Radiation Hazards

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Much has been written about nuclear power and its possible effects on man. Comparatively less attention has been paid to other living organisms except in so far as they form part of a food chain, contaminated by radionuclides and leading to man. Indeed, some workers have gone so far as specifically to exclude from their work the well-being of animals. Others have come to believe that if we take as our safety levels those amounts of radioactivity in the natural environment that are an acceptable risk to man, other organisms will not suffer significantly.

Awareness of radiation hazards in medicine can be said to have passed through three stages, though, of course, in reality the stages have overlapped.

1. Some of the hazards of somatic effects upon individual workers with radiation were recognised very soon after the discovery of X-rays and of natural radioactivity. Large doses of the order of hundreds of thousands of R of external radiation cause obvious and rapid tissue destruction, skin burns being the most easily observed. The haemopoietic tissues, however, are more sensitive and their activity may be temporarily depressed or even permanently altered by much smaller doses.

   The carcinogenic effects were also recognised fairly soon, though it took many years to establish the fact that ionising radiation of only a few hundred rads might, often after a long latent period, cause malignancy.

2. The second stage came with recognition of an especial danger to the developing ovum. It was found that the frequency with which malignancies developed within the first ten years of life was increased, probably by about 40 per cent, in children exposed to diagnostic X-rays while in utero (Stewart et al., 1956; MacMahon, 1962). The incidence of such malignancy was considered by UNSCEAR (1972) to correspond to fetal absorbed doses varying from 0.2 to 20 rad, with a mean of less than 2 rad, and to a risk of fatal malignancy of about $23 \times 10^{-6}$ per rad per year to the age of 10.

3. The third stage came when it was realised that, since ionising radiation can be shown to cause mutations in developing cells, there might be a genetic risk of deterioration in the population as a whole. This danger is related both to the fact that the mutation rate may be linear in relation to dose, with no lower threshold, and to the high proportion of the population
exposed, especially to diagnostic X-ray examinations, in wealthier and more developed countries. It presupposes that most or all mutations are detrimental.

Thus, during this century, concern has embraced first radiation workers, then patients and their unborn children, and finally whole populations. Recommendations for the use of ionising radiation have become steadily more stringent, and maximum permitted dose levels have steadily fallen.

The great attention paid to the effects of radiation has few parallels in safeguards sought against other natural or man-made hazards. Many of man's activities are far more dangerous and far less well controlled. Nevertheless, it is right to be concerned. The level of background radiation is rising and nothing learnt about the possible effects of radiation is comforting. It now seems likely that thyroid cancer may occasionally follow radiation of the order of 10 rads (Modan et al., 1974) in about $\frac{1}{3}$ or $3 \times 10^3$ of those irradiated.

Because we value the individual, it matters that maternal irradiation may carry a risk of about $23 \times 10^{-6}$ per rad per year to the age of 10 for the child. Human longevity makes things worse, for man lives through the latent periods of delayed effects. Part of our concern for individuals is perhaps due to the paucity of our brood compared with some other species and the corollary, the high survival rate of the human fetus.

An opposing factor separating man from some other species is his adaptability to variations in environment and his wide distribution over the earth, both of which are safeguards. Man need not be sedentary like a bivalve in the stream of contamination. Even if a whole village or country became affected by radiation-induced genetic deterioration, in a few generations emigration and mixing could swamp the defects within what is virtually a world-wide genetic pool. To draw an ornithological parallel, there should be less worry about genetic damage to a group of house-sparrows or herring-gulls than to a group of Dartford warblers constrained by their special and restricted bushy heathland.

In approaching the subject of radiation effects on non-human organisms, one must be careful to balance the usefulness of the industrial process against any possible damage it may do.

In spite of great efforts there is still not a lot to report of the known effects of the nuclear power programme upon living organisms other than man. The interactions of ecology are so complicated that speculation built upon isolated findings is generally mistaken. Only prolonged observation will do. Much, also, of what is believed depends upon extrapolation in a linear way from the experimental effects of larger doses of ionising radiation to the so far unobserved effects of very small doses. This linearity may not be accepted, though the burden of evidence seems to be edging towards it.

The list of radionuclides potentially released into the atmosphere, lakes, rivers, sea, or on to dry land as a result of all the processes associated with the generation
of power from nuclear energy, is very long, and it is not yet possible to forecast accurately the exact inventory that would be derived from any new installation. A convenient list of those likely to be more important after an accidental release has been given by the MRC committee (1975) which drew up criteria for controlling radiation doses to the public after accidental escape of radioactive material.

However, the first consideration is the radionuclide burden derived from the routine running of a nuclear power programme.

An important paper is that of Blaylock and Witherspoon (1975) which outlined the effects upon aquatic living organisms of radioactive releases from the different processes in the enriched uranium dioxide fuel cycle. These processes are:

- uranium mining,
- uranium milling,
- conversion of uranium oxide to uranium hexafluoride,
- uranium enrichment,
- fuel fabrication,
- the operation of light water reactors,
- fuel reprocessing.

Far greater releases occur at the mining and milling stages than at any subsequent operation. Uranium mining from a typical model mine releases $^{234}\text{U}$, $^{235}\text{U}$, $^{238}\text{U}$, $^{226}\text{Ra}$, $^{230}\text{Th}$, $^{210}\text{Pb}$ and $^{210}\text{Po}$. The radiation involved in mining and milling is shown in Table 1.

| Aquatic plants | 63 | 1,200 |
|----------------|----|-------|
| Invertebrates  | 100| 350   |
| Fish           | 1.2| 22    |

The next three stages, conversion, enrichment and fuel fabrication, produce a different list of radionuclides, but, in total, not more than about 4 rads/year.

At the reactor site, the potential list is very long, tritium being by far the greatest source of radioactivity in the surrounding environment.

The radioactivity of effluent from a pressurised water reactor differs from that of a boiling water reactor. The radioactivity of the effluents also differs according to their disposal into fresh or saline water (Table 2).
Table 2. Waste disposal from reactors (Blaylock and Witherspoon, 1975).

| Effluent discharged into: | Pressurised Water Reactor | Boiling Water Reactor |
|---------------------------|---------------------------|-----------------------|
| Radioactivity in:         | Fresh water rads/yr       | Saline water rads/yr  |
| Aquatic plants            | 0.12                      | 1.9                   |
| Invertebrates             | 0.079                     | 4.6                   |
| Fish                      | 0.12                      | 0.39                  |
| Algae                     | 6.9                       | 0.23                  |
| Molluscs and crustaceans  | 1.4                       | 29.0                  |

Finally, fuel reprocessing is said to be responsible for the release of only a few millirads, but there is room for doubt.

The expected effects of these doses, the highest overall coming from $^{226}$Ra, $^{210}$Po and $^{230}$Th, upon aquatic organisms have been briefly reviewed by the same authors and more extensively by Templeton et al. (1971), Auerbach et al. (1971) and Ophel et al. (1974).

The largest doses are received in habitats that should be within the restricted zone of the facility forming part of the radioactive waste disposal system.

EFFECTS OF RADIATION

Chronic radiation dose rates of 3 to 4 rads a day within the range produced by uranium milling operations cause slight growth inhibition in higher plants such as conifers, but lower plants such as algae have a higher degree of radio-resistance.

At Oak Ridge National Laboratory (specifically in White Oak Lake, which was an area of radioactive waste disposal) natural populations have been exposed to chronic radiation for many generations. The dose rate to the snail Physa Heterostropha at the time of the study was 0.65 rads/day, but it had been higher in the past. There was a change in the number and proportion of eggs and egg capsules per snail, but in laboratory experiments it was found that doses in excess of 1 rad/hour were required to produce any observable somatic effect. The frequency and kind of chromosome aberrations observed in midge larvae (Chironomus) were compared with a control population. The calculated dose was 230 rads/year. More chromosome aberrations were being produced in the irradiated populations than in the controls. However, these did not persist but were eliminated by selection and genetic drift.

In the lake, the mosquito fish, which received an estimated dose of 11 rads/day, had a significantly greater number of dead and abnormal embryos than did the control population (Blaylock, 1959). However, it required 1.3 to 5.4 rads/hour in the laboratory to produce atrophy of the testes (after 18 days), so it would be very hard to detect any somatic changes resulting from the maximum
exposure of 22 rads/year in uranium milling. Moreover, because less than 1 per cent of the zygotes of many aquatic organisms mature and reproduce themselves, it is most unlikely that there would be a cumulative genetic deterioration except, perhaps, in the immediate vicinity of uranium mining and milling.

Roushdy et al. (1975) also carried out experiments on fish that might be exposed to effluent in fresh water, taking the Nile Cat-fish *Clarias lazera*, a carnivore, and *Tilapia nilotica*, a herbivore, and subjecting them to different regimes of irradiation from $^{60}$Co sources. They found that in *Clarias lazera*, 100 R given at 5 R/min temporarily reduced the level of haemoglobin and blood cells. Higher doses of 400 R caused a persistent anaemia. Larger doses of 500 and 1,000 R caused metabolic changes in Tilapia. Both species showed alterations in weight and growth rate.

These doses were given acutely by external radiation. Even 100 R is of the order of 25 times the dose of internal radiation received by invertebrates from an average boiling water reactor and 400 times the dose received by fish in a year. Nevertheless, the demonstration of metabolic and somatic damage with doses of this order brings the possibility of man-made ecological change within the compass of the waste disposal associated with substantial light water reactor building programmes.

HAZARDS OF WASTE DISPOSAL
As to the reactors themselves, Hermann et al. (1975) examined the behaviour of radioactive waste products from a small boiling water reactor of 237MW at Gundremmingen on the ecological system of the Upper Danube. Before it reached the Danube, the effluent water (mixed with the cooling water) contained 25 nuclides. Ninety-nine per cent of the total activity came from tritium and amounted to just under 134 Ci in 1973. The rest, 1.2 Ci, was made up as follows:

| Nuclide | % of Total Activity |
|---------|--------------------|
| $^{89}$Sr | 46% |
| $^{131}$I | 20% |
| $^{137}$Cs | 4% |
| $^{133}$Ba | 4% |
| $^{90}$Sr | 3% |
| $^{58}$Co | 2% |
| $^{134}$Cs | 3% |
| $^{140}$Ba | 3% |
| $^{140}$Ca | 3% |
| $^{144}$Ce | 1% |
| $^{60}$Co | 2% |
| $^{95}$Zn | 1% |

and a number of radionuclides in smaller quantities.

This activity was immediately diluted in Danube water, which already contained a fair inventory of nuclides from fall-out (from weapon testing and so on). A waste volume of $7.62 \times 10^3$ m$^3$ and a discharge of $4.42 \times 10^9$ m$^3$ raised the Danube water radioactivity (without taking $^3$H into account) by 0.3 pCi/litre. The increase due to $^3$H was 30.3 pCi/litre, the concentration of $^3$H above the outfall in the river being already 530 pCi/litre.

The doses of radiation received by aquatic plants, invertebrates and fish as a
result of all the radioactivity of this Danube water from whatever source is perhaps less than twice that predicted by Blaylock and Witherspoon from an average boiling water reactor. The release of $^3$H into the Danube from the Gundremmingen reactor alone could account for a dose of about $6 \times 10^3$ m rads/year to each of the different types of biota, while the $^{89}$Sr might expose fish to $1 \times 10^2$ m rads/year. At such a rate one would be unlikely to detect even the small alterations in snail egg-laying behaviour until the number of similar reactors had been increased to about 1,400, if the radioactivity were evenly distributed.

The Gundremmingen reactor is, however, a very small one, perhaps one-third the size of what may become economic; the even distribution of radionuclides is improbable, particularly in the case of those which are insoluble and concentrate downstream, as would be likely with metal-cooled fast-breeder reactors (Schaeffer, 1975). As far as existing commercial reactors are concerned, it does not seem beyond all probability that minor damage to other species might occur if most of the electricity generated in countries bordering the Danube were one day to be nuclear. For the safety of organisms other than man, the disposal into rivers of low level activity waste and cooling water from commercial reactors must not be ignored.

The convenience and relative safety of discharging cooling water and effluent into large bodies of water such as seas has meant that the majority of existing commercial reactors operate at coastal sites and much work has been done to measure the uptake of radionuclides in aquatic species.

Ettenhuber and Rönsch (1975), who were interested only in the pathways that led to man, tested the edible flesh of seven kinds of fish, comparing their $^{137}$Cs uptake in water in different situations: a Baltic bay, an oligotrophic lake, a river lake, the Elbe and the Oder. The concentration factor for this important product of the fission process varied greatly but was highest in the oligotrophic lake and lowest in the Elbe and the river lake. In Perca fluavivilis, the flesh contained from 6,170 pCi/kg down to 190 pCi/kg, and showed concentration factors up to 25 times greater than in the Danube, thus underlining the difficulty of predicting the result of radionuclide releases from observations made at other sites. With these concentration factors there would have to be a greater constraint upon the building of nuclear power stations than along the Danube if natural organisms were not to be put at some risk.

The physical and biological pathways for radionuclides are as important as their concentrations and much has still to be learned. For example, $^{60}$Co uptake by shrimps is mainly by feeding: there is a rapid turnover with much of the cobalt being lost in the moult (van Weers, 1975), but prolonged and continuous feeding leads to greater retention in the digestive gland.

Few published investigations describe the effects of nuclear power stations alone but Preston, Mitchell and their co-workers (for example in Mitchell, 1973), showed that, at Bradwell, $^{65}$zinc and $^{110}$silver were critical discharges as far as the
oyster beds were concerned. Consumption (by a human adult) of 75g of oyster flesh a day would give about 0.1 per cent of the ICRP recommended dose limit. Preston did not consider that the dose rate to the oysters themselves was likely to be ecologically significant.

**FUEL PROCESSING CONTAMINATION**

Fuel reprocessing contamination has been studied very intensively; for instance at Windscale and from the Bombay harbour area. It is apparent that their outfall contaminants differ markedly; the concentrations of $^{95}$Zr and $^{95}$Nb are important at Windscale but very low at Bombay.

In a bad year at Thania Creek Bridge, Bombay, $^{137}$Cs could have been responsible for something of the order of 0.5 rad/year dose to fish, a concentration 400 times greater than that allowed for by Blaylock and Witherspoon in their model. However, out in the bay, in a good year the fish would receive only 0.01 rad/year from $^{137}$Cs. The radioactive dose to Arca flesh in the region is probably about 30 times that to oysters at Bradwell, and those bivalves near the outfall may be receiving radiation doses only about 300 times less than the smallest dose found to affect the snails of White Oak Lake.

Elsewhere, radioactive waste has been tipped into the ground or small waterways. At Steel Creek, prior to 1968, nuclear production reactors discharged large volumes of heat-exchange cooling water and purge water from disassembly.

| Table 3. Concentrations of radiocaesium in various organisms at Steel Creek. |
|--------------------------------------------------|
| Component | Radiocaesium pCi/g dry weight |
| --- | --- |
| Soil — | |
| Fine Clay | 145.8 — 189.5 |
| Coarse Sand | 13.1 — 23.5 |
| Herbaceous leaves | 35.2 — 2,300.0 |
| Woody leaves |  |
| e.g. *Salix nigra* | 178.8 — 1,600.0 |
| *Alnus serrulata* | 215.6 — 1,000.0 |
| Invertebrates |  |
| e.g. Coleoptera | 440.0 — 990.0 |
| Amphibia and reptiles |  |
| e.g. *Hyla cinera* | 189.2 (wet weight) |
| Birds— |  |
| omnivorous | 25.0 |
| Mammals— |  |
| *Sigmodon hispidus* | 607.9 |
| *Oryzomys palustris* | 214.2 |
basins. Recent measurements by Garten et al. (1975) are shown in Table 3. The authors concluded that the uptake by plants and animals at this site might be typical of other areas in south-east U.S.A. where kaolinite was the most abundant clay mineral. Kaolinite does not fix radio caesium into unavailable forms.

These body burdens are 10 to 300 times those in fish around the Bombay discharge and the dose rates must be similar to those that produced detectable changes at White Oak Lake. This underlines the reasons for discharging waste into the sea, but gives no comfort to those who fear an eventual accumulation of marine contamination.

**AIRBORNE RADIATION**

Any nuclear power station sited at a distance from large bodies of water must rely on cooling towers to replace direct water cooling. Some low-level streams of radioactive waste would need to be discharged independently of evaporation in the cooling tower. Inland nuclear power stations are so rare that there is little knowledge of radioactive contamination from the plume of vapour coming from the tower. The liquid metal cooled fast breeder programme in the U.S.A. may, however, have a plume of wind-borne radionuclides. This fast breeder programme emphasises the hazards of plutonium and its derivatives, to a large extent α-emitters and generally very insoluble.

The dose received from the inhaled transuranics, plutonium, americium, and curium depends upon the particle size and the region of the respiratory tract involved. In man, those regions may be divided into nasopharyngeal, tracheobronchial and pulmonary. ICRP has reported an extensive summary of the deposition, migration and retention of particles introduced into the respiratory tract of man and has constructed descriptive mathematical models of these phenomena (Health Physics, 1966). Doses dependent on particle size and solubility vary widely, and there are no adequate data for estimating the risk of cancer. The BEIR estimates of the risk in the population of the U.S.A. of excess deaths from cancer caused by exposure to ionising radiation are interpreted in this report in terms of risk of death per person per rem of whole body radiation, or inhalation reckoned to give an equivalent unit dose, and they have assumed equivalent biological effects for low and high linear energy transfer processes (LET). The tables provided give, for a 50-year life, the inhalation intake in microCuries of slightly or almost completely insoluble transuranic radionuclides estimated to yield the same risk of cancer as 1 rem total body dose; for instance (about the most favourable case), 9.8 μCi of $^{237}$Pu at 0.05 μm particle size.

This means that for all ‘other cancers’ the excess risk from 9.8 μCi is 2.5 deaths/10⁶ in the United States population or, for instance, a 50 per cent increase in leukaemia in the first 10 years of life or a 2 per cent increase in leukaemia in adults after a two-year latent period and for 25 years thereafter. The least favourable cases are provided by $^{241}$americium and $^{239}$plutonium, which have
these same excess risks from about 1.0 µCi (inhaled). The number of cases involved in the United States population from these stated dose equivalents would amount, for example, to 25/10^6 years for leukaemia in the first 10 years of life.

Because of their physical nature plutonium and most transuranics end as particulate contamination of the soil or sediment in the floor of waterways.

Martin and Bloom (1976) have calculated the inhalation and ingestion of plutonium for a standard man in a contaminated area in terms of ^{239}Pu equivalent.

In order to reduce the maximum dose rate for individual members of the public to the ICRP recommendation of 1.5 rem/year, the average concentration of ^{239}Pu in the soil should not exceed 3 nCi/g. This figure is worth remembering for later comparison of the risk to man with that to some other mammals.

Measurements made at the Nevada test site show that certain strata are well above this level, other strata well below it, but living in a highly contaminated area, if such a thing were permitted, could increase the childhood leukaemia rate about five-fold.

The programme for building liquid metal fast breeder reactors (LMFBRs) in the U.S.A. assumes that there will be 2,200 reactors each with 1,000 MWe electricity capacity by the year 2025. This reactor’s fuel cycle includes ^{238}Pu, ^{239}Pu, ^{240}Pm, ^{241}Am, ^{242}Cm and ^{244}Cm. The radiation likely to come from the airborne plume of its cooling towers has been calculated by Cuddihy et al. (1976), who examined the possible tumour-producing and genetic effect in man. They assumed that material absorbed into the blood stream from the atmospheric dispersion was deposited in the liver (45 per cent), in bone (45 per cent), in kidney (1 per cent), in testes (0.05 per cent) and in ovaries (0.01 per cent), and that man had a life-span of 70 years. The projected organ burdens were more or less stable by the year 2075 and the estimated risk factors of cancer of lung, bone and liver were —

| Organ  | Incidence per man-rem |
|--------|-----------------------|
| Lung   | 16 - 110 x 10^{-6}    |
| Bone   | 2 - 17 x 10^{-6}      |
| Liver  | 1 - 1 x 10^{-6}       |

Genetic damage was calculated as —

(a) specific defects $50 - 500 \times 10^{-6}$
(b) of complex aetiology $10 - 1,000 \times 10^{-6}$

This gives about 20 tumours in 200 years in the whole United States population, and sensitivity tests on the model show that about the least favourable factors have been used in the calculations. One should note that the calculations apply only to the fall-out from the LMFBR programme and in no way to the liquid wastes, which would be a considerably greater though more localised environmental burden.
COMPARISON OF RISKS

To put these figures into perspective it may be helpful to make some comparison derived from Mayneord and Clarke (1975). The cancer risk of Hiroshima survivors has been estimated as one to two cases per rad of whole-body irradiation per $10^6$ population per year over the first 13 years.

Pochin (1972) concluded that among irradiated spondylitics 100 rads to the whole body would (by 20 years) cause other malignancies in $0.5$ to $3$ per cent with possible final figures of $1$ to $7$ per cent.

The calculated risk of cancer in man in one generation from the fall-out of the projected LMFBR programme is of the order of 1:2,500 of that which would follow medical radiotherapy of the whole population.

It would be unreasonable to expect man to be the least sensitive of mammals, so one may be justified in assuming that the response in wild and domestic animals may vary around man as a mean with certain strains suffering up to 20 times as many cancer casualties.

At the Nevada test site the intake and radiation dose to beef cattle feeding on typically contaminated vegetation was estimated. To extend these very complex calculations one may estimate the 2 years' body burden and thus the whole body dose to a single steer as $1.06 \times 10^{-2}$ rads. From this it might be reasonable to expect a one in a thousand chance of a single case of bone cancer in a 1,000 head herd, or one example of genetic damage in 2 years if feeding took place on soil only contaminated to a level acceptable to man, namely $3 \text{nCi/g}^{239}\text{Pu}$.

Similar figures, showing risks much higher than those for man, could be expected to refer to other herbivores such as deer, rabbits, hares and fieldmice, though carnivores would have a risk more comparable to man, unless they lived on the more contaminated portions of the herbivorous population. Moles feeding largely on worms, though they reject the soil content of the worms' gut (Mellanby, 1971), would be faced with risks perhaps 10 times as great.

Compared with the high concentration of $3 \text{nCi/g}$ in soil contaminated by waste to a maximum acceptable level, Cuddihy's model suggests $0.36 \text{mCi}$ to the atmosphere per 1,000 MW/year, with a dispersal of something like $70 \times 2,000$ miles. From this the genetic risk to 1,000 head of beef cattle in 2 years would only be of the order of $4 \times 10^{-10}$, an altogether negligible hazard.

The possibility that a large inland nuclear power station, far from any large body of water, might be erected near my home in Huntingdon has compelled my attention. It interested me very much to think about the disposal of the radioactive waste from that projected nuclear reactor.

Much of the high activity waste, whatever type of nuclear reactor were built, would be removed for fuel reprocessing. At Windscale such waste is either stored or disposed of by controlled sea drainage. There seems to be no immediate prospect of significant ecological disturbance off the Cumberland coast near Windscale nor in the Irish Sea unless the number of nuclear power stations were very greatly increased.
Concerning the plume of vapour from cooling towers, I have used Cuddihy’s calculation, which refers to a fast breeder reactor and may well not be more than the roughest guide. However, even within the 70 mile wide band to the coast in the direction of the prevailing wind where fall-out is maximal, somatic and genetic changes in the most susceptible animals would probably be undetectable.

Disposal of the low activity stream of liquid effluent would have presented much greater, possibly insuperable, problems. The immediate region is drained by small streams emptying into the Great Ouse and the Nene, both slow-flowing rather shallow rivers. A considerable proportion of their water is extracted well downstream of the immediate region of Molesworth on the Huntingdonshire/Northants border and is pumped back for further use. The possibility of progressive concentration of some of the long-lived radionuclides within organisms (including man) in a wide segment of England between London and the Norfolk coast would have had to be faced, and it would probably have been found wiser to pump this effluent 45 miles to the sea and then mount an investigation of uptake in such likely indicators as mussels and oysters, and the creatures, like oyster-catchers, that feed upon them.

The building of a fast breeder reactor near to active military airfields never seemed very likely but, because plutonium and its derivatives seem to present especial problems, I have also examined some possible results of dilute effluent disposal from such a facility.

Wahlgren et al. (1976) described the situation in Lake Michigan; plutonium and its derivatives from fall-out of weapon testing are found in the sediment and the water. Amounts are very low, nevertheless there is considerable concentration by phytoplankton, and certain littoral organisms such as Cladophora, Equisetum and Chara show similar concentration ratios. In mussels, the half-time is probably long, about two years, while certain benthic invertebrates concentrate up to one-half of these figures. The authors conclude that while fall-out has so far presented no great problem (it has been reported recently (Pillai and Mathew, 1976) that, world-wide, nearly 325 kiloCuries of $^{239, 240}$Pu and about 8 kiloCuries of $^{238}$Pu have been deposited from weapon testing), discharge of waste into enclosed bodies of water would have to be regarded seriously. There are similar findings from the Baltic.

In comparison, since 98.5 to 99 per cent of the total plutonium at a fuel reprocessing works is recovered, the total discharge of plutonium from Windscale in the last 15 years is approximately $10^4$ Ci and only about 0.3 per cent of the plutonium production in the U.K. is released into the sea. Even so, in the immediate vicinity of the discharge the Pu concentration is between 500 and 1,000 times greater than that due to fall-out alone.

In the Bombay harbour area the maximum concentration of $^{239}$Pu and $^{90}$Sr was in Arca flesh (10 times higher than in other species) and this was 30 to 150 times higher than that due solely to fall-out. The behaviour of plutonium, like other radionuclides, cannot be satisfactorily projected from observations at different
sites. In the Buzzards Bay area on the Atlantic Coast of America the sediment (containing some 2.0 mCi/km² Pu and 7 mCi/km² of $^{137}$Cs) is penetrated by large fast-moving worms that not only disturb the sedimentary profile (Bowen et al., 1976) but presumably obtain a radiation exposure which might be sufficient to affect a more radio-sensitive creature. Plutonium seems far less prone to be highly concentrated in higher forms but the reverse applies to $^{137}$Cs (in trout for instance).

As far as individual nuclear power stations are concerned there is little to be feared for animals other than man, either in somatic effect, effect on the developing ovum, or on the genetic pool, unless the programme greatly multiplies discharge into a single river, or the water disposal is not permanently diluted at once. The results of other activities concerned with nuclear fuel are not so clearly innocuous, and radiation sufficient to account for some detectable changes does accompany uranium mining and milling. The waste from fuel reprocessing gives rise to many problems of disposal. There are significant dangers to animals after disposal on land and in confined bodies of water. The sea offers, at least for the time being, a safer place for disposal, but safety cannot be taken for granted at any site. To multiply sea disposal points would be to increase the proportion of aquatic life at risk. To multiply the concentration of disposal at existing points would raise the risk to locally sedentary populations. The choice needs informed consideration. Much more work is required before we can be sure that the nuclear strategy is wisely chosen. Recently we have heard of the devastation of Scandinavian trout streams by the sulphur dioxide of fossil-fuel power stations all over Europe. Compared with this, the apparent effects of radionuclides are miniscule; but some of the radioactive relics of our nuclear power programme will still be with us when fossil-fuel power stations are archaeological curiosities.

This article is based on the Ernestine Henry Lecture given in the Royal College of Physicians in October 1976.

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