DOSIMETRY AND INSTRUMENTATION

INTERNAL DOSIMETRY HAZARD AND RISK ASSESSMENTS: METHODS AND APPLICATIONS

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ABSTRACT

Routine internal dose exposures are typically (in the UK nuclear industry) less than external dose exposures: however, the costs of internal dosimetry monitoring programmes can be significantly greater than those for external dosimetry. For this reason decisions on when to apply routine monitoring programmes, and the nature of these programmes, can be more critical than for external dosimetry programmes. This paper describes various methods for performing hazard and risk assessments which are being developed by RWE NUKEM Limited Approved Dosimetry Services to provide an indication when routine internal dosimetry monitoring should be considered.

INTRODUCTION

The estimation of internal dose needs to be inferred from the measurement of activity in the individual (by in-vivo or in-vitro methods) or in the workplace environment. A variety of monitoring methods are available for making these measurements [1]. A review of the current monitoring practices within European Union countries was included within the OMINEX project (Optimisation of Monitoring for Internal Exposure) [2] and are summarised in Table 1.

Type of operation	*Total number*	*Monitoring method*			*Monitoring*		
		In vivo	*Bioassay*	*SAS/PAS*	*Routine*	*Special*	*Both*
Nuclear power plant	*34*	*33*	*15*	*7*	*8*	*4*	*27*
Reprocessing	*2*	*2*	*2*	*-*	*-*	*-*	*1*
Fuel fabrication	*7*	*6*	*6*	*2*	*3*	*2*	*5*
Decommiss-ioning	*6*	*5*	*6*	*1*	*2*	*2*	*5*
Research	*19*	*17*	*12*	*5*	*3*	*5*	*16*
Non-nuclear	*3*	*3*	*3*	*2*	*2*	*2*	*2*

SAS – static air sampling
PAS – personal air sampling

Table 1: Number of organisations using each monitoring method for routine or special monitoring, classified by type of operation. (from Optimisation of Monitoring for Internal Exposure (OMINEX); NRPB-W60)

The costs of operating these monitoring programmes can be significant. The OMINEX report [2] included a review of these costs; an example of the cost of the measurement of alpha activity in urine and faecal samples is included in Figure 1.

Within RWE NUKEM a variety of routine monitoring methods are employed, depending on the particular isotopes to be measured. A typical programme for monitoring potential exposures to actinides and fission products will incorporate personal air samples (PAS), urine, faecal and whole body monitoring. The actual cost of operating this programme will be dependent on specific commercial arrangements, but it is meaningful to provide a qualitative comparison to the costs

of operating an external dosimetry monitoring programme of monthly whole body photon (passive) dosemeters. For this illustration the cost of the internal dosimetry monitoring programme is approximately 60 times the cost of the external dosimetry monitoring programme. A review of the dose records for all classified (category 'A') workers on a nuclear site for 2004 indicated that the average external dose is 10 times greater than the average internal dose. It is, therefore, apparent that for routine monitoring programmes the normalized cost per mSv monitored is approximately 600 times greater for internal monitoring than for external monitoring. (This apparent cost ratio can vary significantly depending on the type of nuclides monitored.)

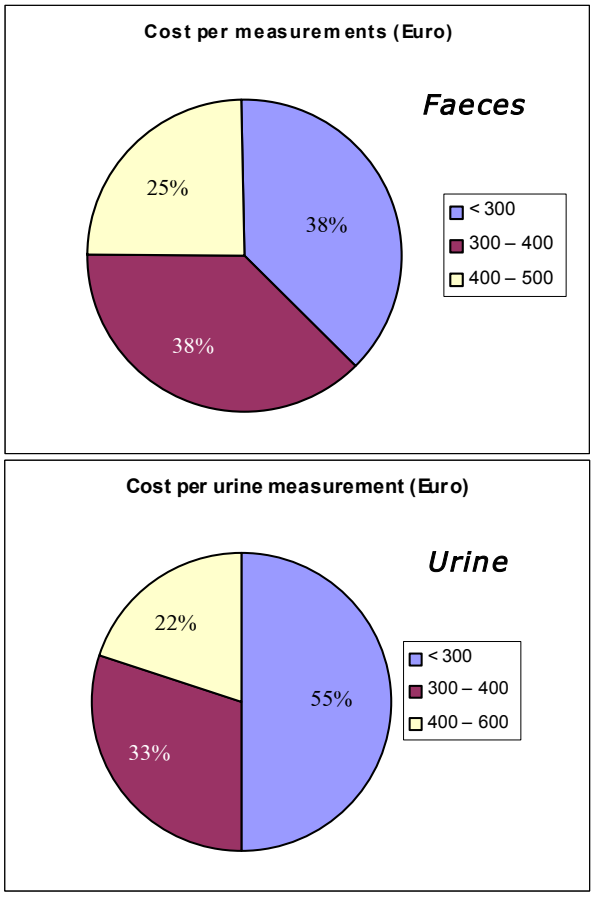

Figure 1: Costs of bioassay measurements of α-emitters in faeces and urine (from Optimisation of Monitoring for Internal Exposure (OMINEX); NRPB-W60)

Due to the comparatively high costs of internal monitoring programmes it may be cost-effective to invest greater effort into the prior determination of whether such monitoring is required, whilst still retaining adequate radiological protection.

IDENTIFICATION OF THE NEED FOR ROUTINE INTERNAL MONITORING

ICRP Publication 60 [3] states that …

"(268) Individual monitoring for intakes of radioactive material … should be used routinely only for workers who are employed in areas that are designated as controlled areas specifically in relation to the control of contamination and in which there are grounds for expecting significant intakes. …"

This principle is restated in IAEA guidance [4] and contained within UK regulation and associated guidance [5][6], which advise that routine monitoring for a specific component of dose (e.g. as arising from intakes of radionuclides) may not be required, provided that the expected magnitude of the dose does not exceed 1 mSv per year, taking into account the expected variability of the dose. However, there is still the problem of how to implement this principle, particularly for low dose/low risk situations where it would be preferable to obtain a quantified estimate of the 'expected dose' (and its variability or uncertainty) without the need for potentially costly individual routine monitoring programmes. Within RWE NUKEM Approved Dosimetry Services the approach is to perform internal dosimetry hazard and risk assessments specific to the workplace or operation.

INTERNAL DOSIMETRY HAZARD ASSESSMENTS

The primary purpose of the hazard assessment is to identify the sources which could give rise to exposures. The results of this assessment will be used in two ways:

- *to define appropriate dose coefficients which will then be used within the risk assessment (and any subsequent routine monitoring programmes); and*
- *to identify which radioisotopes should be monitored within any subsequent monitoring programmes.*

Information needs to be collated to identify the radio-isotopes and their likely physical and chemical nature. The preferred method for identifying the relevant isotopes is by analysis (e.g. spectroscopy) of contaminants found in the workplace, from workplace air samples and surface contamination smear samples. However, it is often the case that the levels of contamination found are too low to enable such analysis. Similar analyses may be performed for contaminants sampled in contained facilities (e.g. in a glove-box) or on ventilation extract and stack samples. It is more often the case that the best information available is inferred from the reported radio-isotopic inventories in 'safety case' or plant design documentation. Occasionally even this information is not available and the only recourse is to refer to the hazard assessments derived from similar workplaces. Each of these steps incurs increasing degrees of uncertainty.

Various analytical techniques are available to provide measurements of the physical and chemical forms of the materials in the workplace: however, it is normal practice to defer to guidance issued by (the then) NRPB[7] (now a division of the Health Protection Agency), which advises that for annual doses less than 5 mSv it is sufficient to refer to default values recommended by the ICRP. The currently assumed defaults for the physical (e.g. aerosol) and chemical (e.g. lung absorption) properties are as published in ICRP66[8] and ICRP68[9] respectively. Where a variety of chemical forms are possible, or if the specific forms are unknown or not defined in ICRP68, then the most reasonably pessimistic assumptions are made.

APPLICATION OF METHODS FOR HAZARD ASSESSMENT (Example)

An internal dosimetry hazard assessment was conducted for a waste treatment facility. The isotopic inventory was derived from the analytical results of a number of samples taken from the waste holdings at various times during the facility's operation (i.e. not specifically for this assessment). The characteristics for the chemical forms of material could not be specifically determined due to a wide range of potential sources of the waste materials being processed; therefore

the most conceivably restrictive default values were assumed from ICRP68 [9]. A default value of 5 microns was assumed for the AMAD of aerosols[8].

Isotope	Inventory (% Bq)	Dose Coef (Sv/Bq)	Lung Type	Dose per mix (Sv/mix)	Normalized hazard
$^{239/240}Pu$	21.7	3.20E-05	M	6.94E-04	8.81E-01
^{238}Pu	1.5	3.00E-05	M	4.50E-05	5.71E-02
^{241}Pu	75.8	5.80E-07	M	4.40E-05	5.58E-02
^{241}Am	0.17	2.70E-05	M	4.59E-06	5.82E-03
^{226}Ra	0.11	2.20E-06	M	2.42E-07	3.07E-04
^{137}Cs	0.6	6.70E-09	F	4.02E-09	5.10E-06
^{90}Sr	0.02	3.00E-08	F	6.00E-10	7.61E-07
^{60}Co	0.01	1.70E-08	S	1.70E-10	2.16E-07

Table 2: Isotopic inventory and normalized hazard assessment

The results are tabulated and the specific hazard for each isotope is simply represented as the product of their dose coefficient and relevant abundance in the overall isotopic mix. This data is normalized and listed in order of the apparent hazard for each isotope, as in Table 2.

The determination of the composite dose coefficients is not only dependent on the data as presented in Table 2, but also on the response of the monitoring techniques. In this case the primary monitoring methods were to be via air sampling, which is analysed by gross alpha and beta counting. It is assumed that all the alpha-emitting isotopes will produce an equivalent response in the counting system; similarly for the beta-emitters. The one exception is ^{241}Pu; the maximum energies of the beta emissions from this isotope are below the effective detectable range of the measurement system. The presence of this isotope cannot therefore be measured and needs to be inferred from measurements of the other isotopes in the mix. In this case the inference is derived from its relative abundance to the alpha-emitting isotopes, because it is assumed that the presence of ^{241}Pu is more reliably mimicked by the presence of other plutonium isotopes rather than other beta-emitting isotopes.

This calculation produces composite dose coefficients as in Table 3. It is noted that these coefficients are converted into Derived Air Concentration values [1], which are more directly relevant to the interpretation of air sample measurements.

	'e(50)' Sv/Bq per Bq measured	DAC (Bq/m^3)	Comment
Alpha	3.4 E-5	0.25	Includes inferred contribution for ^{241}Pu
Beta	7.4 E-9	1105	Assumes beta-emissions have equivalent detection efficiency

Table 3: Composite dose coefficients (expressed as e(50) and DAC values)

Note: if a different monitoring method is to be used then the apparent composite dose coefficient could be significantly different – e.g. if in-vivo monitoring for ^{137}Cs were to be used, then the 'apparent dose coefficient' for the mix would be 1.3 E-3 Sv per Bq intake per Bq measured.

The determination of which isotopes should be subject to specific monitoring are defined by those which have a specific hazard greater than 10% of the normalized hazard – this '10% rule' has been chosen as an arbitrary but reasonable value. This

assessment indicated that monitoring is only really required for $^{239/240}$plutonium. Due to the undefined chemical form, and hence solubility of the plutonium compounds, both urine and faecal monitoring programmes require to be considered. The uncertain solubility of compounds of the other isotopes would have no affect on the selection of appropriate monitoring methods.

INTERNAL DOSIMETRY RISK ASSESSMENTS

The previous section discussed how the assessment is made of the hazards that might be experienced in the workplace: the next process is to make an assessment of the 'expected dose' arising from potential exposures to these hazards. Within RWE NUKEM three different methods are being developed to determine prior risk assessments and subsequent validation, and are described below (with specific applications as examples).

'Managerial' Risk Assessments

In this case the risk of exposure is simply inferred from related risk assessments - e.g. related directly to the designation of controlled areas. The management (with advice from health physicists) defines local rules which require that any worker entering a contamination controlled area is subject to routine internal monitoring. However, this approach may be overly prescriptive in circumstances where controlled areas are designated partially on operational convenience - e.g. may include offices or access areas; or for workers who might access the area but are not directly engaged in active operations - e.g. supervisors, visitors.

The standard method is to limit individual monitoring to workers directly engaged in active operations; other workers do not require monitoring provided that their occupancy time of the controlled area is less than 10% (of normal working time). By the application of these rules we may claim greater than 90% confidence that the un-monitored workers are not at risk of a dose above the defined level (1 mSv per year). This assessment can be validated by reviews of the estimated doses of the workers who are monitored: if the maximum range of these doses is less than 10 mSv then this provides reassurance that the doses for un-monitored workers are less than 1 mSv. NB: the above claims of confidence are critically dependent on two assumptions - that monitored workers have 100% occupancy of the controlled area; and that the risk of exposure within the controlled area is represented by a uniform probability distribution with time.

A similar approach may be used for workers who may work in different facilities with differing hazards. Table 4 presents the respective hazard assessments for two separate facilities. The problem is to determine whether workers should be subject to routine monitoring appropriate for both of the hazards, or just the facility which provides the main hazard.

Liquid waste treatment plant		
Isotope	*Inventory (% Bq)*	*Normalized hazard*
$^{239/240}$Pu	21.7	8.81E-01
^{238}Pu	1.5	5.71E-02
^{241}Pu	75.8	5.58E-02
^{241}Am	0.17	5.82E-03
^{226}Ra	0.11	3.07E-04
^{137}Cs	0.6	5.10E-06

Solid waste treatment plant		
Isotope	*Inventory*	*Normalized hazard**
$^{238/239/240}$Pu	*not defined**	1.00E+00
^{241}Am		1.00E+00
^{226}Ra		1.00E+00
$^{234/235/238}$U		1.00E+00
$^{228/230/232}$Th		1.00E+00
Fission Products		1.00E+00

^{90}Sr	0.02	7.61E-07				
^{60}Co	0.01	2.16E-07				
* relative composition of the inventory cannot be readily quantified; therefore the normalized hazard is set to 1.0 for each possible group of radionuclides						

Table 4: Hazard assessments for two separate facilities

If the monitoring regime for a worker is based on the hazard assessment for the liquid waste plant (i.e. for $^{238/239}Pu$ only) but the worker was then employed 100% of the time at the solid waste plant, the monitoring regime would only be appropriate for 17% of the hazard. If we required the monitoring programme to provide a confidence level of at least, say, 90% that all hazards were effectively monitored we could increase the scope of the programme to include all the isotopes, which could become very costly. An alternative approach is to set limits on the length of time that a worker may be potentially exposed to unmonitored hazards and risks. To do this we need to consider the fraction of the overall hazard that these un-monitored radionuclides represent, and the relative risk between the different facilities. A review of dose records for the two facilities indicated that the relative risk could be considered four times greater for the solid waste facility than the liquid waste facility. This information may be analyzed by the relation in Equation 1.

$$C = H_1(1-T.R) + H_2T.R \qquad \text{Eqn 1.}$$

Where C = *confidence level*
H_i = *proportion of hazard monitored in facility i*
T = *proportion of time exposed to the un-monitored hazard (i.e. in facility 2)*
R = *factor to indicate the relative risk of exposures in facility 2 compared to facility 1*

From the above information this can be expressed as

$$0.9 < 0.94(1- T.R) + 0.17\ T.R$$

so $T < 0.013$ *(R = 4 , form above information)*

This calculation indicates that the length of time spent in facility 2 must not exceed 0.013 of total working time in order to claim 90% confidence in the monitoring programme – i.e. less than 3 days per year. Should this time be exceeded then the scope of the monitoring programme will need to be increased to monitor for a greater proportion of the hazard.

Predictive Risk Assessments

This type of assessment is considered when there is reasonable foreknowledge of the activity levels to be processed. A simple algorithm is then applied, such as is contained within IAEA Safety Standard Series RS-G-1.2[4], reproduced in Equation 2.

$$D = 1000 \sum_j A_j\, e(50)_j\, f_{fs}\, f_{hs}\, f_{ps} \qquad \text{Eqn 2}$$

Where D = *decision level (mSv)*
A_j = *cumulative annual activity (Bq) of isotope j*
$e(50)_j$ = *dose coefficient (Sv/Bq)*
f_{fs} = *physical form safety factor (e.g. the nature of the material handled)*
f_{hs} = *handling safety factor (e.g. nature of the operations)*
f_{ps} = *protection safety factor (e.g. level of containment)*

The IAEA guidance includes recommended default risk factors for a range of different levels of containment, form of the material and type of operations. If the calculated decision level (D) is greater than 1 then routine monitoring should be

considered; specific monitoring should be considered for each radionuclide which produces an individual decision level greater than 0.3.

This method was applied for a sample analysis laboratory associated with the liquid waste plant. The radioisotopes of concern were as listed in Table 2; the algorithm was simply applied for the plutonium alpha-emitting isotopes as these constituted the majority (94%) of the hazard. The other relevant information used is listed in Table 5.

Parameter	*Assessment*	*Assumed value*
Radioisotopes	*$^{238/239/240}$Pu*	*60 kBq per year max (local restriction)*
Dose coefficient	*Assumed ^{239}Pu (type M, 5 μm)*	*3.20E-05 (Sv/Bq) inhalation*
Physical form (f_{fs})	*IAEA RS-G-1.2 default*	*0.01*
Processes (f_{hs})	*Normal chemical operations*	*1*
Containment (f_{ps})	*Fume hood*	*0.1*

Table 5: Parameters for predictive risk assessment of liquid waste plant laboratory

By applying these assessed values into the algorithm in Equation 2 a decision factor (D) of 1.9 is calculated, thus indicating that routine monitoring for plutonium alpha-emitting isotopes is required. Because the hazard assessment (Table 2) indicates that these isotopes constitute 94% of the total hazard, the above assessment also indicates that the individual decision levels for other isotopes will be less than 0.3; so are not required to be monitored. The results of this assessment can provide three options-

- *implement routine monitoring; or*
- *upgrade containment (e.g. a glove box would provide a factor of 10 reduction in D); or*
- *reduce activity levels (by reducing the local restriction by a factor of 2).*

If such an assessment indicates that no monitoring is required then the validation of the assessment is difficult, due to the absence of monitoring data. The simplest approach is to assume that safety factors for containment represent the risk that the containment will be breached (rather than indicating a fractional release factor): from this assumption we can then claim that any evidence that the containment has been breached (by routine contamination surveys, etc) would invalidate the risk assessment, and that special monitoring should be considered.

Retrospective Risk Assessments

The simplest method to perform a retrospective risk assessment is to review past dose records; however, additional considerations may be required in the following circumstances (for example):

- *new or changed operations;*
- *when dose records represent exposures in a range of different workplaces, whereas a risk assessment is required for a specific facility;*
- *validation of a risk assessment, if the assessment concludes that routine monitoring is not required (e.g. by the use of workplace monitoring).*

The current method is to review the results of static air samples (SAS) taken from the work place. These reviews are used as part of the initial risk assessment, and also as a means for providing an on-going validation of the assessment. It is considered that SAS can only be reliably used for this purpose in areas where there is a negligible risk of highly localised exposures, otherwise more direct individual monitoring is applied – e.g. by use of personal air samplers (PAS). The actual methods for using SAS as part of a risk assessment are described by use of the following example.

The past dose records for a group of workers indicated average internal doses << 1 mSv and that the risk of acute exposures > 1mSv was less than 1%. These workers were primarily employed in a facility used for examination, dismantling and packaging fuel assemblies and other sources. All active operations were conducted within highly shielded cave lines or by use of RPE, such that the risks of localised exposures are confined to a limited number of planned tasks. Therefore it was considered that SAS could be used as part of the risk assessment for the general areas and operations within the facility.

It was first assumed that air activity as sampled by SAS is only present during working hours: this is a cautious assumption, and can be easily checked (if required) by separate 'working-hours' and 'silent-hours' sampling. The SAS results for the previous year were summarised to provide monthly averages for the area, and compared to a similar summary of PAS results from the facility. The PAS:SAS ratios are calculated for each month, as shown in Figure 2. The maximum ratio is then assumed as the 'concentration factor' which needs to be applied to extrapolate personal exposures from SAS data.

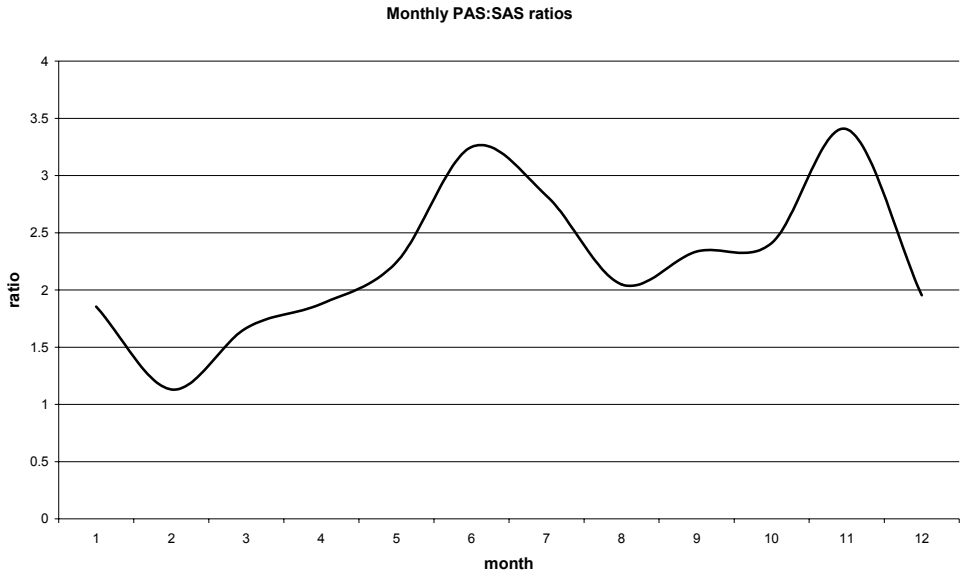

Figure 2: Comparison of SAS and PAS measurements

The next step is to determine the variability of air activity levels within the facility; this is performed by comparing the statistics of the results obtained from individual sampling locations to those from the combined results for all sampling locations. This is presented in Figure 3: the light area in the centre of the plot represents a confidence interval that measured air activity at the specific sample location is within +/- 3 standard deviations of the expected value, as determined from the results from all sampling locations.

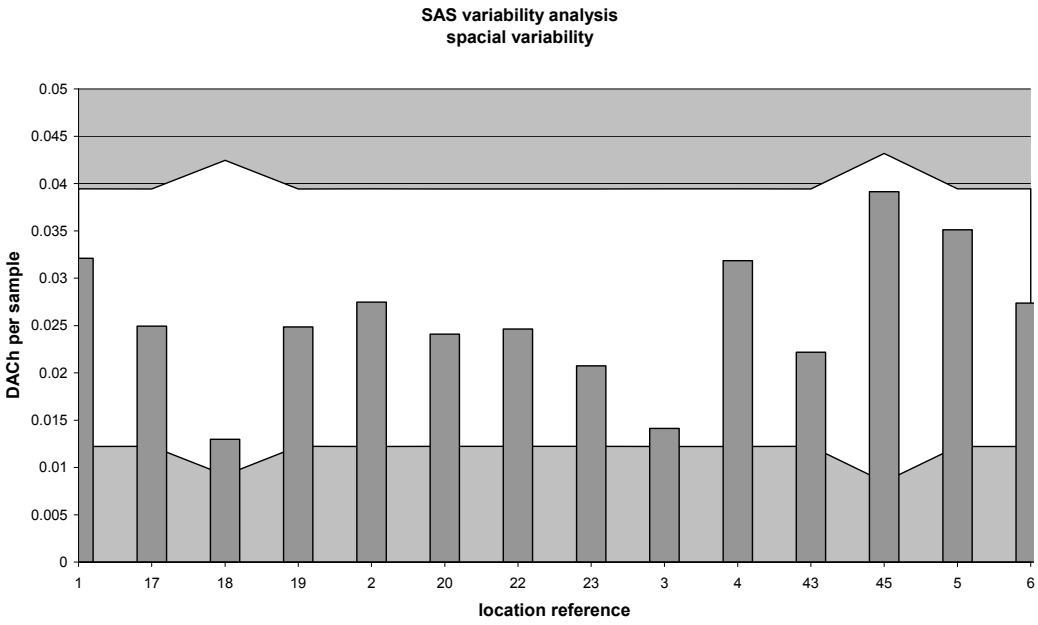

Figure 3: Spatial variability in SAS measurements

If this analysis had indicated that some sampling locations provided results outside the range of the confidence interval (i.e. in the shaded area) then these locations are treated as separate areas and will require specific risk assessments. A similar process is applied for the temporal variability by reviewing the monthly variations for each sampling location.

The final step is to determine the potential dose uptake to a 'nominal worker' who has a full time occupancy of 2000 hours per year within the facility. This is calculated from the monthly SAS data modified by the PAS:SAS ratio, from Figure 2. This is shown in Figure 4, which also depicts a 'threshold' line that equates to a pro-rata annual dose of 1mSv.

From this analysis we can claim that it is highly unlikely that any worker in the facility (in general areas and operations) will be exposed to a dose level which would require to be routinely monitored. This was already known (from past dose records) but this analysis allows the use of SAS data to provide an indication of potential exposure levels even if routine individual monitoring is discontinued. The continuous (e.g. monthly) re-analysis of the SAS data will provide a validation of the risk assessment. Periodic campaigns (e.g. annually) of routine PAS use are also recommended to provide re-validation of PAS:SAS ratios.

At the present this is still, to an extent, a qualitative process: further developments of this process are intended to provide a greater quantitative analysis, such as more rigorous treatment of uncertainties (in both measurements and analysis), which will enable clearer statements of confidence levels to be claimed.

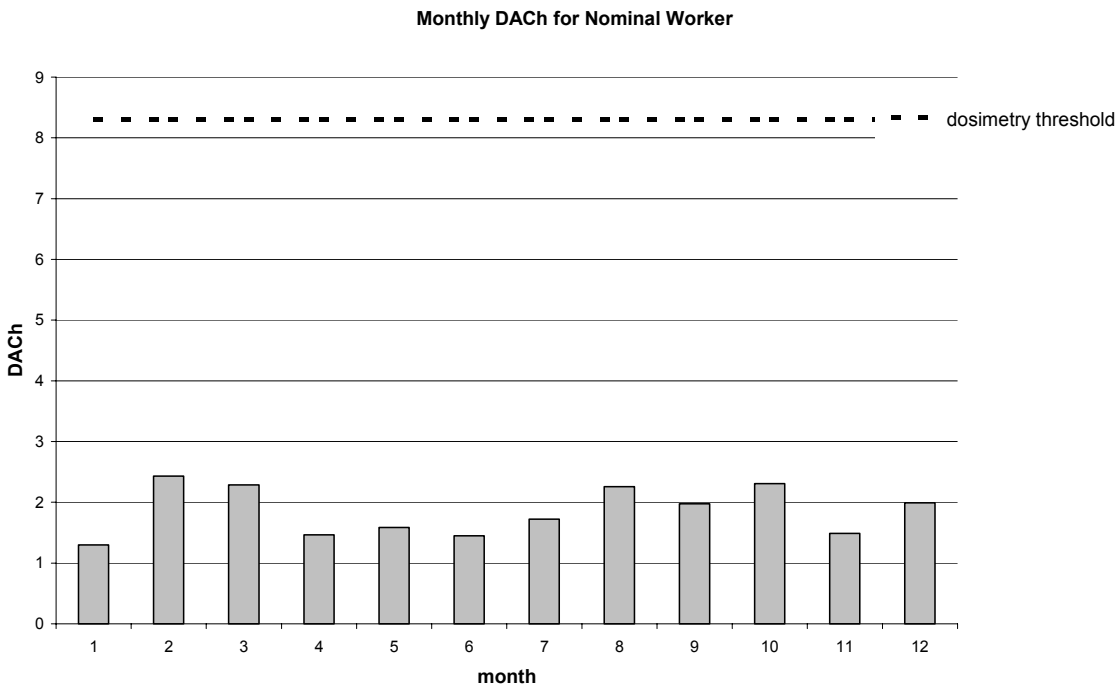

Figure 4: Estimated dose for 'nominal worker'

DISCUSSION

The hazard and risk assessment processes described in this paper are currently only partially implemented; further development is necessary, particularly with regard to the formulation of a more rigorous determination of confidence levels (for example). It is not considered that any of the different approaches discussed are necessarily better than the others (and there may well be other approaches which are equally valid, but not considered in this paper). The decision on which approach is the most appropriate is highly dependent on the circumstances of any specific application.

In any approach for performing hazard and risk assessments the ultimate objective is to be able to make some statements of confidence about the level of 'expected' dose uptakes, without the need to implement potentially costly routine monitoring programmes to actually estimate the dose uptakes. This objective is in accordance with the current principles as defined by ICRP[3] and also with some of the concepts included within a recent draft ISO document[10], particularly the use of workplace monitoring and 'confirmatory monitoring' as indirect means for assessing the risks to workers. These assessments are essentially estimations of the risks of exposure as presented by the workplace, rather than estimations of doses likely to be received by the worker; and would therefore appear to be in accordance with the concept of source-based dose constraints, as discussed in the 2005 draft recommendations of the ICRP[11]. Any statements of confidence arising from such risk assessments are statements of confidence in the risk assessment itself, not statements about the confidence in a dose estimate – e.g. it may be possible to claim 90% confidence that the risks associated with a workplace will not incur doses above 1 mSv; but this will not necessarily allow a claim to be made about the actual 'expected dose' for a specified group of workers.

The determination of confidence in a process requires the consideration of the uncertainties in the process. If we are only required to estimate the uncertainties

in the hazards and risks presented by the workplace (i.e. source-based dosimetry) and not those due to the response of an individual to an exposure (i.e. personal dosimetry) then the potentially very difficult estimation of uncertainties in biokinetic models and parameters etc is not particularly relevant and the use of reasonable but generic defaults is sufficient. The estimation of uncertainties of particular relevance to the source may be easier, and less costly, to determine. This situation would be reversed above higher dose levels – e.g. 6 mSv – where the uncertainties specific to the individual are of prime importance. This implies that the purpose of a dosimetry programme needs to be clearly defined, which will include statements on the levels of dose for which the programme is suitable; the treatment of uncertainties; the claims that can made from the output of the programme, including the desired levels of confidence which is required before any statements can be made[12]. (This is crudely analogous to the need to define the performance specification for an external dosemeter – if a dosemeter is exposed to conditions, e.g. photon energies, outside the scope of its specification it will still report a dose, but at best it won't be meaningful and at worst it will be misleading.)

There may well still be significant costs involved in attempting to obtain a reliable characterization of the source-related uncertainties, and in some circumstances it may well be more cost-effective to implement individual monitoring, even if it may not be strictly required (e.g. for small groups of workers). Further development of some of the concepts expressed in this paper may help aid the decision process to identify cost-effective solutions that also enhance radiation protection.

ACKNOWLEDGEMENTS

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The information presented in Table1 and Figure 1 has been reproduced from the OMINEX report [2] by kind permission of the authors.

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