quantified by counting the combined emissions of both ^{137}Cs and ^{134}Cs, it is unlikely that the latter isotope contributed significantly to the counts observed. The ratio of ^{137}Cs to ^{134}Cs has been shown to be approximately 20:1 or more in waterfowl from the Par Pond system (Ma70).

Results and Discussion

Using combined data from the 180 birds collected during December–February of the two sampling periods, a multiple analysis of variance based on partial sums of squares, revealed no significant effects of sex ($P = 0.08$), age ($P = 0.93$) or body weight ($P = 0.34$) in determining levels of radiocesium contamination. The only factors which had significant effects in determining radiocesium concentrations were the location of collection within the reservoir ($P < 0.001$), and the month ($P < 0.001$) and year ($P < 0.01$) of collection. As discussed in a previous study (Br73), the wintering Par Pond coot population showed a progressively unbalanced sex ratio from fall through winter, with nearly 87% of all birds sampled by February being males. This unbalanced sex ratio in late winter and a relative scarcity of birds in the 3-yr and older age class, made it impossible to evaluate second and third order interactions involving sex and age.

As in the previous years' study, there was a significant interaction between location and month ($P < 0.05$), probably due to the fact that migrant birds from the more highly contaminated North Arm generally came to show higher body burdens of radiocesium with passing months than did birds from the other 2 areas (Fig. 2). Even though sampling areas were less than 4 km apart and were all interconnected with continuous open water, birds collected from 3 arms of the reservoir showed different patterns and levels of contamination, thus suggesting that coots tended to move around the reservoir relatively little once they arrived for their winter stay. This characteristic of the coot would make it an ideal species of waterfowl for purposes of monitoring and locating an unknown source of contamination within an otherwise relatively homogeneous habitat.

No significant effects upon radiocesium body burdens were shown by the interactions of location × year ($P > 0.05$), year × month ($P > 0.05$), or year × location × month ($P > 0.05$), thus indicating that the spatial pattern of radiocesium distribution between coots from the 3 arms of the reservoir did not change over the 4-yr period bracketed by sampling. Rather, as indicated in Fig. 2, there was a tendency for coots from all areas sampled to decrease proportionally in radiocesium

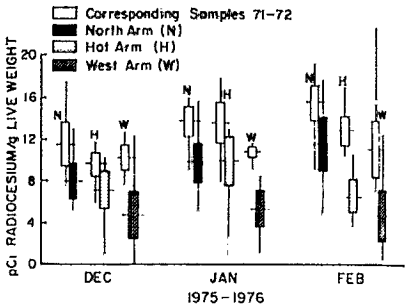

FIG. 2. Whole-body burdens of radiocesium in American coots sampled on the Par Pond reactor cooling reservoir of the U.S. Department of Energy's Savannah River Plant. Horizontal lines represent means of samples of ten birds each, while rectangles represent ± two standard errors and vertical lines represent the range.

contents between years, with no evidence of any increases in body burdens as a result of continuing reactor operations and effluent introductions into the Hot Arm during the 4-yr period of study.

Coots and other members of the family Rallidae are frequently used as food by hunters (Ri76). They could thus represent biota of possibly direct concern to human health and welfare if a major contaminating event should ever occur in an aquatic habitat such as the Par Pond reservoir. If such an event should ever result in waterfowl body burdens that would indicate a risk to human health from consumption, there is at present no available information that would allow an accurate prediction of the time that would have to pass before birds from such an area would once again be safe for hunter harvest. We feel that approaches such as those used in the present study could eventually provide a data base from which such predictions could be made—particularly so, to the extent that such studies could be extended over even longer periods of time.

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The Estimation of Airborne and Surface Contamination Resulting from the Deposition–Resuspension Process

(*Received* 7 *May* 1981; *accepted* 29 *September* 1981)

AN ESTIMATE of the total radiation exposure following an atmospheric release of radionuclides should account for several pathways, including direct atmospheric transport from the release point, contamination of the terrestrial environment by deposition from the airborne contamination, and resuspension of the deposited material and continued atmospheric transport. However, both Lassey (La80) and Kocher (Ko80) have recently commented on the inability of current deposition–resuspension models to predict credible airborne concentrations of resuspended contamination. They specifically comment on the large variance in observed values of the resuspension factor, which relates resuspended airborne contamination to local surface contamination, and they conclude that more observations are necessary for better specification of the resuspension factor.

The intent of this note is to additionally call attention to the theoretical inadequacy of the resuspension factor model. As a consequence, it is recommended that further observations focus instead on the relation between the vertical resuspension flux and the surface contamination. Until adequate empirical data become available, it is further recommended that estimates of the annual-average resuspended airborne contamination be based on the results of Horst (Ho79), which are independent of the values of the resuspension parameters.

The Resuspension Factor Model

Air concentrations of resuspended contamination C are commonly predicted by multiplying the surface contamination G by a resuspension factor K,

$$C(\text{quantity/m}^3) = K(\text{m}^{-1})G(\text{quantity/m}^2). \quad (1)$$

The surface contamination is calculated only from the deposition of airborne contamination transported directly from the original source (i.e. not accounting for the deposition of resuspended contamination). The resuspension factor is assumed to be spatially uniform and to decay exponentially with time at a "weathering" rate λ_K. This method of accounting for resuspension is based on very limited data and does not properly describe the spatial distribution of resuspended contamination. It is best applied, if at all, only in the immediate vicinity of the maximum surface contamination.

Measurements of resuspended airborne contamination are currently limited to a very small number of circumstances, notably the arid environment of nuclear test sites. Even for these particular environments, the resuspension parameters have a large uncertainty associated with them. Observed values of K for resuspension

by the wind range from $10^{-10}\,m^{-1}$ to $3\times10^{-4}\,m^{-1}$ and, since they vary by several orders of magnitude even within a single set of observations, taking an average resuspension factor is not justified (SE80). Estimates of λ_k vary from $7.2\,yr^{-1}$ to $0.68\,yr^{-1}$, but in controlled experiments Sehmel (Se80) has also found no reduction of the resuspension with time, i.e. $\lambda_k = 0$. Lassey (La80) and Kocher (Ko80) have recently discussed these uncertainties and note that as a consequence it is not even possible to determine the importance of resuspended contamination relative to the airborne contamination transported directly from the original source. Slinn (Sl78) concludes that values of the resuspension parameters are "almost totally unknown."

A second weakness of the resuspension factor model is that it does not properly account for the dependence of the resuspended contamination on the spatial distribution of surface contamination or for the dependence on source–receptor separation, because it relates airborne contamination to the local surface contamination, rather than to the upwind distribution on contamination (He77). Observations of the dependence on upwind, rather than local, surface contamination were noted by Stewart (St67) and this defect may, in fact, account in part for the variability of measured resuspension factors. Horst (Ho77) demonstrates that for uniform surface contamination the ratio of resuspended contamination to surface contamination increases with distance from the upwind edge of the contaminated area. For non-uniformly contaminated areas, this ratio increases slowly as the peak surface contamination is approached from the upwind side and increases very rapidly with downwind distance from the peak. Thus in most cases a uniform resuspension factor will not predict the correct spatial distribution of resuspended contamination.

The Resuspension Rate Model

The latter weakness of the resuspension factor can be illustrated by examining the predictions of a more realistic model which assumes that the vertical resuspension flux F is proportional to the local surface contamination through a resuspension rate Λ,

$$F(\text{quantity/m}^2\text{-sec}) = \Lambda\ (\text{sec}^{-1})\ G(\text{quantity/m}^2). \tag{2}$$

This model calculates the resuspended contamination by accounting for atmospheric transport and diffusion between the receptor and the upwind distribution of surface contamination (Ho7 Ho79).

The predictions of this model are shown sch matically in Fig. 1 for the situation where a loca ized source produces annual-average, airbor contamination C at breathing level. There are a scales on the axes of Fig. 1 because its purpose i to show the relationships among the various pr dictions of airborne contamination; actual ma nitudes of the curves depend on the diffusio climatology of the site, source height, depositio velocity, resuspension rate, etc. The various C an averaged over all wind directions and hence an functions only of the distance r from the source With no deposition or resuspension the airborn contamination is C_0; with deposition the direct airborne contamination is reduced to C_d.

The surface is contaminated by deposition both from the direct airborne contamination and from the resuspended contamination C_r. If all of the surface contamination is assumed to be available for resuspension, i.e. neglecting losses by weathering, the resuspended contamination is maximized. The surface contamination increases with

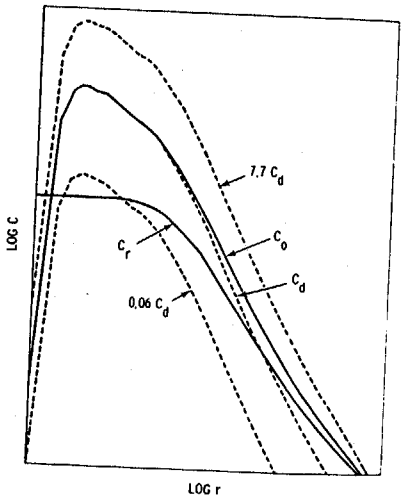

FIG. 1. Annual-average, crosswind-integrated airborne contamination at breathing height as a function of distance from the original source. The resuspended contamination is predicted to be C_r by the resuspension rate model, 0.06 C_d and 7.7 C_d by resuspension factor models.

time until resuspension balances deposition and $C_d + C_r \approx C_0$ (Ho79). More exactly, the resuspended contamination is generally found to be less than $C_0 - C_d$, the deposition loss from the direct airborne contamination. An exception to this conclusion may be expected in the immediate vicinity of an elevated source where the direct plume does not reach the surface. In this case the resuspended contamination is sustained by surface contamination which has been deposited in the past on the opposite side of the source.

The relative magnitudes of the resuspended contamination as predicted by the resuspension rate model and by a resuspension factor model depend on the particular values of K, λ_k and deposition velocity v_d. However, some general conclusions follow from a comparison of the dependence of the predictions of the two models on distance from the source. Resuspension factor models predict the resuspended contamination to be proportional to C_d. Kocher (Ko80) has calculated the proportionality factor for several models using equation (3), which ignores radioactive decay:

$$C_r/C_d = v_d\int_0^{t_e} K(t_e - t')\,dt'. \tag{3}$$

His results, for a continuous release time t_e of 40 yr, range from 0.06 to 7.7 and are included in Fig. 1. A comparison of these curves with C_r predicted by the resuspension rate model shows that a resuspension factor model will agree with a resuspension rate model at no more than two distances from the source. This follows from the relative shapes of the curves for C_d and C_r, which are determined by the diffusion–deposition process and are independent of the resuspension parameters. A resuspension factor model will not reproduce both $C_r \ll C_0 \approx C_d$ near the source and $C_r \approx C_0 \gg C_d$ far from the source.

Discussion

The resuspension factor model is indeterminate due to a lack of observational data. Further, the resuspension factor model predicts the wrong dependence of resuspended contamination on distance from the source. It is best applied, if at all, only in the immediate vicinity of the maximum surface contamination where the local surface contamination may be more important than upwind contamination in determining resuspended airborne contamination. The resuspension rate model is more realistic, but unfortunately is similarly indeterminate, again due to a lack of data on resuspension and weathering rates. It is recommended that future measurements focus on the relationship between resuspension fluxes and surface contamination.

In contrast to the resuspension factor model, the resuspension rate model established an annual-average upper limit for C_r, i.e. $C_r < C_0 - C_d$. Even though it is conservative, this estimate may in many cases be less than resuspension factor predictions. Therefore, until adequate resuspension parameter data become available, it is recommended that the *annual-average airborne contamination*, $C_d + C_r$, be estimated for a continuous source simply by neglecting dry deposition from the direct airborne contamination. This estimate must be modified to account for resuspension immediately below an elevated source. Here the airborne contamination can be conservatively estimated to be equal to the peak value of C_0. For a source of finite duration, the *total exposure* (or time-integrated concentration) may be estimated in a directly analogous manner.

In addition to simplifying the estimation of resuspension, the neglect of deposition also eliminates the need to estimate plume depletion, $C_0 - C_d$. The simplest methods for predicting the reduction of airborne concentrations by dry deposition are approximate, and more exact methods are computationally complex (Ho80). The present recommendation eliminates the need to deal with this matter.

Deposition must be estimated, however, to account for exposure via terrestrial pathways. The present recommendation assumes that the surface contamination increases by dry deposition, from both the direct and the resuspended airborne contamination, until resuspension balances deposition and $C_d + C_r \approx C_0$. Thus the surface contamination is conservatively estimated as

$$G \approx v_d C_0 \frac{(1 - e^{-\alpha t})}{\alpha},\ t < t_e \tag{4a}$$

$$G \approx v_d C_0 \frac{(e^{-\alpha(t-t_e)} - e^{-\alpha t})}{\alpha},\ t > t_e \tag{4b}$$

where the release begins at time zero and ends at time t_e, and $\alpha = \Lambda + \lambda$ is the sum of the resuspension rate and the radioactive decate rate λ. Unfortunately there are few measurements of Λ and these vary from $10^{-12}\,sec^{-1}$ to $10^{-4}\,sec^{-1}$ (Se80). In view of this large uncertainty it seems prudent to take no credit for these losses, i.e. $\Lambda = 0$

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Gamma Activity of Some Building Materials in West Malaysia

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Introduction

LOW-LEVEL exposure from γ sources in buildings has received considerable recent attention (Ko78; Ni80; Ka80). The indoor absorbed dose rate which varies from about 4 to 12 μrad/hr, depending on the type of dwelling and location, contributes on the average about one quarter or more to the average annual dose from all sources of radiation (UN77). As part of the programme on environmental radioactivity, we have undertaken to measure the γ activity of a variety of building materials used in Penang so that some estimation of exposure dose rate may be obtained.

Method

Samples of cement, bricks and other building materials were collected from different building sites and suppliers. The materials collected were ground to powder and then oven-dried at 70°C. The γ spectrum was obtained using a Ge(Li) detector coupled to a 1024 pulse height analyser. Counting time ranged from 500 to 1000 min. Details of the equipment and shielding were as reported by Lowe (Lo78). The peak efficiency of the detector was measured using ^{131}I mixed with low activity sand in the same counting geometry and also with standard ^{226}Ra and ^{232}Th point sources. The γ activity was calculated from the total count under selected photopeaks.

Results and Discussion

Table 1 shows the results of the measurements of the activity concentration in the various materials studied. Although secular equilibrium among the member nuclides in the uranium or thorium series was assumed, the measured photopeak intensities nevertheless indicated certain degree of disequilibrium within each sample. The radium and thorium activities for each sample are mean values obtained by averaging over the selected photopeaks. Although not indicated in Table 1, the samples of each type of material show appreciable variation. The coefficient of variation of limestone and cement bricks are in particular quite high. Part of the reason for this variability may be due to the small number of samples and to sample origin. We intend to extend our measurements to more samples from other parts of West Malaysia.

Table 1. Concentrations of ^{40}K, ^{226}Ra *and* ^{232}Th *in building materials in Penang*

Material	Number of samples	Average activity concentration (pCi/g)		
		^{40}K	^{226}Ra	^{232}Th
Cement	5	5.5	2.2	1.6
Cement brick	4	8.7	2.3	1.4
Cement tile	2	9.7	1.6	1.4
Gypsum	1	5.5	- negligible -	
Mosaic tile	3	9.0	6.4	4.4
Granite chip	4	30.0	6.4	6.2
Iron ore	4	6.0	7.2	2.5
Shale	4	23.0	4.3	3.8
Red brick	3	18.5	6.3	6.2
Sand	5	11.5	1.9	0.9
Lime stone	3	6.0	1.1	0.7

One interesting point arising from the measurements is the relatively higher radium and thorium activities in most of our samples than those reported for the European region while the potassium level is of the same magnitude (see UN77). The cement samples (including tiles and bricks) and gypsum show similar activity levels as those reported. However, in other samples, the radium and thorium activities are generally two to three times higher.

It is worth pointing out that in Penang as well as in other parts of Malaysia, the majority of the urban dwellings built after the late sixties have cement brick walls. This implies a substantial reduction in exposure from those dwellings built with red bricks.

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