performed by UKHSA. The NAME model was validated against measurements from a field experiment that took place at the SCK-CEN site in Mol. Details of the experiment can be found in Drews and others (2002). Because the detectors in the field experiment were all within 1.5 km of the release location, only the Lagrangian particle cloud-gamma calculation method implemented in NAME, and not the semi-infinite cloud approximation method, was considered. Single measurements of photon fluence rate (m^{-2} s^{-1}) from a release of ^{41}Ar were collated over the period of a few minutes, resulting in a data set of measurements over the period of a few hours (obtained from RISØ). Three types of meteorological data were considered: Numerical Weather Prediction data provided by the UK Met Office, single site meteorological data for 2 sites near to Mol (Belgium) provided by the UK Met Office, and single site meteorological data for the SCK-CEN site in Mol and provided by RISØ. Of the 75 sets of model results (scoping a range of release times and dates, detector types, detector locations and meteorological data types) the total adult (the only age group considered in the model) effective cloud gamma dose estimated by NAME were found to be within a factor of 2 of the respective observed values (derived from the photon fluence measurements) in 48 cases (64%), within a factor of 3 in 67 cases (89%) and within a factor of 5 in 73 cases (97%). An explanation for some of the poorer results was likely the accuracy of the calibration of the detectors used in the experiment (31).

Further contributing factors were the limited temporal and spatial resolution of the meteorological data and NAME's inability to consider meteorological data for a single site with a temporal resolution of less than one hour. These factors had a much greater impact on estimates of dose derived for the much shorter integration times of the measurements, for which the correlation of measured and modelled dose was much poorer. It was evident that, for a

given case, there was often as much variation among the model predictions as there was between the model predictions and the measurements. This highlighted the importance of the use of more than one set of meteorological data and indicated that there was probably at least as much uncertainty associated with the application of the meteorological data as there was with the cloud gamma modelling itself.

4.2.2.4 Verification of NAME

Model verification is regularly conducted within NAME, with code testing being performed during all model development work. Basic code testing is reported in some of the early model documentation papers, for example to ensure that predictions based on the puff dispersion scheme were in general agreement with those obtained using particles. Most of the model testing, however, is carried out by NAME developers as part of the code development process and results are recorded in the change management documentation. Comprehensive testing of NAME is conducted whenever a stable release of the model is produced (62).

Bedwell (2009a) documented testing performed to ensure that the model reflects the intended design. The implemented Lagrangian particle approach and semi-infinite cloud approach for estimating external gamma dose from the plume were tested by way of a combination of hand calculations, internal intercomparison between the 2 approaches for scenarios where the model predictions should be analogous and intercomparisons with other models, such as COSYMA (76) and HotSpot (an atmospheric dispersion model developed by the Lawrence Livermore National Laboratory (46)) for short duration releases, and PC CREAM 08 and ADMS for continuous releases. Bedwell (2009b) detailed the testing of the implementation of decay chain modelling within NAME, which again focused on a combination of hand calculations and model intercomparisons. Testing was performed for a range of radionuclides and decay products, across a range of half-lives.

5. Terrestrial foodchain models

5.1 FARMLAND

5.1.1 Model description

The FARMLAND model (Food Activity from Radionuclide Movement on LAND) (15) is a compartmental model used to simulate the transfer of radionuclides through the terrestrial food chain. In FARMLAND the food chain is represented by a set of interconnected compartments within each of which radionuclides are assumed to be uniformly mixed and the movement of radionuclides through the food chain is represented by transfer rates between the compartments. FARMLAND comprises several sub-models each of which represents one of the main food groups of a typical UK diet: green vegetables; grain products; root vegetables and potatoes; soft fruit and orchard fruit; milk, meat and offal from cattle; and meat and offal from sheep. The sub-models developed to describe the transfer of radionuclides to plants and to animals grazing pasture are presented in Figures 1 and 2, respectively. Many radionuclides are considered in FARMLAND, although the degree of complexity with which they are modelled varies according to their radiological importance and the availability of experimental data describing their behaviour. For example, element-specific models have been developed for cattle and sheep to take into account the important biological and metabolic processes for those elements whose transfer to animal products is significant. Figure 3 shows the compartment model structure used to simulate the uptake of radioiodine by cattle. The parameters k_{ij}, are the transfer rates between the compartments and represent specific processes, such as absorption or excretion. More information on how the transfer rates were determined is given in the FARMLAND report (15).

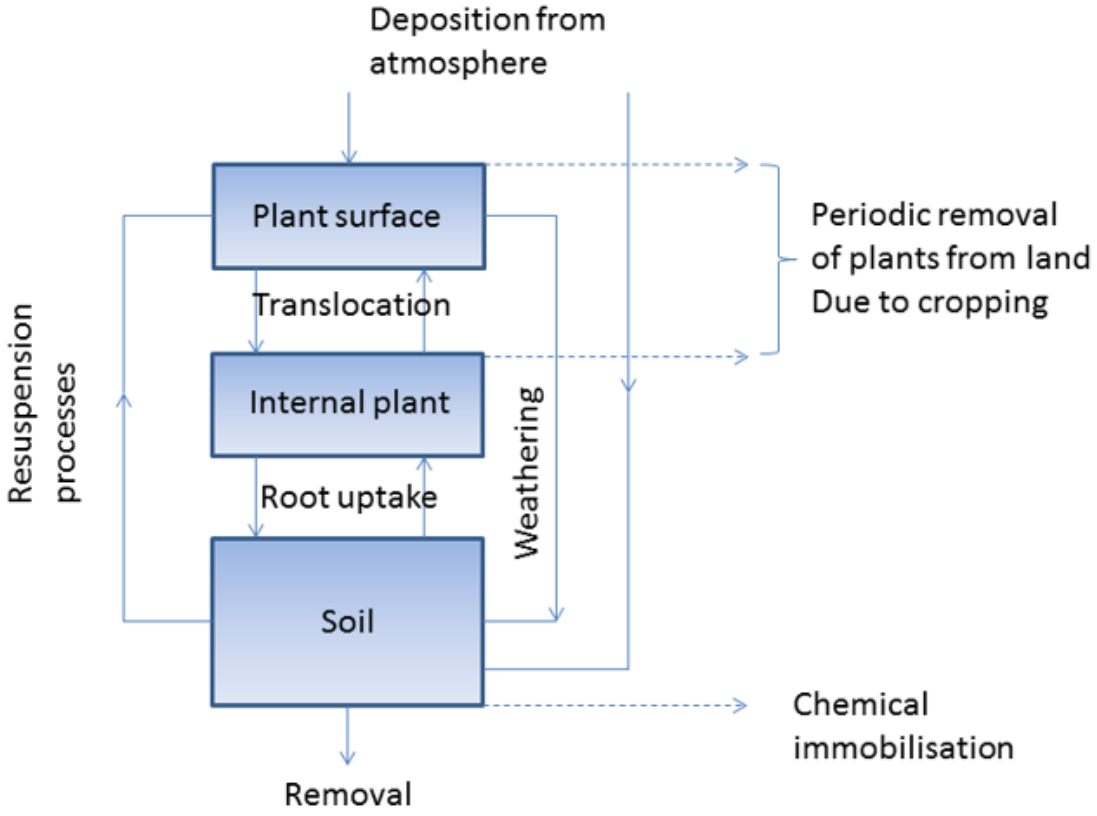
Since the publication of NRPB-R273 (15) new models for calculating the transfer of radionuclides in fruit have been developed and incorporated into the FARMLAND suite of models. For accidental releases, models for soft fruit and orchard fruit have been developed. However, for routine releases, a simpler generic model is proposed which represents both orchard and soft fruit. The fruit models are described in report HPA-CRCE-039 (14).

FARMLAND also contains a sub-model for the transfer of radionuclides through an undisturbed column of soil, typical of that which exists beneath permanent grassland. The soil column is represented by 4 compartments of varying depth and transfer rates between the compartments represent the predominantly downward migration of radionuclides. Many factors influence the rate of migration, particularly the chemical form of the element, soil composition, climate and rainfall. However, in its generic form the soil model uses transfers that are largely element independent and represent the movement of water through the soil. Further details of the soil sub-model are given in report NRPB-R273 (15).

FARMLAND is used to study the transfer of radionuclides through the foodchain following either accidental or routine releases of radionuclides to the atmosphere, and implementations of

FARMLAND are included in both the accident consequence assessment tool PACE and the software program for assessing discharges from routine releases, PC-CREAM 08. However, the way in which it is used depends on the mode of release. For example, agricultural practices are modelled in more detail when the application is for an acute release, such as occurs in nuclear accidents, and activity concentrations in foods are required over timescales of just a few days. PACE, therefore, makes use of data sets of time dependent activity concentrations in foods per unit deposition that have been generated using an implementation of FARMLAND that takes account of detailed agricultural practices assuming that the accident occurred in either summer (July) or winter (January), since the time of the year when the accident occurs will influence the activity concentrations in foods. For PC-CREAM 08, where discharges are assumed to be continuous and constant, versions of the FARMLAND model in which the agricultural practices have been simplified are included in the software.

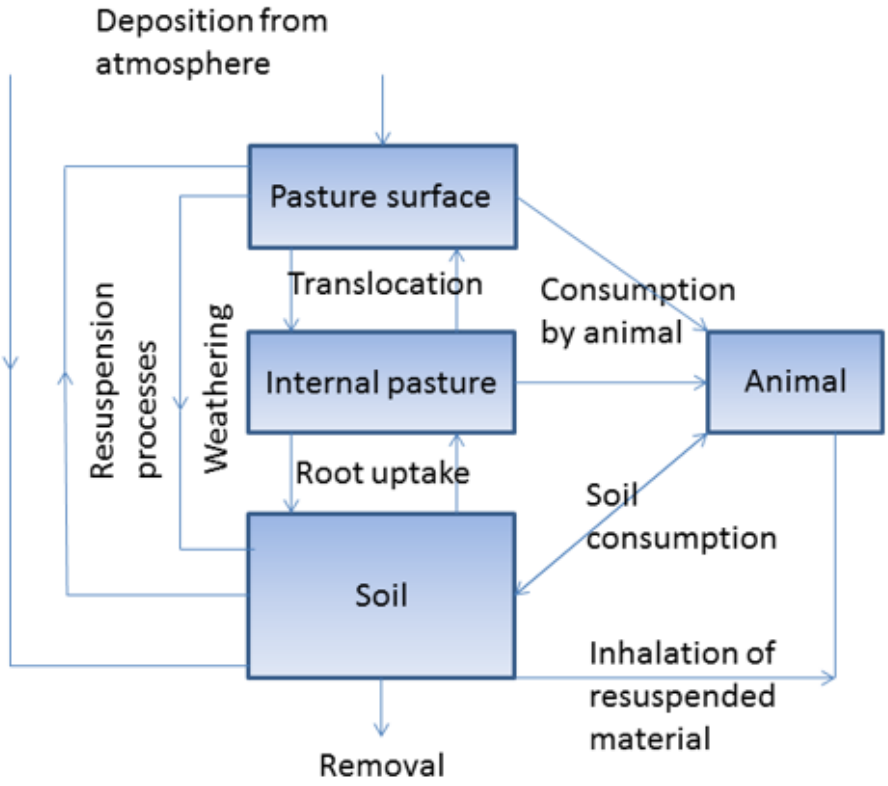
Figure 1. Schematic of the principal mechanisms for the transfer of radionuclides in plants



Accessible text version of Figure 1

This diagram describes the principal mechanisms considered in the FARMLAND model for the transfer of radionuclides to plants. Three compartments are shown representing plant surface, internal plant and soil. Initial deposition of radionuclides from the atmosphere can occur onto soil and plant surfaces. Deposition onto the soil can be transferred to the plant via root uptake or resuspension. Deposition onto plant surfaces can be lost to soil by weathering or moved to internal parts of the plant by translocation. Chemical immobilisation in the soil is modelled for radiocaesium. Cropping of the plant is represented by the transfer of radioactitivty out of the system.

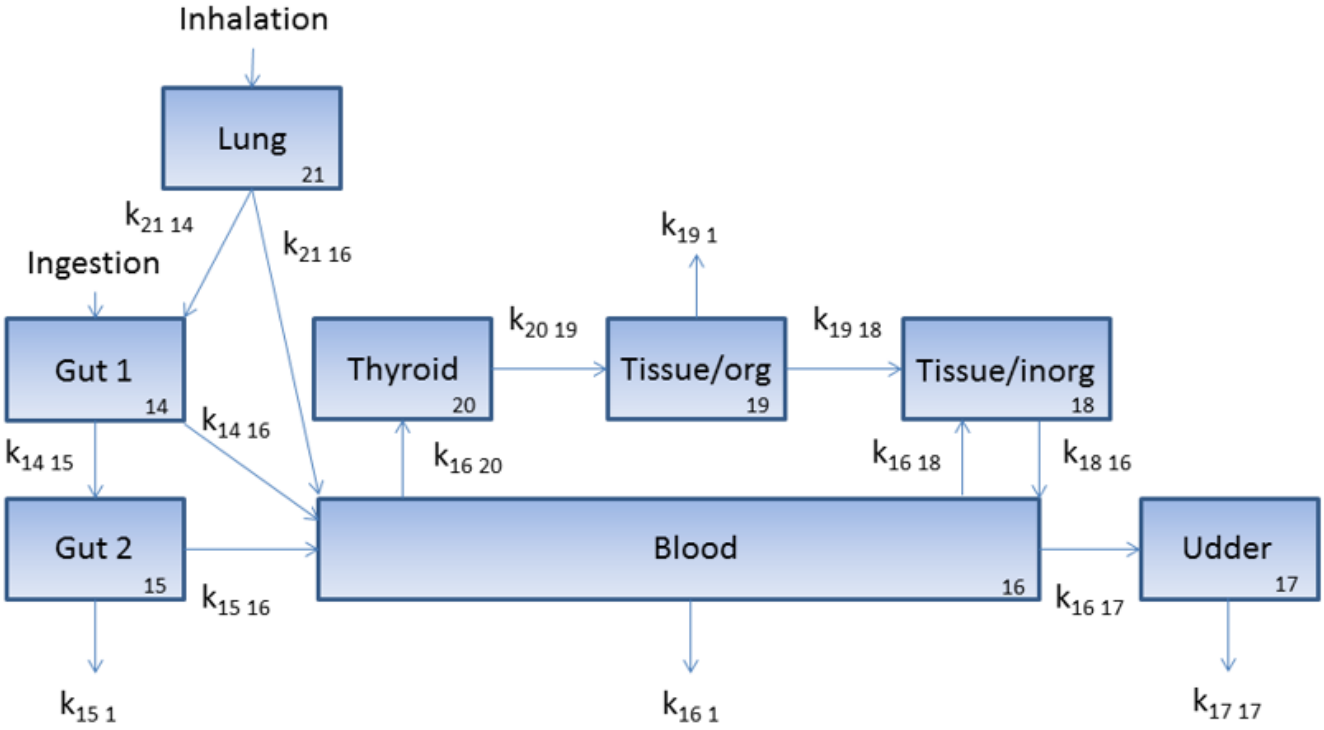
Figure 2. Schematic of the principal mechanisms for the transfer of radionuclides in animals



Accessible text version of Figure 2

This diagram describes the principal mechanisms considered in the FARMLAND model for the uptake of radionuclides by animals. Four compartments are shown, 3 are as described in Figure 1 and the fourth represents an animal. Following deposition onto soil and plants, radionuclides can become incorporated into plants and soils. Transfer to the animal is then modelled by considering ingestion of plants and associated soil, and inhalation of resuspended material. Loss from the animal to soil, for example by excretion, is also considered.

Figure 3. Compartmental model structure for iodine transfer in dairy cows



Accessible text version of Figure 3

This diagram is an example of the FARMLAND model used to represent the transfer of radionuclides within dairy cows. The diagram shows 8 compartments with transfers between them, denoted by the parameter k. Following initial upake via inhalation and ingestion, radionuclides are transferred to body tissues, blood and milk. The model shown here is for iodine and therefore the thyroid is also included. The transfers between the model compartments have been determined by fitting the model to experimental data.

5.1.2 Verification and validation

Verification and validation studies previously carried out using FARMLAND are discussed in reports NRPB-R300 (98), NRPB-M523 (12) and IAEA TECDOC 904 (51). Table 2 summarises the studies carried out. These are predominantly validation studies in which FARMLAND predictions have been compared with those from other models or measurements. The various studies have shown that FARMLAND generally performs well when the model predictions are compared with measurements over a range of sites and particularly with measurements from Chernobyl.

Table 2. Verification and validation studies carried out with FARMLAND (12, 51)

Study	Year
Comparison of 3 UK foodchain models	1983
Comparison of FARMLAND with German model ECOSYS	1985
Comparison of FARMLAND with German model ECOSYS	1988 to 1989
BIOMOVS International model intercomparison: scenario B1 for continuous deposition to agricultural land	1988 to 1990
UK fallout from weapons testing	1980 and 1987
Field data from Cumbria. UK. for pasture·cow·milk pathway	1983
UK monitoring data from the Chornobyl accident for milk. green vegetables and lamb	1987 to 1989
Field data from the EU from the Chornobyl accident for pasture to cow·milk pathway	1987 to 1989
Field data from Cumbria, UK, from the Chornobyl accident for winter feeding of cattle	1987 to 1989
Feeding trials on cattle and sheep	1987 to 1989
Data from the Chornobyl accident for milk. beef and grain at locations In the northern hemisphere (BIOMOVS to Scenario A4)	1987 to 1990
Data from the Chornobyl accident from southern Finland for a number of foods and whole body burdens (VAMP scenario S)	1992 to 1994

Since the publication of NRPB-R300 work to validate FARMLAND has been limited. However, a general review of the FARMLAND model parameter values was conducted in 2005 using data from a number of sources (92, 33, 35). As a consequence of this review changes were made to the equilibrium soil-to-plant concentration ratios and animal equilibrium transfer factors for some radionuclides.

In addition, verification and validation of the new fruit model was carried out and is reported in HPA-CRCE-039 (14). The report draws upon 2 studies from the Biosphere Modelling and Assessment (BIOMASS) programme of the International Atomic Energy Agency (IAEA) (82, 94) and the study of Pinder and others (1987).

The first is a model intercomparison exercise which was carried out using 2 scenarios, one based on a single acute deposition of ^{137}Cs to strawberries and apples and the other on a continuous deposition to strawberries, blackcurrants and apples (82). For the single deposit onto strawberry plants the FARMLAND model estimates fall within the range of those from other models. For the single deposition onto an apple tree the model estimates in the first year agreed well with other models but were outside their range in subsequent years, being lower by a factor of 2. This difference might be due to the choice of root uptake factor which was the dominant transfer route after the first year. The generic fruit model was used for the continuous release as

this is the recommended model configuration for this type of release. The generic model parameter values are based on those for apples, which are the dominant fruit species grown and consumed in the UK. As such estimated activity concentrations for the continuous release scenario agreed best with predictions from other models that represented apples rather than soft fruit.

The second study is a review of measured activity concentrations in strawberries by Ould-Dada and others (2006) and was used to validate the soft fruit model in FARMLAND. The measurements were derived from work carried out at Università Cattolica del Sacro Cuore of Piacenza (Italy) to investigate the short-term transfer of ^{134}Cs and ^{85}Sr via leaf-to-fruit and soil-to-fruit in strawberry plants after an acute release. Activity concentrations in strawberries calculated using the soft fruit model tended to overestimate measurements by factors of as much as 3 when the plant was exposed to foliar deposition. However, activity concentrations in strawberries predicted by the FARMLAND model following soil contamination were significantly lower than the measurements. It is noted that the measurement data were derived from a pot study which used peaty soil and was irrigated regularly, factors that are known to result in more enhanced uptake.

The third study by Pinder and others (1987) considered continuous wet deposition of ^{238}Pu onto orange trees for a period of 42 days. The activity concentration in fruit at the end of this period and the average deposition rate were measured. The ratio of these 2 quantities was found to be in good agreement with predictions of the FARMLAND fruit model applicable to continuous releases, that is, the generic fruit model.

More recently, FARMLAND was included in the review of dose assessment models for regulatory purposes undertaken by Working Group 1 of the IAEA model intercomparison study EMRAS II[1] (55). The version of FARMLAND as implemented in PC-CREAM 08 was used in this study. This version is configured to calculate activity concentrations in foods following continuous and constant discharges as would be expected to occur under normal operations. The scenario considered the release to atmosphere of ^{60}Co, ^{131}I and ^{137}Cs, subsequent deposition onto the ground and accumulation in terrestrial foods. Although there was some confidence that activity concentrations in the air calculated by different models were consistent it was difficult to compare the activity deposition onto the ground and therefore to determine whether differences in activity concentrations in food were as a result of different deposition models or different foodchain models. Nevertheless, in most cases the activity concentrations in foods calculated using FARMLAND were consistent with those calculated by other models used in the intercomparison and were generally lower than those predicted by more conservative screening models such as those described in IAEA report SRS-19 (52).

[1] The IAEA EMRAS II Programme ran from 2009 to 2011. RAD staff members participated in Working Group 1 ‘Reference Methodologies for Controlling Discharges of Routine Releases’ and Working Group 9 ‘Urban Areas’. Further details can be found at EMRAS II: Environmental Modelling for Radiation Safety.

6. Models for external exposure and resuspension

6.1 GRANIS

6.1.1 Model description

GRANIS (Gamma Radiation Above Nuclides In Soil) is a model developed at UKHSA to calculate the external gamma dose from layers of contaminated material which have finite thickness yet are effectively infinite in horizontal extent (79). It is based on a point kernel approach in which the gamma dose rate is derived from a calculation of the flux density of photons of different energies emitted from a single point. This flux is then integrated over the array of point sources that make up the contaminated soil layer, taking into account backscatter and attenuation between the contaminated layer and the receptor. Backscatter in the material of the contaminated soil layer, shielding layer and air is modelled using a build-up factor. The amount of backscatter and size of the build-up factors depends on the composition and thickness of the material through which the photons pass and the photon energy. The number of soil layers, their thickness, soil density and elemental composition can be specified in the model inputs. The model allows different vertical activity profiles to be considered and can be applied to predict doses due to external exposure to gamma radiation from the deposition of radionuclides to the ground. The vertical activity profiles can be based on measurements or model predictions, the default model being the undisturbed soil model for grassland, as used in FARMLAND and described in section 5.1.1. GRANIS calculates effective doses and equivalent doses in 4 tissues (gonads, breast, thyroid and skin).

6.1.2 Verification and validation

The latest version of GRANIS (version 3) is used in PACE and PC-CREAM and has been compared with other calculation methods. These are listed below along with a summary of the findings which are taken from the report (79):

- 2 analytical methods, the first representing the surface of a semi-infinite medium, and the second infinite slabs of finite thickness
- published doses (32)
- Monte Carlo computer codes, such as EGS4 (9), MCNP-4 (110, 23)
- the computer programme Microshield (89)

For the case where dose rates at the surface of a semi-infinite medium are calculated the analytical solution assumes that the medium is an absorbing source of infinite volume which is uniformly contaminated and therefore the energy absorbed per unit volume equals the energy emitted per unit volume. The absorbed dose rate per unit volumetric activity in the infinite medium is then calculated using the photon energy and density of the medium. The dose rate

on the surface of a semi-infinite medium is then half the value calculated. A comparison of absorbed dose rates calculated using this method with GRANIS v3 showed that for the energy range 0.05 to 4 MeV, the 2 sets of results are within 5%. Below 0.05 MeV GRANIS v3 calculates smaller dose rates by as much as 30%.

Dose rates from slabs of finite thickness but infinite lateral extent were compared using GRANIS v3 and the analytical method described in Jaeger (1968) for a receptor point 1 m above the slabs and an energy range of 0.01 to 4 MeV. In this case, GRANIS predictions were up to a factor of 2 greater than those calculated using the analytical method. The discrepancy between these 2 models comes from the different use of build-up factors within the calculation of flux.

The method adopted in GRANIS is considered to be a better representation of the radiation scattering processes because the build-up factor is applied before the integration of the photon fluence across the soil layer. In the Jaeger method the build-up factor is applied after the integration of the fluence which assumes that the build-up factor is the same across the soil layer which is not true because it depends on the number of mean free paths through which the photon must travel.

A Monte Carlo method has been developed by Chen (1991) for calculating effective dose rates from slabs of finite thickness but infinite lateral extent. This approach was compared with GRANIS v3 for a receptor 1 m above the slab. GRANIS was found to calculate dose rates up to 20% smaller for photons at energies below 0.1 MeV. Above 0.1 MeV and up to 4 MeV there was good agreement between the 2 methods. Comparisons of effective dose rate at a height of 1 m were also made between GRANIS v3 and the EGS4 Monte Carlo code (9). These showed good agreement, within 15%, for energies in the range 0.1 to 4 MeV. At 0.1 MeV and below, GRANIS v3 results were smaller than EGS4 results by up to about 40%. Finally, the code MCNP-4 (110) was compared with GRANIS. At photon energies of 0.1 to 4 MeV, GRANIS v3 produced larger results than MCNP-4 by up to 30%. Below about 0.05 MeV, GRANIS produced smaller results than MCNP-4, the discrepancy between the 2 methods in this energy region was up to about 25%.

Another comparison was made using MICROSHIELD (89), a software program which calculates effective doses from gamma radiation using the point kernel method. A rectangular slab geometry with a lateral extent of 1000 m was chosen for the purposes of this comparison. Soil thicknesses of 1, 5 and 30 cm were considered with a dose reference point 1m above the contaminated soil surface. Comparisons could only be made for radionuclides with mono-energetic photons in the range of 0.02 to 1.5 MeV; for this energy range the dose rates calculated by the 2 models compared well. Dose rates were also calculated for a 1cm thick soil layer and heights of 0.1, 10 and 100 metres above the soil-air boundary. For heights of 0.1, 1 and 10 metres the results of the 2 models compared well between 0.02 MeV and 1.5 MeV. For the 100 m height and an energy of 0.02 MeV there was good agreement between the models. However, above 0.02 MeV GRANIS increasingly underpredicted the dose rate, reaching a maximum difference of about 2 at 0.05 MeV. Above 0.05 MeV the difference in the model results reduced and beyond 0.1 MeV dose rates were within a factor of 2 for all heights.

Finally, a comparison was carried out using US EPA Federal Guidance number 12 (32) and GRANIS v3 for 5 radionuclides (^{60}Co, ^{137}Cs, ^{235}U, ^{226}Ra and ^{210}Pb) which were assumed to be uniformly distributed throughout 4 separate soil slabs of infinite lateral extent. The slab depths were 0 to 1 cm, 0 to 5 cm, 0 to 15 cm and 0 to infinite depth. The comparison of effective dose rates per unit concentration 1 m above the soil surface showed that the results for the 2 calculation methods did not differ by more than 20%.

The comparison of dose rates calculated using GRANIS v3 and those predicted by other codes and methods showed that the best agreement was observed at energies between 0.1 to 4 MeV. Below this energy range the height of the receptor point above the ground can have an important impact on the dose rate, particularly where this exceed 10 m, but differences are generally within about 40%.

6.2 ERMIN

6.2.1 Model description

The European Model for Inhabited Areas (ERMIN) estimates long-term levels of contamination on various surfaces and doses and dose rates, indoors and outdoors, within built environments. It also accounts for the effects of clean-up on those endpoints and it estimates worker doses, costs, effort and the amount and activity concentrations in the radioactive waste generated by the clean-up operations.

ERMIN proceeds in the following steps:

1. Estimation of the average initial radioactivity deposited on urban surfaces from the initial deposition to a reference surface.
2. Estimation of the radionuclides retained on urban surfaces after being subjected to weathering in the long-term.
3. Application of surface specific unit dose rates to the retained radioactivity to predict as a function of time the doses and dose rates, indoors and outdoors, in user selected idealised environments.
4. Modification of the radioactivity on urban surfaces, its retention and resulting dose rates to account for various types of countermeasures.

ERMIN includes a database of countermeasure parameters that are derived from the European Recovery Handbook (91) and a database of dose rates for urban surfaces within several built-up environment types taken from the literature. ERMIN was developed under a number of EC Collaborative projects, including EURANOS, NERIS-TP and PREPARE. The conceptual model and mathematical formulation were developed and reviewed by a number of institutions involved in these projects including Public Health England, Karlsruhe Institute of Technology, Technical University of Denmark, Helmholtz-Zentrum Muenchen, Danish Emergency Management Agency and Bundesamt für Strahlenschutz, and their precursor organisations. The ERMIN model was integrated into the jRODOS and ARGOS decision support systems by

Karlsruhe Institute of Technology and by Prolog Development Center. The full model description as well as the derivation and justification of the input parameters are given in the report EURANOS(CAT2)-TN(05)-04 (71).

6.2.2 Verification and validation

ERMIN was developed by Public Health England (now the UK Health Security Agency), Centre for Radiation, Chemical and Environmental Hazards under an ISO9001:2008 certificated quality management system that includes a range of verification methods and ensures well controlled testing. ERMIN has been used in several validation exercises which are summarised below. ERMIN was one of the 3 models used in the intercomparison exercise concerned with post deposition contaminant transfer and remediation performed by the Urban Remediation Working Group (WG) of the IAEA EMRAS II programme (56).

The scenario was hypothetical and concerned 2 areas within Seoul; a heavily built up area with tall glass office blocks and a park area with trees and grass. The heavily built up area proved challenging for ERMIN 1, as the idealised environments within the ERMIN database were not a good match. The closest environments were the multi-storey buildings environments, although these represent relatively low concrete apartment buildings and are not similar to the tall glass and steel buildings of the scenario. The environments in ERMIN also have more permeable soil surfaces than the heavily built up area in the scenario. These differences probably explain why ERMIN was the most conservative of the models, with the soil surfaces acting as sinks for the radioactivity; as a result the overall dose rates predicted by ERMIN did not fall as rapidly as those estimated by the other 2 models. The ERMIN predictions of dose and dose-rate started similarly to the other models but were roughly an order of magnitude higher than the nearest values predicted by the other models after a year. The dose reductions that ERMIN predicted following clean-up of various urban surfaces were similarly conservative. For the park area, all the models gave much more consistent results.

ERMIN 2.1 was used in a model comparison exercise undertaken by the ‘Exposures in contaminated urban environments and effect of remedial measures’ Working Group of the IAEA MODARIA[2] programme which followed the EMRAS II programme. The report of the Working Group has not been published yet; the work is summarised in Thiessen and others (2022).

The scenario used in the MODARIA intercomparison exercise was based on the Fukushima accident and was much more suitable for ERMIN than the Seoul scenario used in EMRAS II as it included a suburban area of light wooden buildings and lower rise concrete buildings, both with significant soil and trees within them. Although only one other model participated, the exercise was useful because the other model was specifically constructed for the Fukushima situation and closely calibrated to actual dose rates measured in the area and dose rate

[2] The IAEA MODARIA (modelling and data for radiological impact assessments) programme ran from 2012 to 2015 and continued some of the work carried out under the EMRAS programmes. RAD staff members contributed to Working Group 2 ‘Exposures in contaminated urban environments and effect of remedial measures’ using the ERMIN model. Further details can be found at MODARIA: Modelling And Data for Radiological Impact Assessments.

attenuation factors observed. To calculate indoor dose-rates and doses the other model used location factors derived from measurements within buildings in the Fukushima region. ERMIN compared well with the other model and was well within an order of magnitude for all endpoints. ERMIN has been applied to the Fukushima situation by other researchers (77), who found the estimated value of dose-rate agreed well with the observed data in the Fukushima area.

ERMIN 2.2 was one of the models used in a testing and comparison exercise that used real measurements of doses to large groups of individuals in Fukushima City, performed by the 'Assessment of Exposures and Countermeasures in Urban Environments' Working Group of the IAEA MODARIA II[3] programme. The report of the Working Group has not yet been published but is summarised in Thiessen and others (2022). Modellers were asked to estimate variability of doses to different groups who live and work in the Fukushima area, and the distribution of doses calculated by ERMIN taking into account variability of time spent indoors and outdoors in different locations, was a good match for the variability observed in the groups.

6.3 EXPURT

6.3.1 Model description

EXPURT (Exposure from Urban Radionuclide Transfer) is a compartmental model, which simulates the long-term retention and transfer of radionuclides on urban surfaces (74). The compartments represent 6 different urban surfaces: paved surface – such as roads, pavements and other hard outdoor ground surfaces – roofs, exterior walls, soil or grass, trees or shrubs and internal surfaces. Empirical parameters and simple assumptions are used to estimate the initial deposition to each surface and subsequent transfer between compartments is modelled with first order differential equations. The resultant time-dependent and integrated surface concentrations are then multiplied by factors to give dose rates at locations indoors and outdoors within different built environment configurations.

EXPURT calculates doses and dose rates to individuals in locations both inside and outside buildings using idealised urban environments to represent very broad environment types: lightweight buildings, brick buildings, multi-storey buildings and open areas without buildings. To estimate dose and dose rate reductions following remediation operations, EXPURT represents 3 types of operation: those that remove radioactivity from a surface (for example, road sweeping), fixing radioactivity to a surface to prevent transfer to other surfaces (for example, tie down with paint), and redistribution of the radioactivity (for example, moving radioactivity down the soil column by ploughing).

[3] The IAEA MODARIA II programme ran from 2016 to 2019 and continued the work of MODARIA. RAD staff members contributed to Working Group 2 'Assessment of Exposures and Countermeasures in Urban Environments' using the ERMIN model and to Working Group 3 'Assessments and Control of Exposures to the Public and Biota for Planned Releases to the Environment' using PC-CREAM 08. Further details can be found at MODARIA II: Modelling And Data for Radiological Impact Assessments.

6.3.2 Verification and validation

Version 3, the most recent version of EXPURT, was developed and tested under the formal quality assurance procedures operating in the Radiation Assessments Department. It was used in a model comparison exercise undertaken by Working Group 2 'Remediation Assessment for urban areas contaminated by dispersed radionuclides', part of IAEA EMRAS[4] programme (54). The exercise was based on the settlement of Pripyat which is close to the Chornobyl Nuclear Power Plant and was heavily contaminated during the accident and later abandoned. In the exercise, predictions of dose and dose-rate from 4 models, including EXPURT version 3, were compared to one another and also to a few real measurements of dose rate in outdoor locations taken several years after the event. It should be noted however, that because these were point measurements they were not completely comparable with the dose rates calculated by EXPURT which are generalised over all outdoor or indoor locations, and would represent a person moving around outdoors or changing location within a building.

Dose rates calculated by EXPURT agreed well with those calculated by most of the other models. Dose rates predicted by EXPURT were also within an order of magnitude of the point measurements at the outdoor locations that most resembled the idealised environments, while at other locations, as expected, the agreement was not as close. Appendix III.1 of the report of Working Group 2 (54), gives a full description of the set-up of EXPURT v3 for the EMRAS Pripyat scenario and a detailed analysis of its performance.

6.4 GRINDS

6.4.1 Model description

GRINDS is a code that is no longer used within the Radiation Assessments Department; however some data sets generated with GRINDS are still incorporated within other models, such as EXPURT (see section 6.3). GRINDS predicts dose rates inside buildings from gamma sources on external walls, internal walls, internal floors, internal ceilings, external roof and the ground outside. GRINDS uses a point kernel integration scheme, an analytical approach that represents distributed sources as a number of point sources.

6.4.2 Verification and validation

GRINDS and 2 other shielding codes were compared in an exercise as part of the CEC MARIA project on accident consequence management (84). A set of benchmark problems were developed that required dose rates at different positions in different housing types to be calculated following deposition of caesium onto different surfaces including the roof, external wall, internal surfaces and the surrounding ground. The comparison showed that all 3 models were adequate for most radiological situations and were in good agreement with each other (40).

[4] The IAEA EMRAS (environmental modelling for radiation safety) programme ran from 2003 to 2007. RAD staff members participated in the Working Group 2 'Remediation assessment for urban areas contaminated with dispersed radionuclides' of Theme 2 'Remediation of sites with radioactive residues'. Further details can be found at EMRAS: Environmental Modelling for Radiation Safety.

6.5 Resuspension model

6.5.1 Model description

Resuspension mechanisms can be broadly subdivided into man-made and wind-driven disturbances. In PACE (19) and PC-CREAM 08 (103) wind driven resuspension is modelled using the approach developed by Garland and others (34) and modified in report NRPB-W1 (105) to account for long-term resuspension. The model is based on a resuspension factor that is inversely proportional to time after deposition; the resuspension factor is a convenient way of expressing the observed relationship between surface and air concentrations. This approach, and more recent developments in resuspension modelling, were recently reviewed by the RAD staff members; the review is contained in report PHE-CRCE-047 (109). One of the main findings of this review was that the approach recommended in NRPB-W1 (105) is still widely used for generic assessments, with similar approaches adopted by NCRP (National Council on Radiation Protection and Measurements) (88) and IAEA (53), and appropriate for use in PACE and PC-CREAM 08. Nevertheless, the NRPB-W1 model does have some potential weaknesses as identified below.

6.5.2 Verification and validation

The review in report PHE-CRCE-047 (109) of the resuspension model described in NRPB W1 (105) included a verification of the model results, namely activity concentrations in air per unit deposit and committed effective doses and committed lung doses per unit deposit, presented in that report. Some errors were found and corrected values published (109). These corrected results are consistent with those used in PACE and PC-CREAM 08.

Validation of the Garland model against early data sets is discussed in detail in NRPB-W1. The report recommends that it is an appropriate model to use for the UK and other countries with similar environmental conditions. Limited data were available from more recent studies for a comparison with the values reported in NRPB-W1 (105). However, some literature sources recommend that a resuspension factor based on a power law function, of the type used in NRPB-W1, is preferable to an exponential function because the amount of resuspended material persists for longer than might be expected by an exponential function. One study also claimed that using a resuspension factor based on a power function resulted in a better representation of measurement data on both short and long timescales (37).

The main model limitation identified in report PHE-CRCE-047 (109) was the assumption that airborne contamination originates solely from an area of the surface nearby; in practice, the airborne contamination also includes resuspended material from more distant, upwind surfaces. Also, reported resuspension factors tend to be time averaged and may give erroneous predictions of instantaneous air concentrations, for example, in conditions when there is little or no wind or other disturbances. Wellings and others (2019) includes a discussion of several factors that can impact on resuspension and which may introduce considerable uncertainty to

the NRPB-W1 model predictions. These include topography, meteorological conditions, inhomogeneous deposition and age of deposition.

7. Models for dispersion in water bodies

7.1 DORIS (marine)

7.1.1 Model description

DORIS (Dispersion Of Radionuclides In the Sea) is the marine dispersion model used in PC-CREAM 08 (103) to assess the radiological consequences of continuous radioactive discharges into European coastal waters. It is configured to calculate activity concentrations in the marine environment following continuous and constant discharges as might occur under normal operation of a nuclear power station. DORIS is a compartmental model in which the compartments represent different parts of the marine environment including the water column and seabed sediments. The movement of radionuclides between compartments is modelled using transfer rates, which represent processes such as advection, diffusion and interaction with sediments. DORIS is an adaptation of the model developed by Working Group D of the MARINA II project to assess radiation exposures to the Member States of the European Union from discharges to the marine environment (99).

The main difference between the current version of DORIS and the model used in MARINA II, which comprises 72 compartments, is that DORIS includes a reduced number of compartments specifically in those sea regions remote from the source of the discharge. The consequences of these simplifications, which were introduced to improve model runtimes, were carefully assessed by comparing the results of the 2 models and were not found to be significant.

DORIS models local dispersion using a single water compartment and associated sediment compartments. This local water compartment is interfaced with the regional marine model and exchanges water and suspended sediment with the adjacent regional compartment. The adequacy of such an approach, which has been used previously in radiological assessments (99), depends on the values chosen for the local compartment parameters. The values were originally derived by defining 3 generic types of local compartment to represent estuarine, sheltered coastal and exposed coastal conditions (16).

More recently, the Environment Agency published parameter values which could be used in coastal dispersion modelling (29). These values were reviewed by the Radiation Assessments Department and have resulted in changes to some of the parameter values used in DORIS for different coastal locations (102).

For deep oceans, more than one compartment may be used to represent the water column with the deepest compartment forming an interface with the sediment compartments. The sedimentation process includes both the adsorption of radioactivity onto suspended sediments and the removal of suspended sediments from the water column to the seabed. Both processes are reversible and are modelled using element-dependent distribution coefficients (K_d).

7.1.2 Verification and validation

As discussed above, DORIS is a simplified version of the MARINA II model which has been implemented in the PC-CREAM 08 radiological assessment software. The development of PC-CREAM 08 has been carried out following formal quality assurance procedures which have included a review of the methodology describing the structure of the DORIS model and of the parameter values, and a comparison of model results with output from other models and experimental data.

The MARINA II marine model was tested during the MARINA II project by comparing predictions with environmental measurements (99). For example, activity concentrations of ^{99}Tc, ^{137}Cs and ^{239}Pu in marine media in different sea regions from discharges from European nuclear sites were predicted by the model for the period 1990 to 2000 and compared to measured data. Most predictions of activity concentrations in filtered water were within a factor of 2 of the measured values. Predictions of activity concentrations in sediments were within a factor of 2 to 5 of the measured values.

Further validation of the application of the MARINA II model to UK coastal waters was carried out in 2 studies undertaken by RAD staff members (8, 75), the first of which included a review of model parameter values and made recommendations to revise some sediment partition coefficients and biota concentration factors. The revised MARINA II model was compared in both studies with the more complex CSERAM model developed by CEFAS (Centre for Environment, Fisheries, and Aquaculture Science) (2) and with measurements. In general, the MARINA II model compared reasonably well with measurements although it was noted that at particular locations agreement could be improved by using region specific parameter values.

A comparison made between DORIS and the full model used in MARINA II (103) found that differences in activity concentrations predicted by the 2 models are not very significant. The calculations of individual doses, which are based on activity concentrations in the local and regional compartments closest to the release point, show less than a 10% variation between the models. For collective dose calculations, differences are seen at the greatest distances from the release point but, due to significant dilution, activity concentrations in water and sediment are several orders of magnitude lower than those in compartments closer to the release. It is therefore unlikely that the model simplifications introduced into DORIS will have a significant impact on the calculation of collective dose.

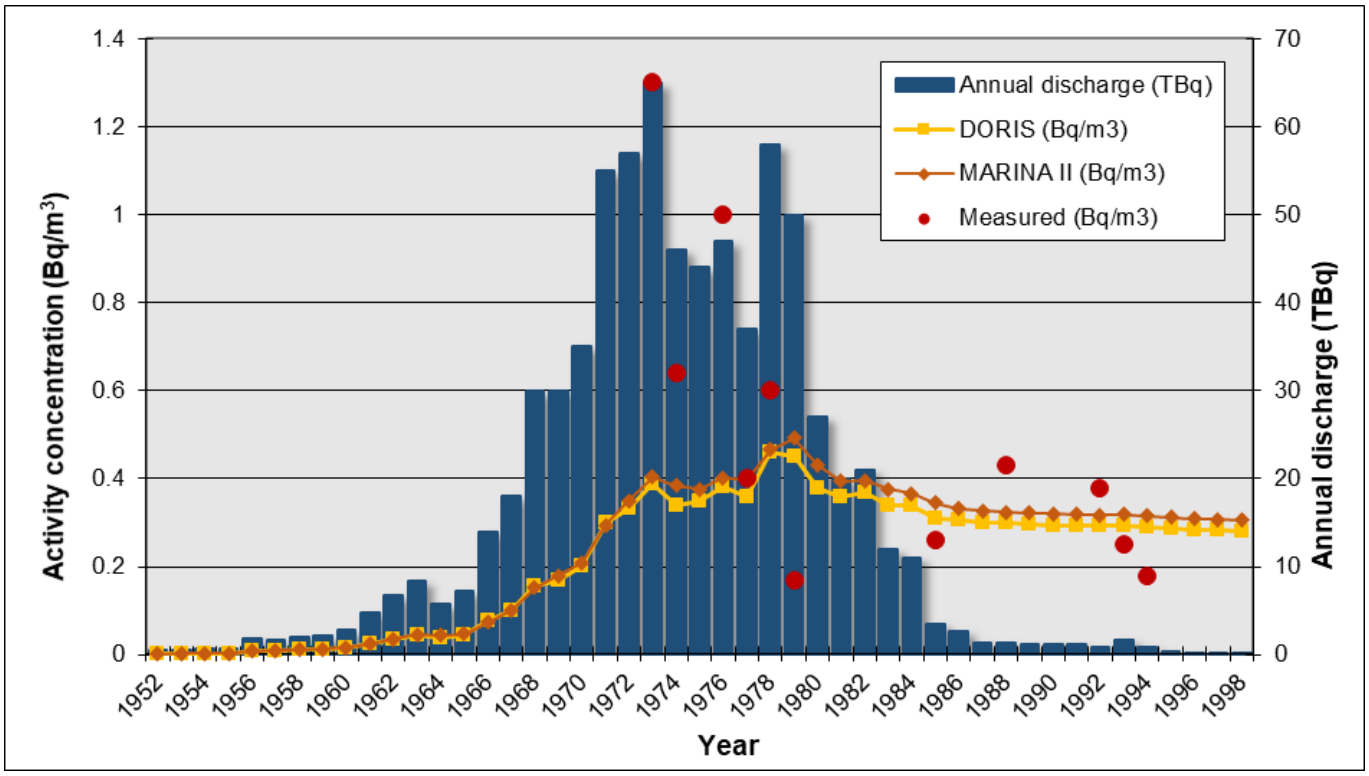
More recently, DORIS has been used in the IAEA model intercomparison study EMRAS II (55) which reviewed the results of dose assessment models for regulatory purposes. The scenario modelled was a hypothetical liquid release of 1 GBq y^{-1} of ^{60}Co, ^{90}Sr and ^{137}Cs from Sizewell B to the North Sea. Activity concentrations in the marine environment were not reported but doses were calculated for the ingestion of fish, crustaceans and molluscs, external exposure to intertidal sediments and inadvertent ingestion of sediments. Doses calculated using PC-CREAM 08 tended to be lower than those calculated by other models generally by one or 2 orders of magnitude. However, the majority of the other models used the simple screening

methodology of IAEA report SRS 19 (52) in which a plume model is used for marine dispersion and conservative assumptions are made concerning the location of individuals and the source of their food. Activity concentrations from DORIS, which assumes complete dilution in the local compartment immediately following the discharge, were much more consistent with the results of the POSEIDON-R model which is also based on the MARINA II methodology (55).

Since the EMRAS II study, a review of parameter values used in coastal dispersion modelling for radiological assessments carried out by the Environment Agency (29) has resulted in changes to the parameter values adopted for some local compartments in DORIS (102). The impact of these changes on environmental activity concentrations predicted by PC CREAM 08 and the doses at various UK coastal sites that discharge radioactive material into the sea are examined in the report PHE-CRCE-051 (102). Some of the more important changes were applied to the Sellafield local compartment and include a decrease in the volumetric exchange rate, an increase in the suspended sediment load and a decrease in the sedimentation rate.

One important development for the MARINA II model was the improvement in the modelling of sediment remobilisation. Figure 4 illustrates the good agreement between activity concentrations of $^{239/240}$Pu in filtered seawater measured in the Irish Sea West compartment and those predicted by the MARINA II model (8). The elevated levels of activity of $^{239/240}$Pu in seawater during the 1990s, despite the reduction in discharges, is characteristic of the improved remobilisation model. Figure 4 also shows activity concentrations in filtered seawater predicted by DORIS which are comparable to those predicted by the MARINA II model; the difference is due largely to a reduction in the volumetric exchange rate between the Sellafield local compartment and Cumbrian Waters regional compartment that was introduced following the review of the EA report on parameter values for coastal dispersion (29).

Figure 4. Comparison of activity concentrations of $^{239/240}$Pu in filtered water in the Irish Sea west region derived from MARINA II model, DORIS model and environmental measurements for discharges from Sellafield (adapted from Bexon and others, 2003)



7.2 River modelling

7.2.1 Models description

The models used in PC-CREAM 08 (103) to estimate activity concentrations in the environment following discharges of radionuclides to rivers include a simple dilution model and a dynamic compartmental model.

The simple dilution model in its most basic form is predicated on the assumption that the discharged activity is immediately dispersed over the breadth and depth of the river. The activity concentration in the river water is obtained by dividing the discharge rate by the flow rate. The inclusion of radioactive decay between the discharge point and the receptor location can also be considered by calculating the transit time. If data are available that show that dilution at the receptor location is incomplete, a simple dilution factor can be included to take this into account.

The dynamic model in PC CREAM 08 is based on that developed by Schaeffer (1976), (93, 100) but has been adapted for implementation as a compartmental model (103). In this model the river is represented by a series of water and sediment compartments and transfer rates between compartments are used to represent dispersion downstream from the point of release and adsorption of radionuclides onto sediments. The model assumes a constant and continuous discharge and instantaneous dilution of effluent in the total flow of the river at the point of

discharge. The flow of bed sediments downstream is modelled at a slower rate than the water and adsorption and settling processes are considered.

In both models, adsorption of radionuclides onto suspended sediments is modelled using a sediment distribution coefficient (K_d). In the simple model the conservative assumption is made that the activity concentration on bed sediments is equal to that on suspended sediments. However, in the dynamic model Schaeffer's empirical sedimentation coefficient k' is used to account for the removal of activity on suspended sediments from the water and its accumulation in bed sediments. The dynamic model implemented in PC-CREAM 08 (103) also includes the transfer of radionuclides to bed sediments from the dissolved fraction in the water column.

7.2.2 Verification and validation

The implementation of the simple dilution model in PC CREAM 08 was carried out under RAD formal quality assurance procedures and checked against hand calculations. The limitations of the model include the assumption that instantaneous and total dilution of the discharge in the river occurs and the activity concentrations in water and sediments are immediately in equilibrium, therefore comparison with other models and measurement data are not particularly useful. The simple dilution model performs satisfactorily if the river flow is turbulent and dilution of the discharged effluent is rapid, but less so if the flow is steady and vertical and horizontal mixing occur slowly.

The dynamic model has also been implemented in PC CREAM 08 in accordance with RAD formal quality assurance procedures and has been checked against earlier implementations of the model. Since publication of the NRPB R-300 report (98) the only additional validation of the model was its inclusion in a study for the UK Environment Agency to model the impact of radionuclide discharges into the River Thames (45). This study compared measured and model predicted activity concentrations in water, fish and sediments. In general, the model under predicted measured values and this was particularly noticeable for activity concentrations in bed sediments, which were underestimated up to about 2 km from the release point and overestimated at distances further downstream. However, it was noted that measurements included contributions from additional sources including naturally occurring radioisotopes, weapons testing fallout and historic discharges.

8. Models for global circulation

8.1 Models description

Because of their behaviour in the environment, some radionuclides with long half-lives may be dispersed globally and act as a long-term source of irradiation to both regional and world populations. PC-CREAM 08 includes models for the global circulation of ^{3}H, ^{14}C, ^{85}Kr and ^{129}I (103). These models are only used in the calculation of collective doses, since doses to individuals arising from the global circulation of these radionuclides are generally negligible.

8.2 Verification and validation

No additional verification or validation studies have been carried out on these models since publication of the NRPB R-300 report (98).

9. Model for economic costs of radiological accidents

9.1 Model description

COCO-2 (Cost of Consequences Offsite - 2) (39) is a model for assessing the off-site economic costs likely to arise following a large scale accident involving radioactive material. The consequences calculated by the model include direct costs, for example from the loss of use of property, and indirect costs such as the temporary loss of customers of businesses closed as a result of the accident. Where possible the COCO-2 model deals with both tangible consequences, which can be valued through the market, and intangible ones, which cannot be valued through the market. COCO-2 is not available as a standalone program but has been implemented in the Probabilistic Accident Consequence Evaluation (PACE) software tool (20).

In COCO-2 the losses incurred are broken down into 3 main categories. Agriculture losses arise through contamination of crops and livestock products that become unsuitable for its intended use. These losses not only include the immediate loss in the value of agricultural production at the time of the accident but also the future value of production that is no longer viable due to continuing soil contamination. Health losses cover the costs of medical treatment for people who become unwell as a consequence of the accident and the losses associated with reduced quality of life and productivity of people affected. Built environment losses include the production losses of industry including tourism and the lost use of capital assets such as accommodation and capital goods.

The COCO-2 model was developed to analyse radiological accident consequence costs in the UK specifically, although the principles of the model are valid for other countries and regions of the world. The default cost data that are provided with PACE are appropriate for the UK. A detailed description of the economics model can be found in the COCO-2 report (39).

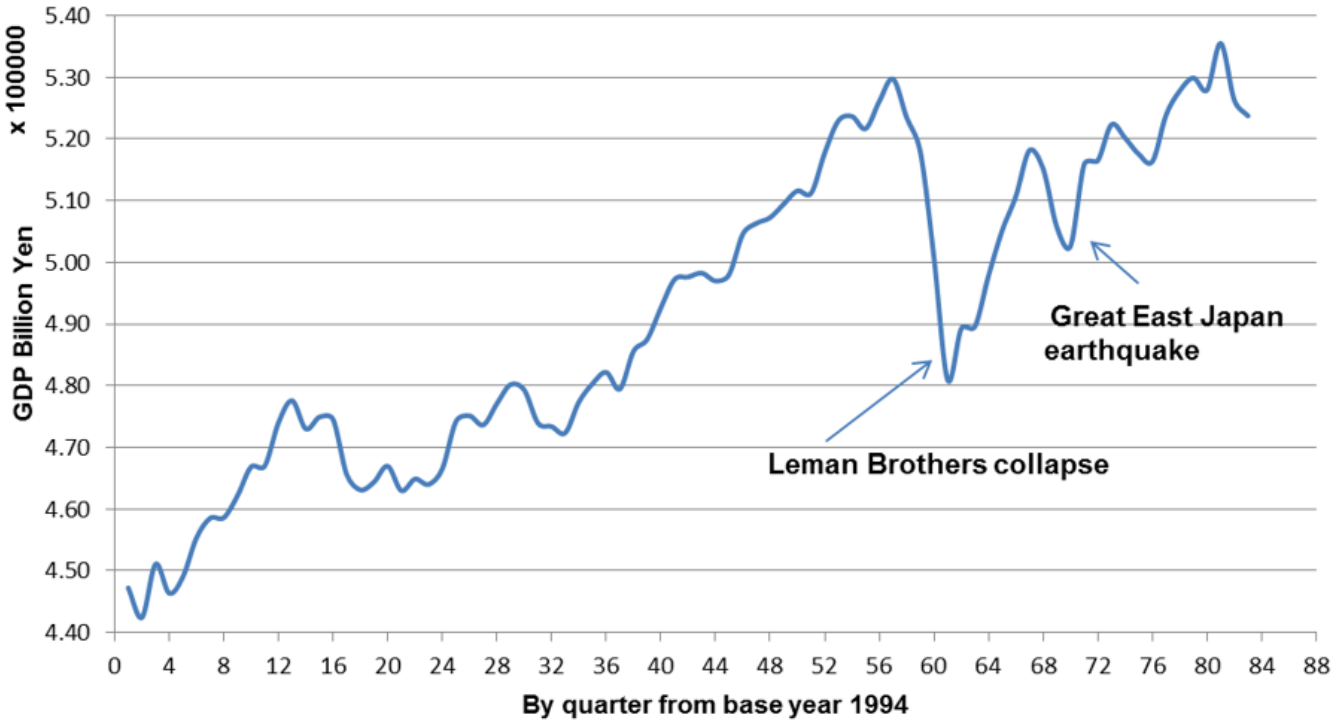
9.2 Verification and validation

Verification of COCO-2 in PACE has focused primarily on peer review by the Health and Safety Executive and extensive software testing. The model is based on well-established methods of analysing economic change and the data sources are fully referenced and consist mostly of UK Government departments and the Office for National Statistics (ONS). Exceptions to this rule are fully discussed in the COCO-2 report, as are the manipulations necessary to generate any required information that cannot be directly sourced.

Validation of COCO-2 is difficult because there are very few large-scale accidents and even for those accidents that have occurred there may be insufficient follow-up to collate all the required data. Although it is likely that the local area affected by the accident may take many years to recover, COCO-2 makes the assumption that, regardless of on-going relocation and

remediation, the economy on a national scale would adjust to recover within a period of 2 years. Therefore, any period of disruption and time away from home that exceeds 2 years, is not considered to incur any further national or whole economy costs (that is, these are truncated and capital losses realised at 2 years for calculation purposes). This view has subsequently been supported by a review of the duration of economic recessions in the UK by Hills and others (2010) that found the great depression of 1930 to 1931 and all subsequent recessions have lasted 2 years or less. Although these recessions have arisen because of the interplay of international factors and systemic structural issues in the UK economy and therefore had a different aetiology from any recession that might arise following a nuclear accident, it is reasonable to expect at least as equally rapid a recovery from an event that has a local focus and a physical and economic impact that diminishes with distance from the site of the accident. This key assumption is supported by data showing the response of the Japanese economy to the Fukushima accident (Figure 5).

Figure 5. The real Japanese GDP as a function of time with the major dip of the Leman Brothers collapse in 2008 and the smaller dip following the Tsunami and Fukushima accident of 2011



One area of particular concern that is not accounted for in the above general considerations are the consequences for the UK economy of any decline in international visitor numbers and their discretionary spending following a major accident. Tourism is an increasingly important sector in the UK economy and health and security related issues in the news can have an effect on visitor numbers (10). General health and security issues are distinct from a radiological accident which may be viewed differently by the public and it is not known if that will change the response in the UK. However, following the Tsunami and nuclear accident at Fukushima there is some evidence that indicates that inbound tourism did recover rapidly despite Japan experiencing both a nuclear accident and large scale structural damage along the eastern seaboard (111, 112, 113).

10. Conclusions

Modelling the transfer of radionuclides through the environment is a key part of the radiological impact assessments carried out using the software tools PACE and PC-CREAM 08. This report describes the work that has been carried out to verify and validate the models included in PACE and PC-CREAM 08 since the publication of NRPB-R300 (98). Steps taken to verify and validate these models have helped to ensure that they are reliable and fit for purpose in terms of the type of application for which they are used.

Since the late 1990s model development in RAD has been carried out within an accredited quality management system in which record keeping, configuration management and testing are essential requirements. Where possible quality controlled development has been backed up by model inter-comparison exercises, such as those coordinated by the IAEA, to give confidence that the models have been implemented correctly.

The verification and validation exercises described in this report demonstrate that the models implemented in PACE and PC-CREAM 08 are adequate representations of the real world and can be used with confidence for the purposes for which they were created.

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