IMPACT OF THE CHERNOBYL FALLOUT IN THE ALPINE ENVIRONMENT

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INTRODUCTION

In Austria the alpine regions received the highest fallout contamination, showing a very inhomogeneous spatial distribution of the surface deposition. About half of the national territory is within alpine regions, which are very different in terms of underlying bedrock and soil characteristic. Since this is the controlling factor for the radionuclide uptake of the vegetation, it is crucial for the long-term effects of radioactive fallout. Different studies have been carried out in the Province of Salzburg (area: 7154 km$^2$) over the past ten years, addressing a broad spectrum of issues, such as: measurement of the spatial distribution of the fallout, research in monitoring techniques, comparison of theoretical calculations with actual in vivo-measurements of nuclide uptake by man for different population groups, and the investigation of biological effects [1,2,3,4,5].

When considering the radioecological effects of the Chernobyl fallout a distinction has to be made between the short-term effects immediately following the fallout and the long-term effects. While the short-term effects are controlled by the physical characteristics of the fallout, similar for the whole region, the long-term effects are more determined by the radioecological properties of the environments affected which are much more variable than the fallout-characteristics.

SITUATION IMMEDIATELY FOLLOWING THE CONTAMINATION

The hot particle issue

The controversy about hot particles was initiated by Tamplin and Cochran [6] who contended that inhalation of hot particles containing a certain amount of alpha emitters represent a substantially higher lung cancer risk than if the same amount of radionuclides were distributed among smaller sources. While Mayneord and Clarke [7,8] also estimated a higher risk for $^{103}\text{Ru}$ hot particles at sufficiently small activities, they predicted a reduced risk for high activities relative to a uniform nuclide distribution. More importantly, however, they also observed the same tendency for beta emitting hot particles ($^{86}\text{Rb},^{35}\text{S}$). In the wake of the Chernobyl incident, Hohenemser et al. [9] raised again the question whether a $^{103}\text{Ru}$ hot particle is indeed more carcinogenic than uniformly distributed nuclides of the same total activity.

In the Chernobyl fallout hot particles of mostly $\beta$-emitting radionuclides have been detected in several countries varying considerably in radionuclide composition, activity, size and shape. These hot particles can be grouped into two categories: (i) particles originated in a process of condensation of vapours of ruthenium, consisting of pure $^{103}\text{Ru}$ and $^{106}\text{Ru}$ (category A); and (ii) fuel fragments containing, besides the ruthenium isotopes, $^{141}\text{Ce},^{144}\text{Ce},^{95}\text{Zr},^{95}\text{Nb},^{135}\text{Cs}$ and $^{137}\text{Cs}$ (category B). For the assessment of the radiological risk associated with the inhalation of hot particles in the Austrian Alps, data reported from Austria [10,11], Germany [11,12], Hungary [13] and Switzerland [14] had to be used since no systematic measurements of hot particles have been carried out in this region. Based on these measurements, the following assumptions were made concerning activity, concentration in ambient air, and size: (i) hot particles are composed entirely of $^{103}\text{Ru}$ with activities of 30, 300, and 3000 Bq, reflecting the distribution of hot particles measured in Central Europe [5,7,8,9]; (ii) the average hot particle concentration is $1\times10^{-3}$ m$^{-3}$, reflecting observed concentrations in the range of $5\times10^{-5}$ to $5\times10^{-2}$ [8,9]; and (iii) the average geometric diameter of the hot particles is 1 $\mu$m [6,9].
For the risk assessment of hot particles via the inhalation pathway, an average respiratory minute volume of 15 \text{ l min}^{-1} and mean exposure time of one week was assumed. Based on a deposition probability of about 20\% for 1 \mu m particles in the pulmonary region (note: particles deposited in the bronchial region are cleared to the GI-tract within a couple of hours), the probability to inhale and deposit a hot particle in the deep lung is about 0.03. Since the probability to find more than one hot particle in the lung is very small (for two hot particles it is $4\times10^{-6}$), higher concentrations of hot particles in the inhaled air will increase the number of individuals inhaling a hot particle rather than the number of particles deposited per person.

For insoluble $^{103}$Ru particles with clearance half-times of the order of hundreds of days, we assume that a hot particle, once deposited, will be retained in the lung for 1 year. This leads to a cumulative activity of about 17 kBq day for a 300 Bq $^{103}$Ru hot particle integrated over one year.

Lung cancer risk calculations are based upon the initiation-promotion model of carcinogenesis. In this model, initiation is interpreted as cellular transformation in surviving cells, while promotion is related to cell killing in the immediate vicinity of the transformed cells [15]. For a uniform nuclide distribution, lung cancer risk is expressed as a function of the average dose received by each cell. In the case of a hot particle, however, with a radial dose distribution around the hot particle, a few cells close to the hot particle will receive relatively high doses, while the majority of cells will not be hit at all. For these dose calculations, a lung mass of 1 kg with a density of 0.26 g cm$^{-3}$ was assumed.

Calculations of the lung cancer risk as a function of the radial distance from the hot particle indicate that all cells in the immediate neighbourhood will be killed due to the very high doses, so that no tumour can arise. At intermediate distances from the hot particle (between 0.7 and 0.8 mm), the probability for cellular transformation and promotion exhibits a distinct maximum, dropping off sharply at greater distances.

The probabilities for lung cancer induction, obtained by integrating over all radial distances, are given in Table 1 for three selected hot particle activities and compared with the corresponding uniform dose distributions. First, risk is computed for a stationary hot particle, i.e., a particle remains at a given site for one year (denoted e.g. as 1 x 300 Bq). In this case, the 30 Bq hot particle has the highest risk of the three hot particles relative to their corresponding uniform distributions, with a risk enhancement factor of 33. Particles \textit{in vivo}, however, do not remain localized for such long periods of time, but instead are continuously moving throughout the pulmonary region. For example, movement of a single 300 Bq particle may be simulated by the action of 10 smaller 30 Bq particles (denoted as 10 x 30 Bq), which is equivalent to one 300 Bq particle staying at 10 different non-overlapping sites for each 1/10th of a year. In general, particle movement increases lung cancer risk for a given activity, producing the highest risk for the most mobile hot particle. Corresponding calculations for other organs with unit density tissue display a similar pattern of response, though risk enhancement factors relative to the

Table 1: Lung cancer probabilities for stationary and moving $^{103}$Ru hot particles in the lungs. (L) (density = 0.26 g cm$^{-3}$) and in other organs (O) with unit density compared with the corresponding uniform radionuclide distributions.

<table>
<thead>
<tr>
<th>Carcinogenic risk* (arbitrary units)</th>
<th>100 x 30</th>
<th>3000</th>
<th>10 x 30</th>
<th>1 x 3000</th>
<th>10 x 30</th>
<th>1 x 3000</th>
<th>1 x 30</th>
<th>1 x 30</th>
</tr>
</thead>
<tbody>
<tr>
<td>Uniform (L,O)</td>
<td>2.3 x 10$^3$</td>
<td>2.3 x 10$^3$</td>
<td>2.3 x 10$^2$</td>
<td>2.3 x 10$^2$</td>
<td>2.3 x 10$^2$</td>
<td>2.3 x 10$^1$</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hot particle (L)</td>
<td>7.5 x 10$^4$</td>
<td>2.5 x 10$^4$</td>
<td>5.6 x 10$^3$</td>
<td>7.5 x 10$^3$</td>
<td>2.5 x 10$^3$</td>
<td>7.5 x 10$^2$</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hot particle (O)</td>
<td>5.2 x 10$^3$</td>
<td>1.4 x 10$^3$</td>
<td>2.0 x 10$^2$</td>
<td>5.2 x 10$^2$</td>
<td>1.4 x 10$^2$</td>
<td>5.2 x 10$^1$</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

*Additional gamma component risk: 8.8 (30 Bq), 8.8 x 10$^1$ (300 Bq), and 8.8 x 10$^2$ (3000 Bq)
uniform distributions are reduced by approximately one order of magnitude. Thus, if a hot particle is
cleared from the lungs and then trapped in an organ with unit density for a longer period of time (note:
the same applies to ingested hot particles), the hot particle effect is smaller there than in the lungs.
Simultaneously emitted gamma radiation produces a rather uniform distribution of risk even in the case
of a hot particle, which has to be added to the risk of the beta components listed in Table 1.

In conclusion, risk enhancement factors for a $^{103}$Ru hot particle, computed relative to a uniform
radionuclide distribution of equal activity, are highest for small and intermediate activities and for hot
particles moving in the lung. Similar results will be obtained for other beta-emitting radionuclides of
comparable half-life and energy, such as $^{99}$Zr, $^{96}$Nb, $^{140}$Ba, and $^{141}$Ce. While inhalation of these hot
particles might exceed all other exposure pathways of the Chernobyl fallout, its potential lung cancer
risk is still smaller than that for radon progeny inhalation and is thus unlikely to be detected against the
reported natural lung cancer incidence.

Chromosome aberration / Effect on blood chromosomes

An increase of chromosome aberrations in peripheral blood lymphocytes of persons living and/or
working in an environment with elevated radioactivity has been found in various places of the world and
the data indicate that the increase occurs already at rather low doses [4]. Therefore we chose 16
volunteers from Salzburg city for blood sampling during 1987 whose additional external and internal
doses at that time were between 15% and 68% of the total pre-Chernobyl environmental burden
(0.9 mGy/yr). On the day of blood sampling for cytogenetic investigation (June, 9 and July, 27) the
$^{137}$Cs content was measured with a whole body counter. From these results the adequate $^{134}$Cs content
and from the dose commitment to the tissue (UNSCEAR 1982 model: $P_{4.5} = 2.4 \times 10^6$ Gy/Bq x kg$^{-1}$) the
entire internal Cs dose was calculated. The average external gamma dose for Salzburg, based on several
measurements at many different sites and times, was found to be 0.12±0.02 mGy/yr in 1987.

The Cs dose for the volunteers investigated ranged from 0.013 to 0.492 mGy/yr. They were at ages
from 24-69 years, all non smokers, took no drugs and had no diagnostics x-rays within the last year.
The procedure of lymphocyte culture, slide preparation and scoring followed a standard method at that
time with 48hrs culture time. The different coded slides were distributed for scoring to laboratories,
chosen according to their former scoring variabilities being only within statistical errors. Altogether
23060 metaphases were analysed.

Two of the volunteers (a female, age 38, and a male, age 24 years in 1985) were also taken as
controls for other investigations so we had chromosome aberration data from the years 1984/85 and
also 1988 and 1990. Their mean external plus internal additional dose rates in April 1988 and in
January 1990 were 0.16 and 0.09 mGy/yr respectively.

Because of the low-doses and the resulting low effects we summarised for statistical reasons all
chromosomal aberrations (,,totals" = dicentrics [D]+ centric and acentric rings [R]+ interstitial and
terminal deletions [ID, TD]) and pooled the results for several persons. The mean values are weighted
according to the number of metaphases scored. The totals from all volunteers investigated 1987,
including those for the two persons in 1984/85, 1988 and 1990, are given in Table 2 and plotted in Fig
1 [5]. From the table it is obvious that in 1987 an unusual high amount of rings occurred. All of these
rings were confirmed by two different scorers in each laboratory. The dicentrics recorded in 1987 were
between a factor of 3 and 4 higher than the pre-Chernobyl values. Moreover 1987 we found a diploid
cell with 6 ID in 736 metaphases (not included in Table 2a). In the same year in an in-vitro
investigation, applying additional low $\alpha$-dose from $^{214}$Po, three cells with large numbers of ID and TD
were found: one cell out 57 metaphases (additional $\alpha$-dose: 0.38 mGy), one out of 141 metaphases
(additional $\alpha$-dose: 1.65 mGy) and one out of 517 metaphases (additional $\alpha$-dose 420 mGy). Because
these damages usually are expected at higher doses and the additional $\alpha$-dose was low, it might be
assumed, that they were caused by the increased background radiation following the Chernobyl fallout.
Table 2: Mean aberration frequencies per 100 metaphases in blood-chromosomes of Salzburg residents

**a) 16 different blood donors, sampled 1987**

<table>
<thead>
<tr>
<th>Mean additional dose $^{134}$Cs+$^{137}$Cs [mGy/yr]</th>
<th>Nr. of persons (year)</th>
<th>Metaphases scored</th>
<th>Total numbers D R ID+TD</th>
<th>Percentage of aberrations D D+R+ID+TD</th>
</tr>
</thead>
<tbody>
<tr>
<td>internal</td>
<td>external</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0.07</td>
<td>0.12</td>
<td>3 (1987)</td>
<td>4861</td>
<td>13 10 27</td>
</tr>
<tr>
<td>0.14</td>
<td>0.12</td>
<td>4 (1987)</td>
<td>3327</td>
<td>8 9 41</td>
</tr>
<tr>
<td>0.26</td>
<td>0.12</td>
<td>4 (1987)</td>
<td>3847</td>
<td>10 7 40</td>
</tr>
<tr>
<td>0.39</td>
<td>0.12</td>
<td>5 (1987)</td>
<td>3575</td>
<td>7 2 40</td>
</tr>
</tbody>
</table>

**b) 2 different blood donors, sampled prior and after the accident**

<table>
<thead>
<tr>
<th>Mean additional dose $^{134}$Cs+$^{137}$Cs [mGy/yr]</th>
<th>Nr. of persons (year)</th>
<th>Metaphases scored</th>
<th>Total numbers D R ID+TD</th>
<th>Percentage of aberrations D D+R+ID+TD</th>
</tr>
</thead>
<tbody>
<tr>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>2 (1985)</td>
<td>4270</td>
<td>3 0 8</td>
</tr>
<tr>
<td>0.18</td>
<td>0.12</td>
<td>2 (1987)</td>
<td>2907</td>
<td>11 12 32</td>
</tr>
<tr>
<td>0.08</td>
<td>0.08</td>
<td>2 (1988)</td>
<td>756</td>
<td>4 0 3</td>
</tr>
<tr>
<td>0.02</td>
<td>0.07</td>
<td>2 (1990)</td>
<td>2424</td>
<td>1 0 14</td>
</tr>
</tbody>
</table>

Fig.1. Chromosomal aberration in blood chromosomes of 16 different donors as a function of additional internal and external dose received after the Chernobyl fallout.

Fig.2. Chromosomal aberration in blood lymphocytes of 2 different donors as a function of time.

PRESENT SITUATION

Analysis of a large number of soil-depth profiles and lichens as biological monitors, sampled from 1986 to 1995, revealed a rather inhomogeneous nuclide deposition pattern in the Province of Salzburg, varying between 10 GBq/km$^2$ and 80 GBq/km$^2$ [3] for $^{137}$Cs. This inhomogeneity is largely due to different meteorological conditions prevailing at the time of passage of the radioactive cloud.

At present the only remaining radionuclides of significance from the Chernobyl fallout are $^{137}$Cs and, with minor importance, $^{134}$Cs. These nuclides enter the food chain to a varying degree and contribute to the ambient gamma-exposure affecting the dose of the population by these two ways. Due to the half-
life of 30 years for $^{137}$Cs its contribution to the gamma-exposure is still of significance and it has been demonstrated (Fig.3) that though the migration into deeper soil layers proceeds and increasingly attenuates the gamma-flux, the average increase of gamma-exposure was calculated to be 14.2 nSv/h/10 kBq/m$^2$ in 1993 [3]. This causes an additional gamma-exposure ranging between 30% and 100%, of the pre-Chernobyl background. Especially upland regions are affected by this contribution in a twofold manner: (1) The increase of ambient gamma exposure is higher because the surface deposition tends to be above average, and (2) the contribution to ambient gamma-exposure decreases slower because the downward migration rates are smaller than in any other environment (Fig 3). In regions with fast migration into the soil the migration rate controls the decrease of gamma exposure and in regions with small migration rates, the exponential decay of $^{137}$Cs is the more important controlling factor. The effect on the dose to the population therefore is higher in alpine regions than in other regions.

At present the average contamination of the food products with radionuclides from the Chernobyl fallout is very low and of minor radiological significance. This is due to the fact that the majority of the production areas are situated within intensively used agricultural regions where the soil-plant transfer-factor tends to be low. However, there are some regions that show a significantly different behaviour in terms of nuclide transfer. These regions are exclusively situated in elevated alpine areas. They are used for agricultural production only to a low degree, i.e. their contribution to national food production is minor. The term "upland ecosystem" might be most suitable for these seminatural environments, mainly alpine pastures with poor vegetation in agricultural terms and the food production being exclusively dependent upon grazing animals. The main radioecological feature of these alpine pastures is their high transfer-factor, which can be correlated with the bedrock geology controlling the formation of the soil: While in limestone regions the transfer-factors are small and similar to the main agricultural production sites, these factors from the alpine pastures with silicic bedrock geology can be extremely high [2]. In areas with high transfer-factors according to specific soil conditions, the locally produced food can still show significant levels of food contamination. Such areas are distributed over the Hohe Tauern region with the highest elevation in the Austrian Alps, where agricultural production during the summer time is common. The contamination of food products from these areas, which are exclusively based on grazing cattle, can still reach up to 30%-40% of the levels measured in the initial period immediately following the fallout deposition. Some production-sites were selected for continuous sampling of milk during summer. The evaluation of these data shows that the average contamination of milk from these sites decreases with an effective half-life between 3 years and 6 years (Fig.4), depending on the production site. These half-life times differ significantly from the major milk-production areas, where the effective

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**Fig 3:** Soil profile of $^{137}$Cs in two different soil types: (o) Intensively used agricultural soil, (c) Alpine soil from upland ecosystem, extensively used. Data normalised to 1 Bq/cm$^2$.

**Fig 4:** Time-dependent $^{137}$Cs-activity concentration in milk from regions with high transfer factors.
half-lives are between 500 - 800 days [16]. Other agricultural products made from milk and the contamination of cattle are affected in a similar way. The peaking of the contamination is restricted to the summer period because the amount of winter forage produced in the regions, where high transfer factors dominate, is negligible. People mostly affected by the consumption of food with high radionuclide content are a small group of seasonal workers on the production sites. Whole body measurements of this population group from 1991 and 1992 showed that the $^{137}\text{Cs}$ activity concentration is comparable to the average values for Austria in 1987, when the average whole body burden reached its maximum [17]. The measurements were made before the working season and shortly afterwards. In both years, 1991 and 1992, the $^{137}\text{Cs}$ content raised significantly during Summer: on average the values measured after the summer season had doubled, with only low significant differences between 1991 and 1992. For the calculation of the daily intake two constant daily intake rates for $^{137}\text{Cs}$, one before and another constant daily intake rate during the summer working period, were assumed. Under these assumptions the daily intake was about 40 Bq/day in the summer period and 10 Bq/day during the rest of the year. Using 100 days for the summer period and applying a biological half-life for $^{137}\text{Cs}$ of 100 days and a recommended dose conversion factor for Cs [18] the average radionuclide uptake corresponds to an additional ingestion dose of 75 μSv in 1991 and 1992.

In this investigation blood chromosomes of nine workers were recorded prior to and after their summer-stay in the alpine regions. The entire individual radiation burden, consisting of external gamma dose, internal alpha dose (from radon and daughters) and beta dose (from $^{137}\text{Cs}$, $^{134}\text{Cs}$ and $^{40}\text{K}$) was calculated from outdoor, indoor and whole body measurements. The results obtained from this study [19] are indicative of a similar dose-effect relationship as found for the study group of urban dwellers in Salzburg City from 1987, when a significant increase of chromosome aberration after the Chernobyl fallout could be found. In view of the low dose values observed, no biological effects are to be expected, with the exception of chromosomal aberrations.

**CONCLUSIONS**

In the period immediately following the Chernobyl fallout deposition in the Alpine region, hot particles as a dominant factor for the dose to the public considered. Due to their high risk enhancement factors relative to a uniform distribution, inhalation of hot particles might exceed the significance of all other exposure pathways of the Chernobyl fallout.

Chromosome aberrations, following low dose irradiation of the Chernobyl fallout, could be observed. Considering long-term consequences, regions with high transfer factors are affected by a prolonged period of increased uptake of radionuclide into the food chain. In the alpine region of Austria these regions can be found at elevated sea-levels within seminatural upland ecosystems of poor agricultural use. The effective half-life of the average contamination with $^{137}\text{Cs}$ in food products from these regions varies between 3 and 6 years.
The population group mainly affected are the workers on the seasonal production sites. Whole-body counting data from 1991 and 1992 are in the same range as the highest average values in Austria shortly after the Chernobyl fallout. Average individual dose values are 75 μSV in 1992 and can be considered relatively low.

REFERENCES


