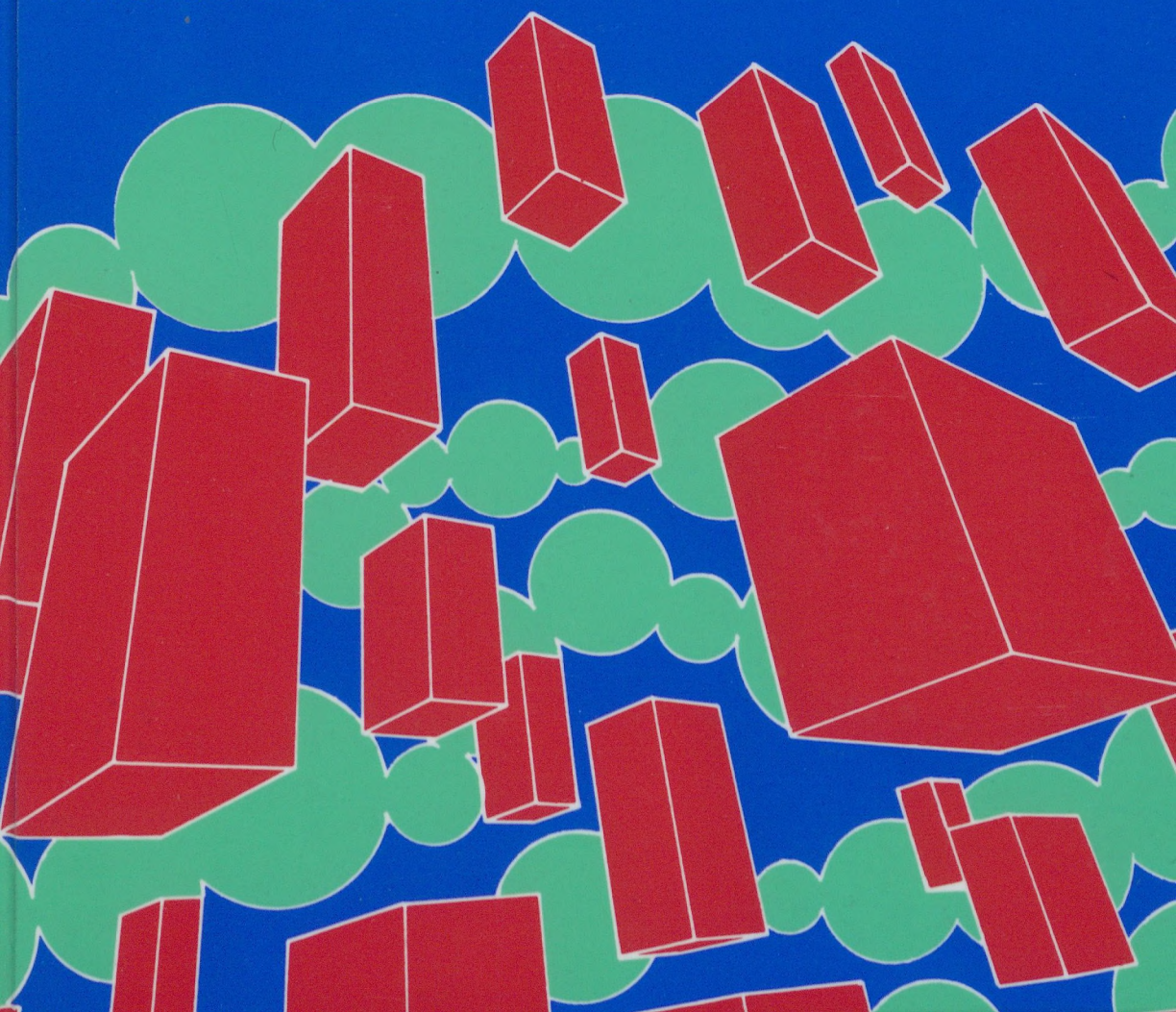


RADIOECOLOGY AND ENVIRONMENTAL PROTECTION

András S. Szabó



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ANDRÁS S. SZABÓ

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Translated by Tamás Kriesch and János Földényi

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PREFACE

Population growth, urbanization, rapid industrial and technical development, and the spreading of intensive agricultural methods have caused a number of undesirable effects in our environment. At times we can witness even alarming phenomena. It is sufficient to take a glance at the Danube, which, like so many other rivers, used to be clear not very long ago, but by today almost all of them have become enormous sewage ducts; or just compare the relatively smoky, sooty, petrol-vaporous air of industrial towns with relatively fresh woodland air. It is a regrettable fact that as a consequence of the various sources of contamination in many areas our environment has become polluted to such an extent that there is no chance of its natural self-cleaning. Therefore active intervention, development of technologies protecting the environment, and decrease of the contamination levels will be necessary. It also remains to be learnt what concentrations of the contaminating components, and among them the several micro-elements and radioactive isotopes, are present in the different segments of the air-water-soil-plant-animal-man biological chain, and what are the factors which influence their concentration or accumulation.

The problem of radioactive contamination of the biosphere and protection against such effects is embedded in the problem of contamination of our environment. Due to the existence of contaminating sources (nuclear tests, radioactive emission of nuclear power plants, etc.) the regular checking of the radioactive contamination of our environment is a current and important task today, and it will also remain as such in the future. In Hungary this issue is especially timely in the Paks area, where highly extensive and very careful radiometric examinations have been conducted for several years to measure background levels, and to detect the possible environmental contamination effect of the nuclear power plant consisting of 440 MW_e blocks, which has been operating for a few years. The World Health Organization (WHO)—realizing the environment-contaminating effect of radioactive isotopes has passed a resolution that data will be regularly collected and published on the extent of radioactive contamination. It is especially important to know what the changes in concentration are, of the radioactive isotopes of the different elements present in the consecutive members of the biological chain process, what the factors influencing the migration, increase or decrease of their concentration are, and how the contamination level is affected by the various parameters determining agricultural production (such as fertilization, irrigation, land use, rainfall and so on) and by the different technological

steps of food production. Among others, these are questions for which answers are sought—and, I hope, found—in this book.

Investigations with the aim of measuring the radioactive contamination and determining the external environmental contamination level do not have a purpose in themselves. With food, drink, and partly by inhalation, radioactive isotopes enter our bodies, where they can accumulate and, depending on their concentration, physical and biological half-lives, may cause deleterious changes (for example, gene-mutations or chromosomal aberrations). And since there is believed to be no threshold minimum dose for a genetic effect, i.e. the smallest dose of radiation can give rise to a mutation of genes, for our present and future it is a decisively important task to reduce the radiation level affecting mankind to the lowest possible level. However, as far as we know, the somatic radiation effect has no threshold dose either (ICRP 1977).

In this book quantitative radioecology is primarily treated as applied to global radioactive pollution. The ecology of natural radioactive substances, the detailed, quantitative radioecology of the environmental effect of nuclear establishments (concentration factor and system analysis methods) are not, or only incidentally are, dealt with since a comprehensive discussion of the subject would be far beyond the scope of this book. However, in the different chapters reference is made to the importance of radioactive isotopes of natural origin, and some data are given on the radioactive emission of nuclear facilities, too.

“There is only one Earth” was the logo of the international conference on environmental protection held in Stockholm in 1972. We live on this one Earth, and it is here where we and following generations shall bring about those environmental conditions which will make possible a sensible human life. I hope that having a clean and healthy environment is not a utopian dream, and that *Homo faber*, who has transformed and curbed the forces of nature, who has begun to understand the interactions and laws of the processes taking place in nature, will, in the possession of this knowledge, really be able to act also as *Homo sapiens*.

In Dubna, Russia, a small town lying about 120 km north of Moscow, where the Joint Institute for Nuclear Research operates, there is a street. Written in capitals on the wall of one of the dwelling-houses in this street the following inscription can be read: “Atom ne soldat, atom rabochiy” (Atom is not a soldier, atom is a worker). That is, nuclear power should not be used for military purposes, for the manufacture of weapons of mass extermination, for the destruction of mankind and the treasures of human culture, but, instead, for peaceful creative work, for the attainment of real humanitarian aims.

However, to achieve all the above, figuratively speaking to bring about an earthly paradise on our planet instead of hell, there is still a lot to be done, and the responsibility of ecologists, and among them, radioecologists, cannot be over-emphasized. The task of ecologists—as it is they who know the changes in the different elements of the biological chain process—is to call the attention of people to any danger and to form the right attitude. It is hoped that this book will also provide some help in this respect.

The book comprises four chapters. In the first, radioecology as a special field of ecology is presented and the build-up of radioactivity of the biosphere, both natural

and artificial in origin is analysed. The author believes that it is essential, or at least advisable to read this chapter in order to understand the following parts.

The bulk of the book is contained in Chapter 2, which deals with the radioactivity of the different elements of the biosphere, and the analysis of the different factors (climatic parameters, configurations of the terrain and amelioration) affecting the radioactive level. Here, the different components of the biosphere (atmosphere, hydrosphere, etc.) are discussed separately, naturally pointing out the permanent and dynamic interrelationships among the individual elements, the interdependence of the air–water–soil–plant–animal–man biological chain, and the necessity of analysing the ecological processes in such a complete system. In spite of the fact that the interrelations among the different members of the chain process are evident, it was deemed necessary to deal with the individual elements of this chain one by one. This is because, obviously, a hydrologist would primarily be interested in the radioactivity of the hydrosphere and the ecological factors influencing it. For him, for instance, the possibilities of decreasing the radiation load on humans may be less interesting. Or to give another example: knowledge of the radioactive contamination of the soil is of secondary importance to an expert investigating the metabolism of animals. An effort has been made to write the six subchapters in such a way that they be understandable in themselves, that is, it is not absolutely necessary for the understanding of one of the subchapters to have a knowledge of the rest. Naturally—since the final component of the above-mentioned biological chain is Man—the most important question ultimately raised is: How do the different ecological factors, with the interposition of the different components of the biological chain, affect man? Therefore the sixth subchapter, which deals with the problems of radiation load affecting the human organism, is of primary importance. Naturally in Chapter 2 the author also treats in detail the problems of the Chernobyl nuclear power-plant disaster which caused radioactive contamination. The author believes that the Chernobyl disaster—in the negative sense of the word—will be considered a milestone in the history of nuclear technology, one which will serve as an eternal memento for later generations as well.

In the field of ecology—and within that also in radioecology—there are still a lot of blank areas, there are a number of questions which are waiting not only to be answered, but also to be raised. Thus, similarly to the other fields of science, coordination of research is also indispensable in radioecology. In the third part of the book I give a very short coverage of the research programmes in Hungary that are of radioecological nature, or closely connected with the subject. A list of the institutes participating in radioecological research is supplied, and a brief analysis is given also on the activity of the radiometrical monitoring network of the Ministry of Agriculture and Food in Hungary.

The fourth part of the book calls attention to the dangers which threaten mankind if nuclear power—the second lighting of fire—is not used for the sake of mankind, but the contrary. Finally answers are sought to two exciting questions, whether nuclear power plants are dangerous to the environment, and whether nuclear technology is a blessing or a curse for mankind.

At the end of the book a bibliography is provided, including a separate list of Russian publications. The bibliography contains cited as well as recommended

literature and has some items which are not referred to in the different chapters of the book.

It is to be emphasized that the discipline of radioecology is making rapid advances as reflected by the fact that the number of publications on radioecology published from the National Laboratory, Oak Ridge, USA, and from the Laboratories of Environmental Research of the Atomic Energy Committee of France by 1980 have increased by fifty times since the 1960s (Kirchmann 1983). More than 20% of the budget of the US Atomic Energy Committee is allocated for biological and medical research, and approximately the same percentage of the European Economic Community budget for radiation protection has been spent on radioecological research work.

Since the book was in the first place written for those interested in topics of an ecological nature, and not for experts engaged in radioanalysis, questions related to analysis and measurement techniques are not treated at all or are dealt with only incidentally in the main part of the book. Evidently the explanation of concepts and measuring methods is necessary, therefore a brief Glossary of all radioecological terms used in the book is provided. There is also an Appendix where information of prime importance is provided, a useful guide to the theoretical basis of the phenomenon of radioactivity, and also to the analysis and measurement of radioactive isotopes.

Finally, a few words about those to whom this book is addressed. First of all, it is obviously meant for experts on radioecology, those who investigate the questions of radioactive contamination, and those who study the migration and possible accumulation of radioactive isotopes in the different elements of the biosphere. As a matter of course it is also addressed to those who are active in the fields of radiobiology, radiation technology, dosimetry, and radiation protection. I think, however, that this circle is necessarily wide, as anybody who works in any field of ecology, environmental protection, physiology, or nuclear engineering, let alone in agricultural production or the food industry, may glean useful, sometimes perhaps even surprising and interesting information by reading the book. It has to be pointed out that since the book was published in Hungarian tremendous changes have taken place in this branch of science which the author has updated. However, the drastic political changes were entirely unanticipated. As the publication of this manuscript was under way, we have not attempted to implement any political changes in names and countries.

András S. Szabó

1. THE SUBJECT OF RADIOECOLOGY AND THE DEVELOPMENT OF RADIOACTIVITY IN THE BIOSPHERE

In order to first introduce the field of radioecology, it seems necessary to delimit this branch of science and to mark its place in the comprehensive system of ecology. It is all the more necessary to do so, since radioecology is a relatively new term—it could perhaps be described as radio-environmental science, or radio-environmental biology.

Radioecology—as will be analysed later in more detail—deals with the ecological questions in connection with the radioactive isotopes found in the biosphere. As to their origin, these radioactive substances can be divided into two groups: radionuclides of natural or of artificial origin. It is necessary to familiarize ourselves with those natural and artificial radioactive isotopes which, due to their concentration, half-life or biological effects, are the most important—that is, with the natural radioactivity produced by geological processes and cosmic radiation, and with the radioactive contamination consequences of the nuclear weapon tests and emission of nuclear power plants, nuclear facilities and isotope laboratories.

The presence of radioactive isotopes may cause somatic and genetic radiation damage. Therefore our aim is to reduce the amount of radioactive isotopes of artificial origin being released into the environment, and to achieve as low a level of radioactive contamination of the biological chain as possible. Comprehensive environmental protection will also include protection from radioactive contamination occurring in all components of the biosphere and causing damage to the living organisms. In addition to protection of the natural constituents of water, air, soil, protection from noise and the disposal of waste materials, it will also include protection from radioactive contamination.

In the first chapter radioecology is introduced as a special field of the science of ecology, the more important natural and artificial radioactive isotopes are discussed, and finally the importance of protection from radioactive pollution as an integral part of environmental protection is pointed out.

1.1 RADIOECOLOGY AS A SPECIAL FIELD OF ECOLOGY

The subject of ecology

Since there are numerous references—Ellenberg 1973; Osche 1973; Stugren 1974; Dreux 1974; Larcher 1976; Tischler 1976; Lerch 1980; Remmert 1980; Kultiasov 1982—which deal with ecology in general, this book is confined to a very special field of ecology, radioecology, that is, the ecological discipline concerned with radioactive materials. Ecology itself—for readers other than ecologists—is therefore dealt with only to the extent absolutely necessary to enable one to place radioecology, as a special field of ecological science, into the comprehensive system of ecology.

The term ecology is derived from the Greek word *oikos*, which means “farmstead, property, home, house”. The literal meaning of the word has been expanded, so that ecology is now taken to be the science studying the interactions among living beings, biocoenoses, and the environment. As Kovács (1975) formulated, “The interrelations of living organisms, their interactions with the environment, quantitative and qualitative changes of living matter, biological production, connections of Man with the living and non-living environment constitute the actual subject of ecology, the study of the environment, as an independent branch of science.”

Ecology, or, by its other name, environmental biology, is therefore that branch of biology which analyses the intricate interactions of living organisms and their environment, and the regularities of the changes of interactions. In our days this scope of knowledge—precisely as a consequence of the ecological crisis—has come to the focus of attention and the science of ecology, because of its complex nature, strives to establish general rules by compiling knowledge from the different branches of the discipline (soil ecology, water ecology, plant and animal ecology, human ecology). Summing up, ecology is that branch of biological science which deals with the external (geographical) conditions of life, its effects on the living organisms, and the reactions, interactions, accommodations and acclimatization of the living organisms. Although ecology is basically a biological science, due to its varied and complex nature it is characteristically an interdisciplinary field. Therefore during research on ecology and environmental protection, the close cooperation of specialists of different branches of science—biologists, agriculturists, physicists, technical experts—is required.

Interrelationships

Living organisms form a dynamic unity with their environment. On the one hand, the different components of the environment have an influence on each other (e.g. water on soil) and these factors jointly on living organisms, and on the other hand, living beings also have an influence on their environment (e.g. plants on soil).

Living organisms respond with a special ecological accommodation to the changes of the ecological factors. The ability to accommodate to changes in the conditions of

life is a basic characteristic of living organisms, and it is one of the constituents of self- and race preservation. When there is a change in the conditions of the environment, only those individuals of a species can survive and reproduce which are able to adapt themselves to the new conditions. In the course of adaptation it is obvious that not only accommodation to the non-living environment, but also accommodation to each other will be necessary.

Abiotic and biotic factors

Ecological factors can be divided into two groups: thus we can speak about non-living (abiotic), and living (biotic) factors. Classified as abiotic factors—which characterize the physical, and chemical parameters of the environment—climatic, edaphic, and orographic factors should be mentioned. Temperature, light, air, and water are classed among the climatic factors. The edaphic factors comprise the characteristic physical or chemical features of the soil, while the orographic factors, also called the structure of the terrain, describe the characteristics of relief (altitude, vertical configuration, and inclination).

By biotic factors we mean the effects living organisms exert on other organisms. Within these, distinctions are made between plant-plant, animal-animal, and plant-animal effects and interactions, and the influence of human activity (anthropogenic factors) on the environment and living organisms. The effect of the anthropogenic factors on living organisms can be direct (e.g. forest clearing) or indirect (e.g. fertilization or applications of pesticides), when some of the physical or chemical characteristics of the environment are altered in such a way that it has an effect also on living organisms.

In addition to the effects of the biotic or abiotic factors in the ecological phenomena, processes or interactions, the time factor should also be taken into consideration, since the ecological factors have an influence not only in space but also in time. It is important to take into account the time factor, because as a function of time the organisms undergo transformation and develop in a way different from that before, but in accordance with the new, changed circumstances.

Ecosystems and metabolism

The biosphere consists of a continuous series of different ecosystems (e.g. deciduous forest, tundra), which are also interrelated with one another. The ecosystem or bio-geocoenosis is the unity, the close and mutual connection of the living area (biotope), and the community of organisms living there (biocoenosis). A characteristic feature of ecosystems is the fact that there is a continuous metabolism between the living and inanimate components. Ecosystems are basically the units of material and energy transport of the biosphere, systems in which the input and output of materials is energetically determined.

The continuous interaction between the living organisms and the environment is

mediated by metabolism (exchange of materials). Metabolism is the basis of all symptoms of life. Nutriment is taken up from the environment, and transformed to materials characteristic of the body (assimilation, anabolism). The materials of the body are used (dissimilation, catabolism), and finally the detrimental or unused materials, and metabolic end-products are removed. The questions of material and energy transport of ecosystems are investigated by production biology. Production biology (the study of the connection between living organisms and the environment) considers environmental conditions necessary for the production of organic matter and explores the possibilities of increasing the production of organic matter.

Autotrophs and heterotrophs

The material and energy intake process of metabolism is nutrition which is an essential condition for the survival and propagation of living organisms. From the point of view of their nutrition, living organisms can be divided into two groups which are basically different from each other and which are called autotrophs and heterotrophs. The green plants and some bacteria are capable of producing organic materials from water, carbon dioxide, nitrogen, and inorganic mineral matter using the energy of light (photosynthesis) or chemical energy (chemosynthesis). These are the self-supplier, or autotrophic organisms. The animals and the saprophytic plants (e.g. fungi) are not capable of producing organic matter from inorganic substances. These are called heterotrophic organisms which are not self-suppliers and are consumers of the organic matter produced by the autotrophs.

The food chain—which indicates the dynamic structures of the ecosystems—can essentially be characterized by the following major partial processes:

- green plants by means of photosynthesis produce organic matter from inorganic substance;
- phytophagous animals (including Man) transform organic plant matter into substances of their own body;
- carnivorous animals (including Man) transform the substances of the body of phytophagous animals into substances of their own body;
- inanimate organic matters are destroyed by the bacteria in the course of decomposition and decay.

Thus bacteria return the substance to the original stage, and after mineralization organic matter is synthesized from these substances again by the plants. The circle is thus completed and the process starts all over again.

On the basis of what was mentioned previously, it is clear that the transport of matter is not only characteristic of living organisms, but also takes place in the environment, and between living organisms and their environment. This migration is of a cyclic nature, the processes of which are analysed by biogeochemistry. Obviously the cycles are not always regular, they are very often incomplete. In some cases, points of stagnation can be observed when organic matter accumulates and temporarily does

not take part in the chain process. Of the biogeochemical cycles those of water, carbon, nitrogen, phosphorus, and sulphur have been studied in the greatest detail.

After this short survey let us see how radioecology is linked to the science of ecology, and what the subject of radioecology actually is.

Radioecology

In my interpretation radioecology, or radio-environmental biology is a special field of ecology, and it deals with the interactions between the environment containing radioactive materials and living organisms. That is, it investigates how radioactive materials affect different living organisms, what are the factors which influence the concentration, accumulation and distribution of these radioactive isotopes in the different elements of the air–water–soil–plant–animal–man biological chain and what are the methods by which the level of radioactive contamination can be decreased. So, in contrast with the horizontal division of the different ecological sub-fields, radioecology comprises vertically the whole spectrum of ecology, that is, it covers all ecosystems and each element of the biosphere, only its viewpoint is a special one, since it approaches the question from the aspect of the radioactive materials. Consequently we may say that the subject of radioecology is the migration of radioactive substances in the different elements of the biosphere, the effect of radioactive isotopes on living organisms, and the interaction between the radioactive environment and living organisms.

Radioecology, similarly to the comprehensive field of ecology itself, is an interdisciplinary branch of science, its progress and methods of research cannot be separated from the fields of several other sciences. It is closely related to radiobiology, radiology, dosimetry, nuclear chemistry and nuclear physics, nuclear technology, isotope-technique, nuclear measurement-technique, physiology, ethology, genetics, biochemistry, nutrition physiology, toxicology, health physics and naturally to the more or less readily definable subfields of ecology and environmental protection (e.g. soil science–soil ecology–soil pollution–soil protection, hydrology–water ecology–water pollution–sewage purification–protection of the quality of water) and several other branches of science. Radioecology is a typical borderline interdisciplinary subject, on the one hand it is characterized by strict specialization, that is differentiation, within ecology, and on the other it builds on, links up and integrates the different fields of the other branches of science.

Naturally it is not possible within the scope of this book to give an answer in detail to all questions of radioecology, nor to explore the system of relations between radioecology and the other branches of science. The main subject, that is to say, the principal feature of this book is the radioactivity of the different elements of the biosphere, and the introduction of factors which influence radioactivity. Primarily, questions related to the level of radioactive contamination as well as to the changes in the concentration of the radionuclides in the different elements of biological chain process under the effect of ecological factors and agricultural and food industrial activities are treated.

It was not my aim in writing this book to give a comprehensive and detailed review of the radioactive contamination levels which have been and which could be measured in the different elements of the biosphere. There is an abundant literature on the subject, and the available body of knowledge is unbelievably profuse. To sum up the material of all papers written on the subject, or simply to refer to the publications which are considered to be the most important ones, would exceed the scope of this book. Instead, it was my intention to call attention to those general rules, biological connections, ecological parameters and interrelations which can cause changes in the levels of radioactivity, and to the possibilities of modifying radioactive contamination. As a matter of fact voluminous data are provided in the book on the levels of radioactivity measured in the different elements of the biosphere. However, these by no means constitute the main part of the book but rather serve to substantiate the relationships, at times provide additional information and assist in drawing certain conclusions. The book is mainly concerned with the factors which can affect radioactivity levels and to what extent such changes occur.

1.2 RADIOACTIVE ISOTOPES OF NATURAL ORIGIN IN THE BIOSPHERE

In nature—independently of the pollution caused by nuclear weapon tests, nuclear power plants, and isotope laboratories using artificially produced radioactive materials—there are a number of radioactive isotopes. These radioactive substances of natural origin emit α -, β -, and γ -rays.

Table 1 Radioactive isotopes of natural origin with small and medium mass

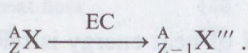
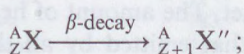
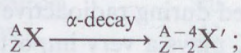
Isotope	Percentage of isotope in the element	Half-life (year)	Emitted radiation
^3H	variable	12.3	β^-
^7Be	variable	0.15	γ
^{14}C	variable	$5.57 \cdot 10^3$	β^-
^{22}Na	variable	2.6	β^+ , γ
^{24}Na	variable	0.0017	β^- , γ
^{40}K	0.0119	$1.3 \cdot 10^9$	β^- , γ
^{50}V	0.25	$4 \cdot 10^{14}$	γ
^{87}Rb	27.85	$6 \cdot 10^{10}$	β^-
^{115}In	95.77	$6 \cdot 10^{14}$	β^-
^{138}La	0.089	$2 \cdot 10^{11}$	β^- , γ
^{144}Nd	23.87	$1.5 \cdot 10^{15}$	α
^{150}Nd	5.60	$1.0 \cdot 10^{15}$	β^-
^{147}Sm	15.07	$1.2 \cdot 10^{11}$	α
^{176}Lu	2.6	$2.1 \cdot 10^{10}$	β^-
^{187}Re	62.93	$5 \cdot 10^{10}$	β^-

In the case of α -decay a transformation of the nucleus takes place, the atomic number decreases by two and the mass number by four, and since the α -particle is a He-nucleus (helium) with two positive charges, two protons and two neutrons leave the nucleus.

In the case of β -decay the mass number remains unchanged, the atomic number is, however, increased by one. The process of β -decay can be considered as follows: in the nucleus one neutron is transformed to a proton and an electron, the proton stays in the nucleus (increase of atomic number) and the electron is emitted.

Generally γ -radiation is concomitant with α - and β -radiation; its energy is discrete. As γ -radiation is an electromagnetic wave originating from the excited nucleus, neither the atomic number, nor the mass number will change. However, γ -radiation is very often the concomitant symptom of electron capture. One proton of the nucleus captures a shell electron and uniting with it forms a neutron in addition to the characteristic X-ray radiation.

This is the case in an electron capture if the product nucleus is formed in an excited, unstable state. The transformation of an X isotope of an A mass number and Z atomic number by α - and β -decay and electron capture (marked by EC) is as follows:

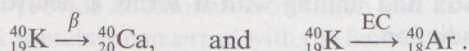


The α -particles are rays of a strong ionizing effect, but have a small range of penetration (in the air they get absorbed after a few centimetres), so they are dangerous for humans only when incorporated, that is when they enter the body. The ionizing capacity of the β -radiation is smaller and its range of penetration is longer (in the air a few metres). The radiation with the greatest range of penetration is γ -radiation, which at the same time has the least ionizing capacity.

Living organisms are affected by the external radiation sources of natural origin through terrestrial and cosmic radiation, as well as through the radiation of the radioactive isotopes produced by the latter. Rocks containing radioactive isotopes are the sources of terrestrial radiation, while cosmic radiation is a high energy radiation arriving at the Earth from space which consists of 77% protons, 21% α -particles and 2% heavier nuclei. The particles constituting cosmic radiation—the so-called primary radiation—collide already in the top layers of the atmosphere with the atoms which are contained in the air and through different kinds of nuclear reactions several kinds of radioactive isotopes (e.g. ${}^3\text{H}$, ${}^7\text{Be}$, ${}^{14}\text{C}$) and elementary particles (e.g. mesons, hyperons, neutrons) are produced. The half-life of the radioactive isotopes, generated by primary cosmic radiation is relatively short, thus an equilibrium is formed between their production and decay. The more important radioactive isotopes of natural origin with small and medium mass are shown in Table 1. Actually the direct external dose of cosmic radiation on the living organism is one order of magnitude greater than the internal dose of isotopes produced by it.

Of the elements given in Table 1, the radioactive effect of potassium has been known for the longest time, since the abundance of the ^{40}K -isotope is considerably higher than the concentration of the other radioactive isotopes of natural origin. Since potassium is a very common element in nature—it is the seventh most frequent element on Earth—the potassium-concentration of the soil is generally around 1–2%. The potassium-content of granite rocks is 3–4%, while its concentration in its minerals (orthoclase, microcline, muscovite, plagioclase, potash mica) reaches as much as 10%. The highest potassium-concentration of all is that of sylvite (KCl) occurring in salt sediments.

During radioactive decay, the ^{40}K -isotope emitting β -radiation transforms into calcium at a probability of 89%, and by means of electron capture taking place at a probability of 11% transforms into argon.



The maximum energy of the emitted β -radiation is 1.33 MeV, the characteristic energy of the γ -radiation is 1.46 MeV.

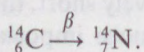
It is known that heat is released during radioactive decay, and the amount of heat evolved during the decay of ^{40}K plays a very important role in the development of the thermal balance of our planet. The amount of heat released by potassium decay is approximately the same as that emitted by all the natural radioactive families together.

During one year about 10^{-5} J of heat energy is released as a consequence of the radioactive decay of 1.0 g of elemental potassium.

Potassium is an essential macro-element for plant, animal, and human organisms. From the point of view of radiation load, the ^{40}K isotope is of crucial importance, because it is an isotope with long half-life and its concentration is high. For example, it often represents 20–30% of the mineral material content of plants, and the potassium content of the human organism is generally between 1.5–2.2 g/kg body mass for men, and 1.3–1.7 g/kg body mass for women (Lloyd et al. 1973; Bogolyubov 1975). The potassium content of some foodstuffs of vegetable and animal origin is shown in Table 2.

Next to ^{40}K , ^{14}C is the most important radioactive isotope of natural origin. ^{14}C is a pure β -emitting nuclide and, in spite of having very small energy ($E_{\text{max}} = 0.156$ MeV), it plays an important role among the radioactive isotopes because in the assimilation process its behaviour is similar to that of natural carbon, so it is incorporated into the different living organisms. As a consequence of its global circulation, carbon is permanently exchanged between the atmosphere, hydrosphere, lithosphere and biosphere. The carbon cycle is shown in Figure 1.

From isotope ^{14}C ^{14}N is formed by emission of β^- -radiation,



As a matter of fact, this reaction can also be used to determine the age of finds dating back as far as the Neolithic and Upper Pleistocene periods. The basis of

radiocarbon age-determination is that the ^{14}C isotope in the form of CO_2 is taken up by the plants from the atmosphere and the concentration of ^{14}C was approximately constant (until the beginning of the nuclear weapon tests), since on the one hand ^{14}C has decayed with its half-life of 5570 years, but on the other hand it has also been generated by cosmic radiation throughout this time thus forming equilibrium between decay and production. However, carbon and ^{14}C present in dead organisms will not be replaced, consequently the concentration of ^{14}C , that is the ratio

Table 2 Average K content of some foodstuffs of vegetable and animal origin

Foodstuff	K concentration mg/100 g original material
Spinach	526
Green peas	623
Green pepper	165
Mushroom	580
Apple	112
Cherry	174
Peach	183
Walnut	1710
Wheat flour	140
Beef	381
Chicken	400
Pork	442
Pork liver	368
Bologna sausage	250
Cow's milk	160

between ^{14}C and ^{12}C gradually decreases. Knowing the specific activity of ^{14}C the age of the test samples containing carbon can be determined. This method can be applied for a period of about 50 000 years, since in the case of samples dating back to earlier periods the specific activity is very low, and thus the measurement or calculating error can be too great.

Radioactive isotopes are widely used as indicators to solve various scientific tasks and problems. The basis of this method was explored by a Nobel Prize laureate of Hungarian origin, George Hevesy. The essence of the method is that since the chemical characteristics of the inactive and active isotopes of the elements are identical, or at least very similar to each other, so for example, the "labelled" (radioactive) isotopes can readily be used for indicating the biochemical processes taking place in different living organisms. In physiological tests, e.g. for the investigation of the phenomenon of photosynthesis, ^{14}C labelling is used very often, for there is no essential difference between natural inactive carbon (^{12}C) and ^{14}C isotope from a chemical point of view. Today nearly 2,000 radioactive isotopes are already known—the majority of these were artificially produced in high-flux nuclear reactors, or

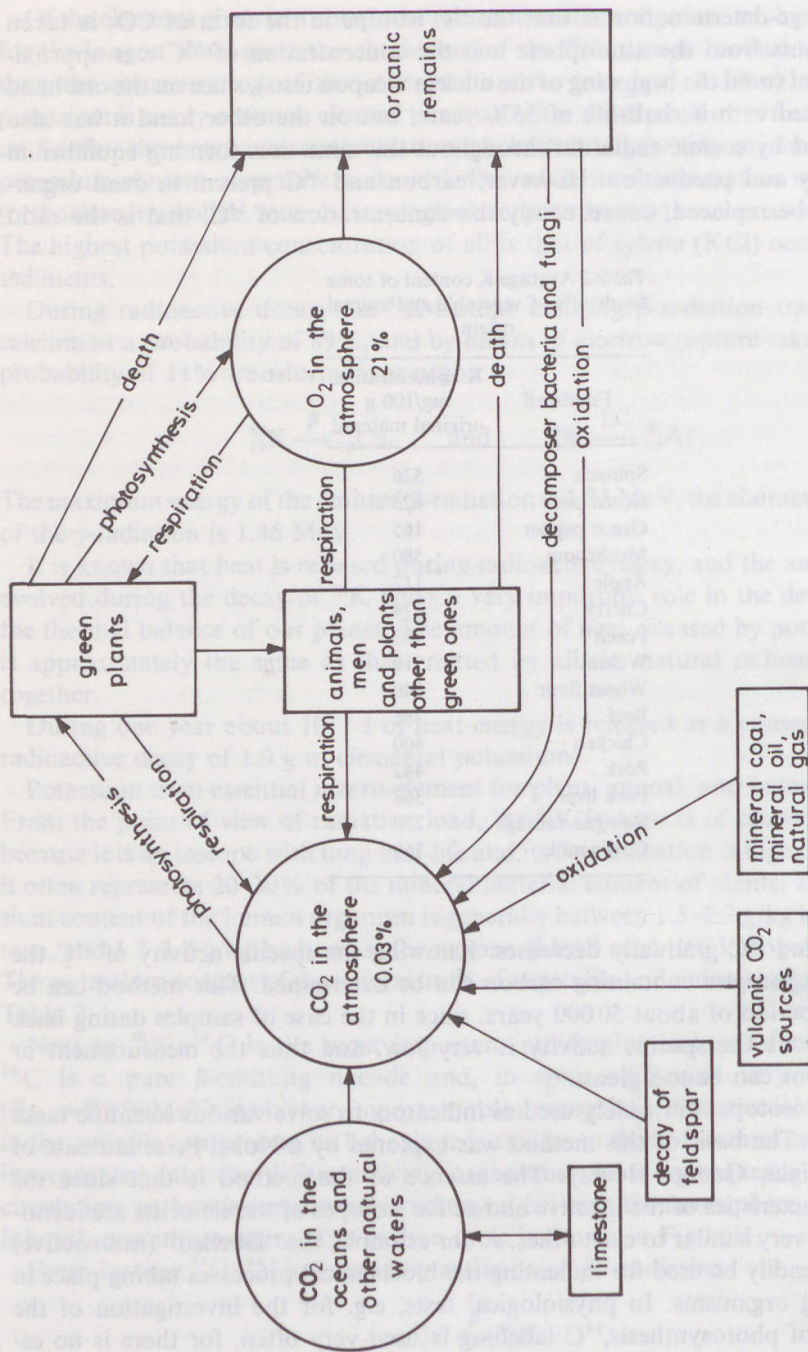
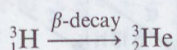


Fig. 1. The carbon cycle in nature

cyclotrons—and a wide selection of radioactive isotopes are utilized. In addition to the ^{14}C isotope already mentioned above, the application of for example, ^{32}P and ^{35}S isotopes is very common in studies of the changes of phosphorus and sulphur content.

Naturally the difference—the unequal number of neutrons, that is the different mass number—which, in addition to radiation, also makes a distinction between the inactive and active isotopes of the same element, results in certain differences as regards the physical and chemical characteristics (melting point, boiling point, density, diffusion, etc.). However, this difference is generally small to such an extent that in most of the cases it is even negligible. The smaller the relative difference between the mass numbers, the smaller the difference between the properties of the isotopes. For example, the relative mass difference between ^{12}C and ^{14}C is 16.7%, between ^{31}P and ^{32}P 3.2%, and ^{39}K and ^{40}K 2.6%. The greatest difference is between the isotopes of H; radioactive ^3H (tritium) apart from the proton consists of two neutrons, and deuterium, marked by ^2H or D consists of one proton and one neutron, while ordinary H consists of one proton only. Thus the mass number of ^3H used for the indication of hydrogen is three times that of ordinary hydrogen, so as a consequence of the very considerable difference between the mass numbers there is a relative greater—and not always negligible—difference between other characteristics as well.

Tritium is a pure β -emitting isotope with very small radiation energy, the maximum energy of the emitted β -particles is only 0.018 MeV. During its decay, helium is produced from tritium.



The activities of the rest of the radioactive isotopes of natural origin listed in Table 1 are considerably less than those of ^{14}C and ^{40}K isotopes, so their contribution to the radiation load is very small.

On Earth the radioactive isotopes of natural origin can be divided into three groups. One of them is the group of radioactive isotopes produced by cosmic radiation. The half-life of these radioactive isotopes is relatively short, consequently their decay is rapid.

The second group comprises isotopes of long half-life, the formation of which is already connected to the evolutionary period of the Earth as a planet. The radioactive isotopes that can be placed in the first and second groups are shown in Table 1. All these isotopes belong to the nuclides of light and medium mass.

The third group consists of the class of heavy nuclides, the initial isotopes (primeval elements) of these having existed as early as the cosmogonical stage of the evolution of the Earth. The isotopes belonging to the third group originate from either of the three radioactive isotopes ^{232}Th , ^{238}U and ^{235}U which can be found in relatively large quantities in nature, and are the product of their successive radioactive decay. The three radioactive decay series (families) are called thorium, uranium and actinium series, or, on the basis of the divisibility of the mass number of the isotopes by four, are sometimes also called as $4n$ or $(4n+0)$, $4n+2$ and $4n+3$ series. Formerly there

The series starting with ^{232}Th , ^{235}U and ^{238}U isotopes through successive α - and β -decays at the end are transformed into the stable isotopes of lead with mass numbers of 208, 207 and 206, respectively. The final product of the neptunium series, which can be considered as a decayed one, is the stable bismuth isotope with a mass number of 209.

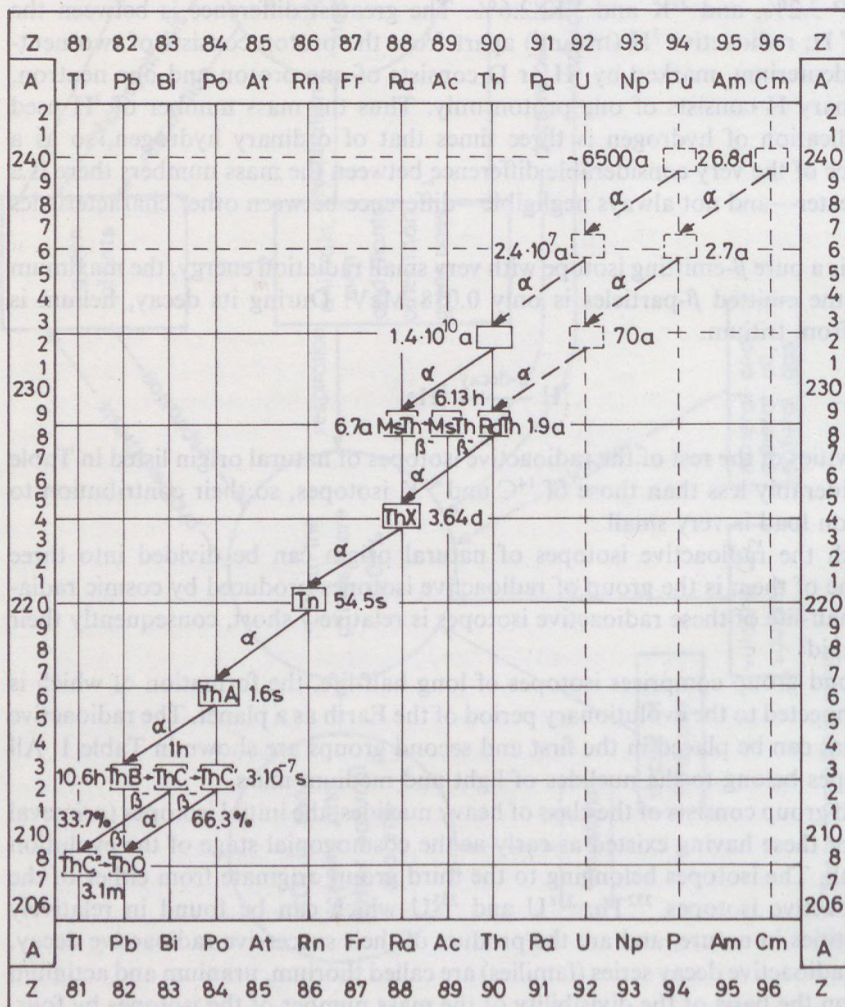


Fig. 2. Radioactive decay series of thorium ($4n+0$)

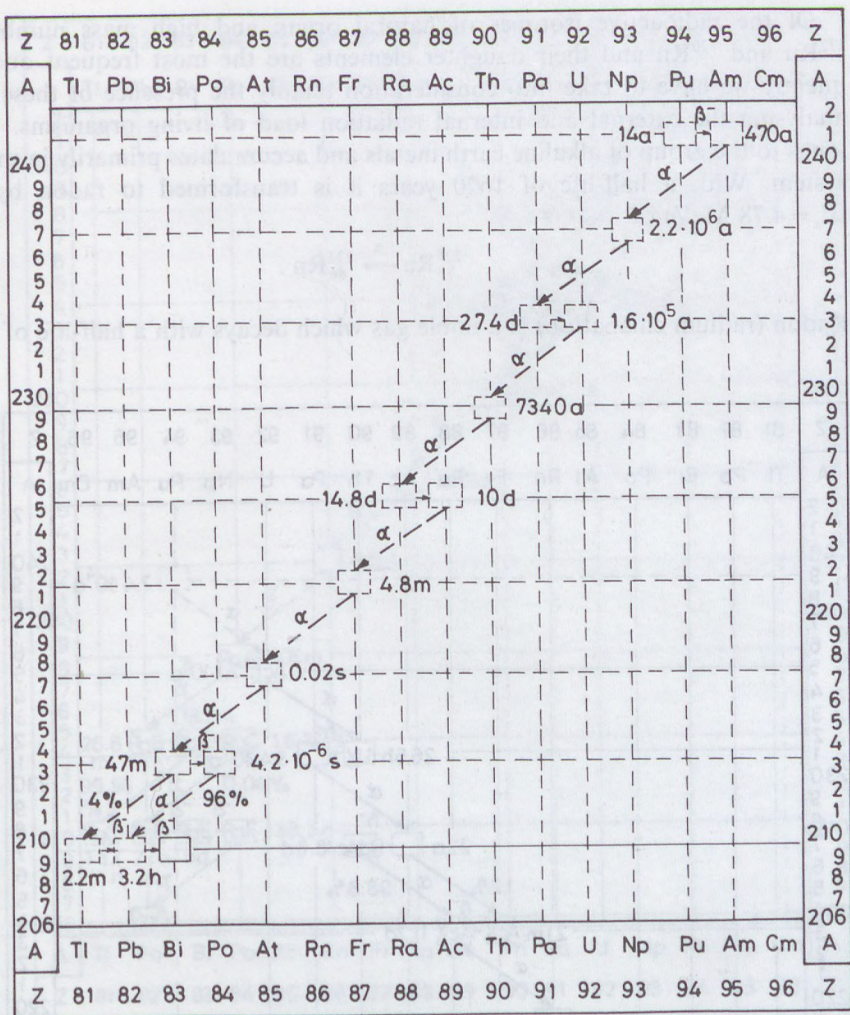
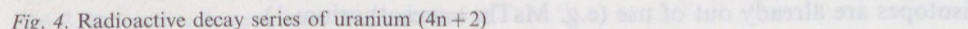
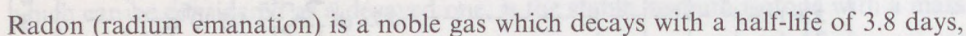
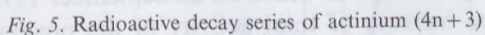


Fig. 3. Radioactive decay series of neptunium ($4n+1$)

The schemes of the four radioactive decay series are shown in Figures 2–5. In the figures the members of the neptunium series already decayed and no longer existing (Figure 3) and the series whose original element have already decayed (Figures 2, 4, 5) are marked by a broken line. “Z” on the horizontal axis symbolizes the serial number, while “A” on the vertical axis shows the mass number. In addition to the atomic number the chemical symbols of the elements are also given.

Most of the members of the three radioactive decay series which can be found in nature have (used to have) a name, the symbol of which can be seen in the square corresponding to the different members. However, the common names of the natural isotopes are already out of use (e.g. MsTh_1 = mesothorium 1).

$${}_{88}^{226}\text{Ra} \xrightarrow{\alpha} {}_{86}^{222}\text{Rn}.$$




^{210}Pb and ^{210}Po are members of the same decay series; lead can also accumulate in the bone-tissues. The ^{210}Pb -isotope is a β - and γ -emitting one with a half-life of 21.4 years, the maximum energy of β -radiation is 0.017 MeV, while that of γ -radiation is 0.047 MeV. ^{210}Po emits α -radiation ($E_\alpha = 5.31$ MeV); its half-life is 138.4 days. The decay of ^{210}Pb and ^{210}Po is as follows:



1.3 RADIOACTIVE ISOTOPES OF ARTIFICIAL ORIGIN IN THE BIOSPHERE

There are several radioactive isotopes of natural origin in the biosphere as a consequence of which living organisms are affected by a certain level of radiation load. In addition the utilization of nuclear energy was unfortunately accompanied also by a very considerable amount of nuclear contamination, so that today our environment is also contaminated by different sorts of artificial radioactive isotopes. The radioactive isotopes of artificial origin come from the following sources:

- nuclear weapon tests;
- operation of nuclear power plants;
- reprocessing;
- isotope laboratories.

The nuclear weapon tests were chiefly responsible for the development of global radiation contamination of the biosphere. When high energy tests are carried out irrespective of whether they are surface or stratospheric tests, a considerable amount of the radioactive materials from the atomic cloud rise to the stratosphere (the part of the atmosphere at a height beyond 10 km), and due to the slow fallout the radioactive isotopes can reach almost any point on Earth. Since the velocity of radioactive materials flowing in the stratosphere after a test is 20–25 m/s, one or two weeks after the nuclear test the presence of fission products can be detected in the air or in precipitation samples taken at great distances from the site. The distribution of the radioactive contamination is determined by stratospheric processes, there is little danger of radioactive material passing from the northern to the southern hemisphere or vice versa. The effect of the Chinese nuclear tests that started in 1964 was already detectable e.g., in the German Democratic Republic 10–18 days after the nuclear weapon tests (Zinder 1969, 1970). To give another example, in Hungary the radioactive contamination of the environment increased as early as October as a consequence of the atmospheric nuclear test performed in China on 26 September 1976. A recent nuclear test is indicated by the increase of total activity of the precipitation and fallout, the increase of the $^{89}\text{Sr}/^{90}\text{Sr}$ ratio, and the presence of ^{144}Ce , $^{95}\text{Zr} + ^{95}\text{Nb}$ and ^{131}I detectable by γ -spectrometry. In spite of the fact that the radioactive contamination of vegetable samples (spinach, lettuce, sorrel) collected in October 1976 was, as a consequence of the Chinese nuclear weapon test, higher by an order of magnitude than that of the samples taken in the preceding months, it did not result in a prolonged increase of radioactive contamination as the majority of the isotopes causing the pollution were of a short half-life.

On the basis of their origin, the radioactive isotopes produced in a nuclear explosion can be divided into three groups:

- the scattered fraction of the charge which did not take part in the nuclear fission (^{235}U , ^{238}U , ^{239}Pu);
- radioactive materials produced in the ambient materials (bomb shell, structural

units, soil, building structures) resulting from different nuclear reactions as a consequence of the neutron flux (also called induced radioactivity);

– fission products.

Of these, the fission products are of the greatest importance as regards both their quantity and danger. The fission of U and Pu nuclei can take place in many different ways, resulting in more than 60 primary fission products. The primary products can get rid of their excess neutrons through decay series of 3–4 members and by emitting β -radiation (and concomitantly γ -radiation).

The mass number of the isotopes produced in a nuclear fission is between 70 and 170. The distribution of the fission products of ^{235}U as a function of the mass number is shown in Figure 6. Figures 7, 8 and 9 show what ratio of the total radioactivity is represented by the fission products originating from a nuclear test at different times following the test. The rise of fission products is described by the $\text{U}(n, f)\text{X}$ reaction, where the (n, f) reaction stands for the fission produced by the effect of the neutrons, and X is the product nucleus produced in the fission.

The following empirical formula was proposed by Way and Wigner to express the time-dependence of the radioactivity of the fission products:

$$I_t = I_0 \cdot t^{-n},$$

where:

I_t = radioactivity of the fission products after time t following the explosion;

I_0 = radioactivity immediately after the explosion;

$n = 1.2$.



Fig. 6. Distribution of the fission products of ^{235}U as a function of the mass number

On the basis of two measurements of radioactivity on the same sample (I_1 and I_2) carried out at two different points of time (t_1 and t_2); t_1 = that is the time which passed since the nuclear test; t_2 = time elapsed since measurement I_1 ; $\Delta t = t_2 - t_1$ can be calculated by means of the following formula:

$$t_1 = \frac{\Delta t}{\sqrt[n]{\frac{I_1}{I_2} - 1}}$$

The applicability of the Way–Wigner formula has been experimentally confirmed by several authors. For example in Hungary, using the precipitation radioactivity measurements taken in the Nuclear Research Institute of Debrecen since 1952 the value of n was found to be 1.18 ± 0.10 , and on the basis of the formula the time of a nuclear weapon test could be ascertained with an accuracy of 2 days. Note that the first nuclear weapon tests by the United States, the Soviet Union, Great Britain, France and China were carried out in 1945, 1949, 1952, 1960 and 1964, respectively.

The radioactivity of the fission products decreases considerably as a function of time because the majority of the produced radioactive isotopes are of a short half-life. For example, after 24 hours following the nuclear test a decrease in radioactivity of almost 5 orders of magnitude can be observed. In a nuclear weapon test with a TNT

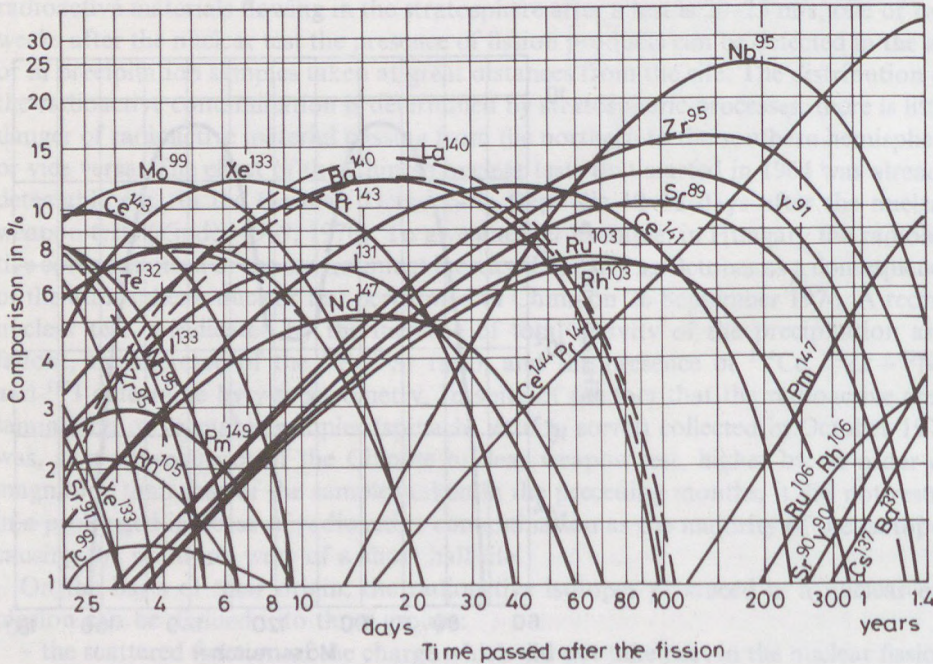


Fig. 7. Proportion of the radioactive isotopes produced in the fission of ^{235}U in the total radioactivity as a function of the time passed after the explosion

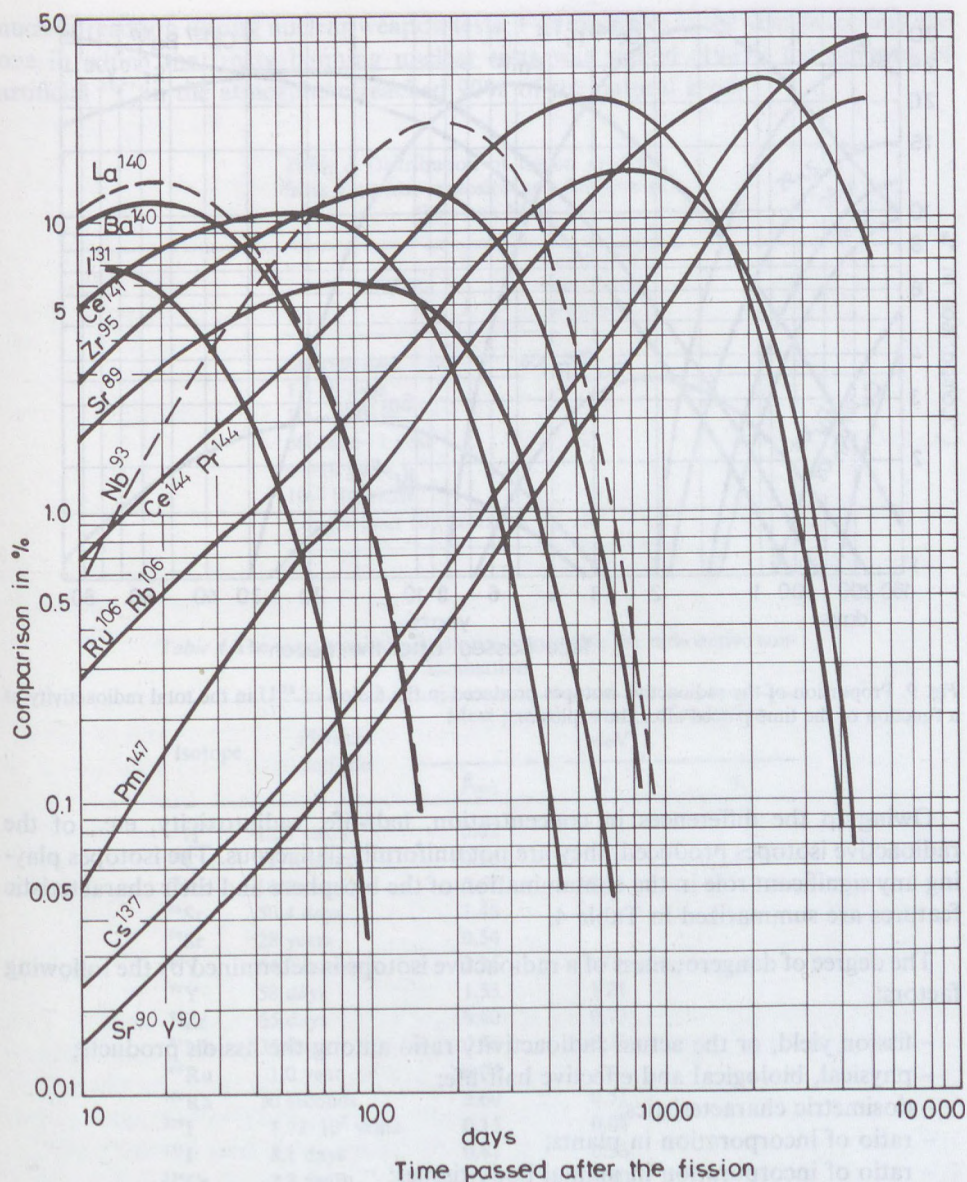


Fig. 8. Proportion of the radioactive isotopes produced in the fission of ^{235}U in the total radioactivity as a function of the time passed after the explosion

equivalent of 1000 t, 49.5 g fission product is released, the radioactivity of which is $3.7 \cdot 10^{21}$ Bq 1 minute after the test, while after 24 hours it is $5.9 \cdot 10^{16}$ Bq. The distribution of the radioactive products of the explosion as a function of their half-life is shown in Table 3.

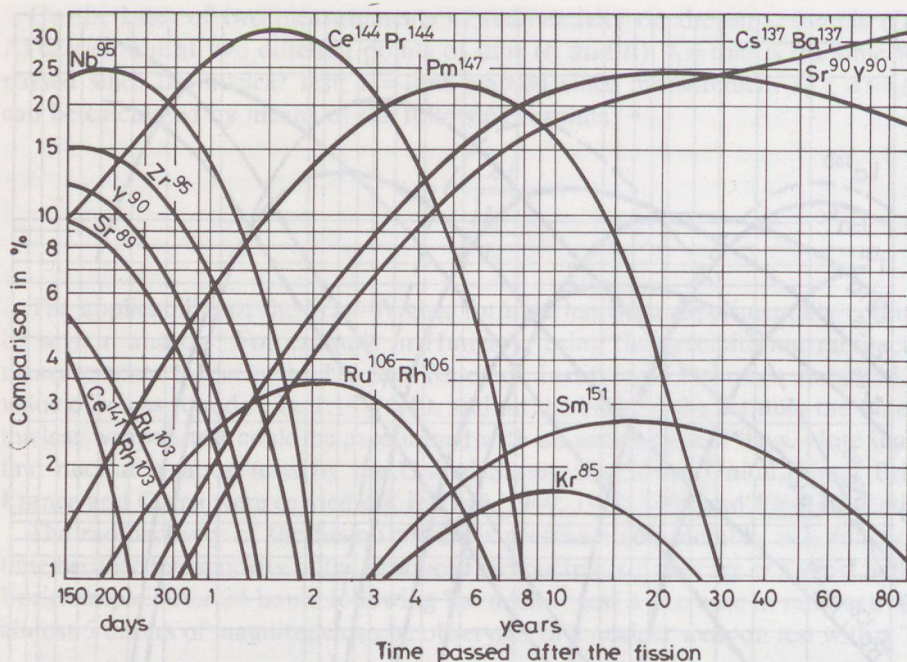


Fig. 9. Proportion of the radioactive isotopes produced in the fission of ^{235}U in the total radioactivity as a function of the time passed after the explosion

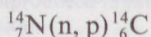
Owing to the differences in concentration, half-life, radiotoxicity, etc., of the radioactive isotopes produced, they are not uniformly dangerous. The isotopes playing any significant role in the contamination of the biosphere and their characteristic features are summarized in Table 4.

The degree of dangerousness of a radioactive isotope is determined by the following factors:

- fission yield, or the actual radioactivity ratio among the fission products;
- physical, biological and effective half-life;
- dosimetric characteristics;
- ratio of incorporation in plants;
- ratio of incorporation in animal body tissues;
- ratio of incorporation in human organisms.

Taking these factors into account it can be stated that lasting radiation load is caused only by ^{14}C , ^{90}Sr and ^{137}Cs .

The ^{14}C isotope has already been mentioned in the review of the natural radioactive isotopes. However, the amount of natural radiocarbon has been significantly increased by ^{14}C produced in the



nuclear reaction during nuclear weapon tests. For example, in the year preceding the one in which the treaty banning nuclear tests was signed (1963), the amount of artificial ^{14}C in the atmosphere reached 90% of the natural level.

Table 3 Distribution of fission products from a nuclear explosion as a function of their half-life

Half-life	Number of radioactive isotopes
Shorter than 1 day	131
1 – 10 days	17
10 – 30 days	9
30 days – 1 year	12
1 – 10 years	7
10 – 100 years	3
Longer than 100 years	10

Table 4 The most significant isotopes responsible for radioactive contamination

Isotope	Physical half-life	Most probable energy of radiation (MeV)		
		$\beta_{\max}$	γ	α
^1H	12.3 years	0.02		
^{14}C	5570 years	0.16		
^{85}Kr	10.8 years	0.67	0.52	
^{89}Sr	50.4 days	1.46		
^{90}Sr	28 years	0.54		
^{90}Y	64.2 hours	2.26		
^{91}Y	58 days	1.53	1.21	
^{95}Zr	65 days	0.40	0.73	
^{95}Nb	35 days	0.16	0.76	
^{106}Ru	1.0 year	0.04		
^{106}Rh	30 seconds	3.60	0.51	
^{129}I	$1.72 \cdot 10^7$ years	0.15	0.04	
^{131}I	8.1 days	0.61	0.36	
^{134}Cs	2.3 years	0.66	0.80	
^{137}Cs	30 years	0.51	0.66	
^{140}Ba	12.8 days	1.02	0.03	
^{140}La	40.2 hours	1.36	1.60	
^{144}Ce	285 days	0.32	0.13	
^{144}Pr	17.5 minutes	2.98	0.69	
^{147}Pm	2.6 years	0.22		
^{147}Nd	11.1 days	0.81	0.09	
^{239}Pu	$2.44 \cdot 10^4$ years	0.05		5.15
^{240}Pu	6580 years	0.04		5.16

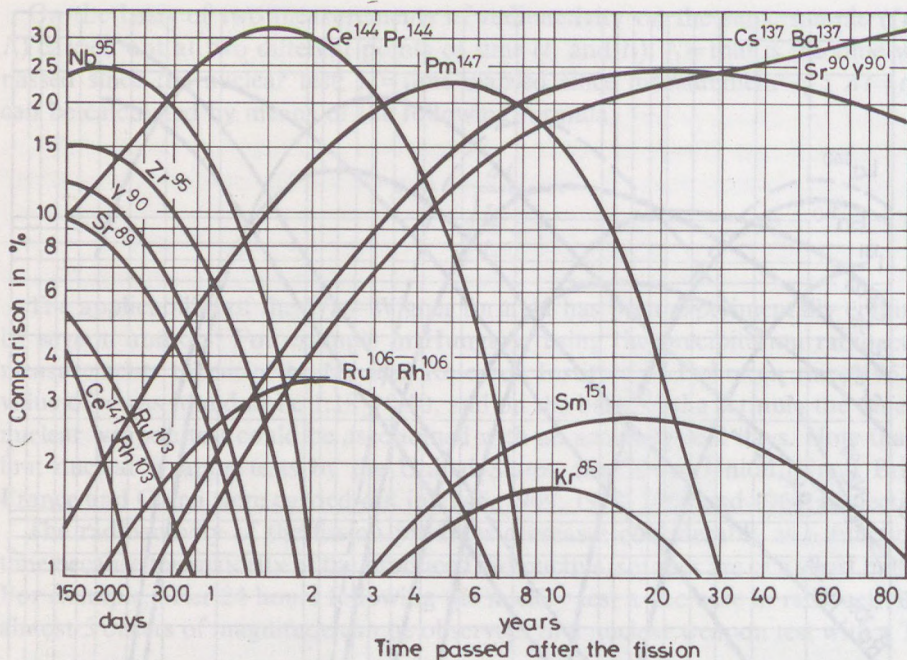


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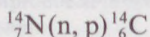
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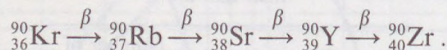
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^{89}Sr	50.4 days	1.46		
^{90}Sr	28 years	0.54		
^{90}Y	64.2 hours	2.26		
^{91}Y	58 days	1.53	1.21	
^{95}Zr	65 days	0.40	0.73	
^{95}Nb	35 days	0.16	0.76	
^{106}Ru	1.0 year	0.04		
^{106}Rh	30 seconds	3.60	0.51	
^{129}I	$1.72 \cdot 10^7$ years	0.15	0.04	
^{131}I	8.1 days	0.61	0.36	
^{134}Cs	2.3 years	0.66	0.80	
^{137}Cs	30 years	0.51	0.66	
^{140}Ba	12.8 days	1.02	0.03	
^{140}La	40.2 hours	1.36	1.60	
^{144}Ce	285 days	0.32	0.13	
^{144}Pr	17.5 minutes	2.98	0.69	
^{147}Pm	2.6 years	0.22		
^{147}Nd	11.1 days	0.81	0.09	
^{239}Pu	$2.44 \cdot 10^4$ years		0.05	5.15
^{240}Pu	6580 years		0.04	5.16

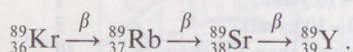
Of the radioactive fission products—from the point of view of the long-lived internal contamination of the living organisms—the ^{90}Sr isotope is one of the most important ones. Sr, which is an alkaline earth-metal—chemically it is similar to Ca—and the Sr isotopes, together with Ca, can be accumulated in the living organisms in significant amounts.

The production and decay scheme of ^{90}Sr is as follows:



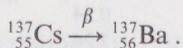
^{90}Sr has a long half-life, so it is in equilibrium with the ^{90}Y isotope, its daughter element of a relatively short half-life.

The development and decay scheme of ^{89}Sr , another Sr isotope of major importance, is similar to that of ^{90}Sr with the exception that ^{89}Y is not radioactive. Thus the scheme is as follows:



From the point of view of radiation hygiene, ^{89}Sr is of lesser importance since its half-life is significantly shorter than that of the ^{90}Sr isotope, and it constitutes danger only for about one year following a nuclear weapon test, while ^{90}Sr can be dangerous for decades.

The ^{137}Cs isotope also plays a crucial role in the emergence of the radioactive contamination of the biosphere. The concentration in which it occurs and its half-life are similar to that of the ^{90}Sr isotope; however, chemically it is an alkaline metal, and takes part in the metabolic processes similarly to potassium, forming with it a common transport system. As regards contamination, ^{137}Cs is the most important one since it results in an effective dose equivalent commitment four times larger than that of ^{90}Sr (UNSCEAR 1982). The ^{137}Cs decays by β -decay to barium isotope:



Of the fission products resulting as a consequence of the nuclear weapon tests, ^{95}Zr , ^{106}Ru , ^{140}Ba and ^{144}Ce isotopes, and their radioactive daughter elements ^{95}Nb , ^{106}Rh , ^{140}La and ^{144}Pr isotopes, and the ^{131}I nuclide must be mentioned. However, these isotopes—due to their relatively short half-life—are dangerous only during the first few months following the nuclear weapon test; later on their contribution to the contamination level decreases.

As a result of nuclear weapon tests our environment is also contaminated by several plutonium isotopes, of which, from the point of view of their radiation effect, ^{239}Pu and ^{240}Pu isotopes are of major importance. The ^{239}Pu and ^{240}Pu isotopes occur together, about 60% of the activity of their mixture originates from the ^{239}Pu isotope (Marey 1980). A considerable amount of $^{239}\text{Pu} + ^{240}\text{Pu}$ in the biosphere originates from the nuclear weapon tests performed until 1963. Up to 1973 the amount of $^{239}\text{Pu} + ^{240}\text{Pu}$ produced by nuclear weapon tests and reaching the surface of the Earth and the hydrosphere from the atmosphere, could be assessed at about

$1.2 \cdot 10^{16}$ Bq activity (Harley 1971; Wrenn 1974). The amount of $^{239}\text{Pu} + ^{240}\text{Pu}$ that can be found in the stratosphere is about two orders of magnitude smaller.

In the radioactive contamination of the biosphere, nuclear power plants and nuclear facilities play a role that is not unimportant since occasionally they emit a relatively significant amount of radioactive materials to the atmosphere and the surface waters. For example, in the case of a pressure water reactor with a capacity of 1000 MW_e—leaving tritium out of consideration—the possibility of discharging liquid form radioactive waste of about $1.9 \cdot 10^5$ Bq/hr radioactivity is reckoned with, which on an annual basis means an activity of $(1.5\text{--}1.9) \cdot 10^9$ Bq.

The radioactivity of the major fission products generated during one year's operation of an atomic power station of 440 MW_e is shown in Table 5. The reactor at Paks, Hungary, of a total output of 1760 MW_e has been built of such blocks.

Table 5 Radioactivity of the fission products developed in the fuel units of a light-water power plant of 440 MW_e capacity during one year of operation

Radioactive isotope	Radioactivity (Bq)
Total	$2.7 \cdot 10^{19}$
Volatile noble gas isotopes	$5.6 \cdot 10^{18}$
^{131}I	$5.6 \cdot 10^{17}$
^{89}Sr	$1.1 \cdot 10^{18}$
^{90}Sr	$1.9 \cdot 10^{16}$
^{137}Cs	$1.8 \cdot 10^{16}$
^{144}Ce	$5.2 \cdot 10^{16}$

Unfortunately, in addition to the low level of radioactive emission during normal operation, in the case of serious disasters in nuclear reactors or nuclear power stations, the release of a very significant amount of radioactive material must be reckoned with. Although the possibility of serious level contaminations, due to the strict safety rules, is very small, the example of Chernobyl (26 April, 1986) has regrettably proved that it cannot completely be excluded. It is also a fact that in the history of nuclear power engineering there have already been several minor or major breakdowns or accidents, but of these it was only the Chernobyl catastrophe which caused dozens of casualties and caused a very high level contamination over a huge area. I will not go into details here—later on the contamination effect of the disaster at the Chernobyl nuclear power plant will be discussed—but I feel it necessary to emphasize at this point that in May 1986 in most of the European countries the radiation contamination level of the environment was at least an order of magnitude higher than that developed as a consequence of atmospheric weapon tests. As is known, the maximum of the latter was observed in Europe between 1961 and 1964.

Special attention should be paid to the question of contamination from nuclear reactors if we take it into account that in the year 2000 about 20–25% of the electric energy requirement of the world is expected to be covered by nuclear energy. Previous

forecasts estimated that the ratio of nuclear energy would reach 40–50% by the turn of the millennium, however, in most countries of Western Europe the rate of construction of nuclear power plants is considerably slower than was planned. For example, in 1973 in Western Europe 59 nuclear power plants were operating at a total capacity of 15 GW. Until 1981, 91 further power plants had been planned to be put into operation with a total capacity of 79 GW; however, by the end of 1981 only 47 were actually operational with a total capacity of 39 GW (Jeffs 1981).

It is worth mentioning that the radioactive emission of the Paks nuclear power plant in Hungary is at a very low level. The concentration of the radioactive isotopes in the Danube downstream from the power plant is significantly below the drinking-water standards and reference levels. However, it is a fact that in the case of nuclear power plants atmospheric release is generally of much greater importance than the liquid form emission. The subjects of both the migration and the radiation load effect of the radioactive isotopes being released into the environment from the nuclear facilities are discussed in detail by the UNSCEAR reports, and also several scientific conferences were held dealing with the ecological questions of emissions from nuclear facilities. Recently the number of facilities using radioactive isotopes—A-, B-, and C-level laboratories—has increased almost exponentially in all countries of the world. It is also necessary to mention that contamination, primarily of a local nature, may also be caused by an inadequate treatment of radioactive wastes or negligence.

1.4 PROTECTION AGAINST RADIOACTIVE CONTAMINATION AS AN INTEGRAL PART OF ENVIRONMENTAL PROTECTION

When one hears or reads the phrase “protection of the environment”, probably one or another of its special fields come to mind. These include the problem concerning the protection of the purity of the air and water, soil conservation, protection from noise, the deposition and utilization of waste materials and, in addition to the above, perhaps also the protection of nature, including the protection of national parks, landscape or nature conservation areas, in order to preserve their scenic beauty, native flora and fauna. Of course, there is no rigid borderline as the different fields cannot be sharply separated from each other, since for example, not only noise contamination is caused by the motor vehicles, but from the exhaust gases toxic lead can also get into the air, and from there either by inhalation, or through the nutrition chain, directly or indirectly reaches the human organism. Or, for example SO_2 , which is emitted into the atmosphere primarily as the result of coal-burning, is considered to be typically an air contaminating component. It can also cause very significant, and often unfavourable, changes in the soil. The SO_2 gas in the atmosphere is precipitated to the soil in the form of highly acidic rain, which can significantly reduce soil pH levels and thereby adversely affect fertility. Consequently, efficient protection of the environment will require the application of very careful, and from all points of

view complex methods, taking into consideration the interactions among the different constituents of the biosphere.

The application of even more complex methods is required in the protection against radioactive contamination, since the transport and migration of the radioactive isotopes among the different ecosystems and the different elements of the biological chain process are at times very rapid processes indeed.

The following are the major events in the development of the different levels of radioactive contamination:

The level of radioactive contamination due to nuclear weapon tests reached its maximum between 1961 and 1965, then it began to decrease, and in recent years it can be considered as constant (Miserez 1967; Scheel et al. 1967; Ocker 1968; Poulheim 1969; Gedeonov 1972; UNSCEAR 1977, 1980, 1982, 1988).

However, it must be mentioned that in the autumn of 1976, as a consequence of the Chinese nuclear weapon tests, the contamination level increased again. However, since a considerable proportion of the isotopes causing the contamination were of a short half-life (e.g. ^{89}Sr , ^{140}Ba), no lasting increase in the level of radioactive contamination was observed in fall-out, rainwater, and vegetation. On the other hand, no significant increase of contamination was noted in the other components of the biological chain process (animal, man), which is due to the fact that accumulation of the radioactive isotopes in the tissues of the animals and humans is a process which requires a relatively longer period of time, during which a significant proportion of the isotopes of shorter half-life could decay.

Further regular monitoring of the level of radioactive contamination and preparation for protective measures are also justified in the future. The reasons for this are as follows:

(a) Since every year only about 10% of the radioactive materials reaching the stratosphere are deposited onto the surface of the Earth (thus the "half-life" of radioactive fall-out is about 7 years) so contamination can still be caused by the previous nuclear weapon tests. So for example, 1–2% of the radioactive materials lifted into the stratospheric layers as a consequence of the nuclear bombs dropped on Hiroshima and Nagasaki in 1945, has not yet fallen back to the Earth.

(b) The nuclear powers (France, China, India), which have not joined the treaty on banning nuclear tests, still continue with their nuclear weapon tests.

(c) Due to the rapidly increasing number of the nuclear power plants—now there are about 600 research reactors and about 300 high-capacity nuclear power plants in the world—the amount of radioactive materials emitted by the nuclear reactors during normal operation also increases.

(d) A considerable amount of radioactive material may be leaked into the environment if there is a disaster at a nuclear reactor.

Finally it must be mentioned that radioactive material can also get into the environment from laboratories using radioactive isotopes. In addition to the above, more and more people work with radioactive materials or equipment causing radiation load (e.g. X-ray apparatus, accelerators, reactors), and in the vicinity of these facilities, it is a task of top priority to provide adequate protection for them against both somatic and genetic radiation damage.

Today it is not necessary to stress the importance of the protection of the biological environment. It is evident that one of the most important pledges of our human and social development and future is a clean and healthy environment, a biosphere free from toxic materials. Therefore our environment and the human being himself, will be protected from some of the dangerous effects of human activities, from smog, soot, toxic heavy metals, carcinogenic materials, refuse and radioactive materials.

The contaminating of the biosphere from an emission source often cannot be localized to a smaller or greater area, but the contamination, without respect for state boundaries can also extend over continents. Moreover, we can also speak about global contamination affecting the whole of the Earth. In such a case protection of the environment coordinated at international level is necessary. It would be in vain for some of the individual countries do everything to moderate environmental pollution by the emission sources, if from neighbouring countries highly contaminated air-masses and surface waters containing large amounts of pollution continue to enter the country or territory in question. This is even more so in the case of protection against radioactive contamination, since, for example, if a surface nuclear weapon test takes place as far away as somewhere in the Pacific Ocean, for after a few weeks the presence of the fresh fission products will also be detected in Europe.

Our environment is contaminated not only by nuclear weapon tests but also by the radioactive emission of nuclear reactors. Nevertheless it is true that in this way, presuming a normally operating reactor, the radioactive materials contaminating the atmosphere and the hydrosphere will affect a smaller area only. Again, the radioactive materials from laboratories using radioactive isotopes contaminate their immediate vicinity, and this activity is usually considerably smaller than the amount of fission products generated by nuclear weapon tests or by nuclear power plants.

The maximum amount of active materials escaping into the environment from nuclear facilities and laboratories using radioactive isotopes is regulated by legislation based on rules of health physics and work safety. Of course, certain compromises are inevitable. Although it is possible to build a nuclear power plant with an ideally small amount of radioactive emission, in practice such solutions are necessary in power engineering which can keep the costs within feasible range. As consequence it has to be accepted that a certain—preferably minimum—amount of active material will reach some components of the biosphere from nuclear facilities and isotope laboratories.

It is completely different if surface or stratospheric nuclear weapon tests are taken into consideration. Since their contamination will affect the whole of the Earth, banning nuclear weapon tests is a question concerning the whole of mankind. In 1963 a treaty was signed in Moscow partially banning nuclear weapon tests—partial because it did not include underground tests. However, some countries have not joined the convention and continue their nuclear weapon tests even today. For this reason the most important constituent in an efficient system of protection against radioactive contamination of the biosphere would be to extend the treaty banning nuclear tests to all countries. Accordingly, protection against radioactive contamination, similarly to other fields of environmental protection, can be solved successfully only by means of international cooperation.

1.5 SUMMARY

In this chapter radioecology has been presented as a special field of ecology, which, however, covers the whole biological chain process. The more important radionuclides of natural and artificial origin have been reviewed and I have emphasized the necessity of considering protection against radioactive contamination as an essential part of environmental protection.

Statements of major importance are as follows:

1. Radioecology is a special field of the science of ecology which analyses the interactions between the environment containing radioactive materials and living organisms therein. Its subject is the migration of radioactive materials in the air–water–soil–plant–animal–man biological chain process, and the analysis of the effects of the different factors on the changes of concentration of the radioactive isotopes.

2. Radioactive isotopes of natural origin are either:

(a) the decay products of radioactive isotopes of an exceedingly long half-life, which have existed already in the cosmogenic state (heavy nuclei), or

(b) are isotopes of a long half-life, which have come into being during the planetary development period (medium-weight nuclei), or

(c) are nuclides of a relatively short half-life, which are generated by cosmic radiation (light nuclei).

Of these, considering the radiation load of the biosphere, ^{40}K , ^{14}C , ^3H , and ^{226}Ra , ^{222}Rn and the daughter elements of a short half-life of the latter two are of primary importance.

3. The nuclear weapon tests are mainly responsible for the aggravation of global radiation pollution of anthropogenic origin affecting the biosphere. After a high-power nuclear weapon test the majority of the generated fission products rise to the stratosphere and from there, within a relatively short time, can reach practically any point on Earth. Lasting radioactive contamination can essentially be attributed to two radionuclides, ^{90}Sr and ^{137}Cs , with a half-life of 28 and 30 years, respectively.

4. In 1986, as a consequence of the disaster at the Chernobyl power plant, very high contamination levels could be measured in several countries of Europe. The Chernobyl disaster resulted, in the majority of European countries, in elevating the contamination levels to one order of magnitude higher than the levels due to nuclear weapon tests at the beginning of the 1960s.

5. Since contamination from the nuclear tests affects the whole of the Earth, prevention of further pollution of our planet is a question concerning the whole of mankind. Protection against radioactive contamination, similarly to the other fields of environmental protection, requires international cooperation, the most important issue of which in the field in question would be adherence to the provisions of the Moscow Treaty on banning nuclear tests. Furthermore, it is important to guarantee the reliable operation of nuclear power plants and nuclear facilities, to observe under all circumstances the appropriate rules of work safety and through this to prevent any situation of hazard which could lead to contamination, and to minimize the likelihood of the occurrence of disasters which could result in contamination.

2. FACTORS INFLUENCING THE RADIOACTIVITY OF THE BIOSPHERE

The main questions are: how is radioactivity modified in the connected components of the biological chain process, and what are the factors influencing the level of contamination. This is the most important part of this book, both in extent and significance. Knowledge of the facts written here gives us the opportunity not only to have an overall appreciation of radioecology but even to go deeply into it; not only to take some introductory steps but by proceeding on certain special fields obtaining a more profound knowledge of this field of science.

In this chapter, in order to meet the requirements of investigations on the metabolic processes among the sequential components of the biological chain process, those factors will be examined which contribute to the change in concentration of radioactive isotopes when going from one step to another in the sequence of metabolism. The air–water–soil–plant–animal–man biological chain will be investigated, data on the level of radioactivity of these components of the biosphere will be listed and the ecological factors influencing the increase or decrease of concentration of the radioactive elements will be discussed. The so-called discrimination factors will be discussed in relation to the concentration change of radioactive isotopes within metabolic processes between soil and plant, as well as plant and animal. The influences of agrotechnical and food-technological operations on radioactive contamination will be emphasized.

The migration of the various radionuclides among the components of the biosphere is rather different, therefore not many general statements relevant to all or most radioisotopes can be made. (This is not a surprise as the only common trait between the radionuclides is the fact that all are radioactive; the chemical behaviour of the radioactive substances is determined by their position within the periodic system of the elements. Thus, for example, there is not much similarity between ^{14}C and ^{89}Sr , or ^{131}I and ^{144}Ce .) In this book the main emphasis is placed upon the examination of migration, distribution and accumulation of the isotopes ^{90}Sr and ^{137}Cs , which play a very important role in the lasting radioactive contamination of the biosphere. However, some data will also be presented about the concentration and migration of the other radioactive isotopes and factors determining their concentration.

2.1 RADIOACTIVITY OF THE ATMOSPHERE AND INFLUENCING FACTORS

As a consequence of the growing use of nuclear energy and radioactive isotopes, there is a real danger of a significant amount of radioactive substances being released into the atmosphere. Some of the radioactive substances in the atmosphere are in gaseous state, others float in adhesion to aerosols.

As is well known, the radioactive substances within the atmosphere, based on their origin, can be classified into two groups. To the first group belong the radioactive isotopes of natural origin, present in the atmosphere independently of any human activity, the other group is formed by the active substances produced by the activities of mankind.

The first group is formed by gaseous products of natural radioactive materials which enter the air usually from the soil and the hydrosphere, mostly by thermal exchange. Even the active isotopes produced by cosmic radiation can be classified here.

Part of the substances belonging to the second group are released into the atmosphere as a consequence of nuclear tests, others are produced through the operation of nuclear power plants as well as the processing of their used fuel rods. It must be emphasized that the contamination caused by nuclear tests is actually decreasing since

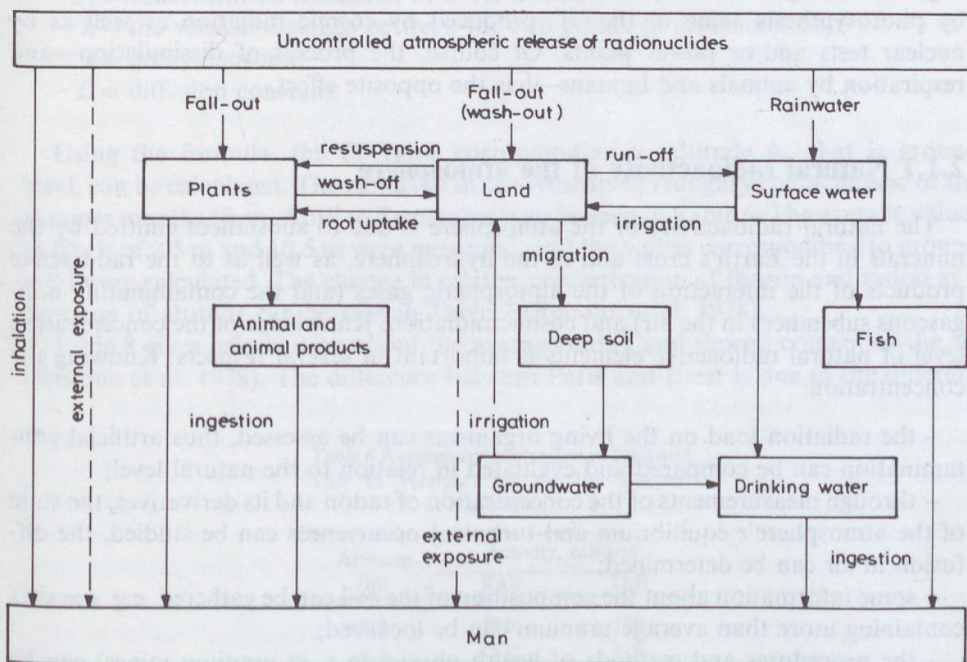


Fig. 10. Major pathways of radionuclides to man due to uncontrolled release of radioactivity

the treaty banning nuclear tests was signed in Moscow; however, the amount of active substances emitted by nuclear power plants, because of their growing number, is increasing significantly. The radioactive uranium and plutonium contamination of the atmosphere is further increased by satellites with nuclear materials which, when returning into the atmosphere, burn away.

In addition to the sources producing radioactive materials, the radioactivity of the atmosphere is influenced by climatic and meteorological circumstances. The geographical position of a given area, the velocity and direction of wind, the amount, nature and frequency of precipitation, even the air pollution by other substances (dust, smog) are all crucial factors, because the radionuclides are able to form aerosols in the atmosphere and they can be absorbed to the particles of the dust, producing condensation nuclei.

The path of the radioactive substances from the atmosphere to the components of the biological chain can be seen in Figure 10.

In this subchapter the buildup of radioactivity of the atmosphere as a consequence of natural radioactivity as well as of nuclear tests and emission from nuclear power plants will be examined. The climatic and meteorological factors significantly influencing the level of radioactivity will be demonstrated. Of course, the buildup and change of atmospheric radioactivity is in interaction with the level of contamination in the other components of the biosphere, because, as a consequence of the continuous interaction among the atmosphere, hydrosphere and lithosphere, the radioactive isotopes are continuously changing. In addition, the influence of the flora cannot be neglected as regards atmospheric radioactive contamination. For example, plants fix by photosynthesis some of the ^{14}C produced by cosmic radiation as well as by nuclear tests and/or power plants. Of course, the process of dissimilation—and respiration by animals and humans—has the opposite effect.

2.1.1 Natural radioactivity of the atmosphere

The natural radioactivity of the atmosphere is due to substances emitted by the minerals in the Earth's crust and in the hydrosphere, as well as to the radioactive products of the interaction of the atmospheric gases (and the contaminating non-gaseous substances in the air) and cosmic radiation. Knowledge of the concentration level of natural radioactive elements is important in several respects. Knowing the concentration:

- the radiation load on the living organisms can be assessed, thus artificial contamination can be compared and evaluated in relation to the natural level;
- through measurements of the concentration of radon and its derivatives, the state of the atmosphere's equilibrium and turbulent occurrences can be studied, the diffusion in air can be determined;
- some information about the composition of the soil can be gathered, e.g. ore sites containing more than average uranium can be localized;
- the procedures and methods of health physics (e.g. in uranium mines) can be modernized and developed.

The lion's share of natural atmospheric radioactivity is due to the activity of the gaseous derivatives of ^{238}U and ^{235}U . The first β -radiating element of radon gas is RaB ($^{214}_{82}\text{Pb}$) with a half-life of 27 minutes, it produces 56% of the total β -activity of the series. The next daughter element is RaC ($^{214}_{83}\text{Bi}$), with a half-life of 20 minutes, and it makes up 41% of the activity. The percentage of activity of the derivative RaC' ($^{210}_{81}\text{Tl}$) is only 3%.

The first β -radiation-emitting daughter element of the gas thoron is the isotope ThB ($^{212}_{82}\text{Pb}$) with a half-life of 11 hours. The next derivative is ThC ($^{212}_{83}\text{Bi}$), which produces only 9% of the total activity.

The vertical distribution of the atmospheric natural radioactivity, i.e., the magnitude of the radioactive concentration above the soil and its change is determined mainly by thermal exchange. Natural radioactive emanations reach higher altitudes by thermal exchange, thus the higher the elevation, the less the concentration.

On the basis of two measurements made at different altitudes, the diffusion coefficient and even the concentration at ground level can be calculated. The relation is as follows (Junge 1963):

$$C_2 = C_1 e^{-h \sqrt{\frac{\lambda}{D}}}$$

where:

C_2 = concentration measured at higher altitude;

C_1 = concentration measured at lower altitude;

h = the vertical distance between the two points of measurement;

λ = decay constant;

D = diffusion constant.

Using the formula, the C_0 value corresponding to altitude h_0 , that is ground level, can be calculated. The results of measurements in Hungary characteristic of the summer months (from April to September) can be seen in Table 6. The average values at levels of 2.5 m and 10.5 m were measured, and the values corresponding to ground level were calculated. The change in relative concentration of thoron and radon as a function of altitude can be seen in Table 7 (Renoux et al. 1978).

Table 8 gives information about the average radon and thoron content of the air (Renoux et al. 1978). The difference between Paris and Brest is due to the different

Table 6 Average concentration of RaB and ThB in Hungary during the summer months

Altitude (m)	Activity, mBq/m ³	
	RaB	ThB
0.0	1846	130
2.5	1521	118
10.5	818	96

geological settings, because around Brest the Earth's crust contains much more granite with relatively high radioactivity.

In Budapest the yearly average value of the ThB concentration (β -activity) is 170 mBq/m³. The stability of the air-layer near to the ground that is the extent of the vertical exchange which determines the measurable concentration, is the most important meteorological parameter. Up to about +14 °C there is a positive correlation between temperature and the ThB-activity temperature and the ThB-activity of the

Table 7 Change of relative concentration of radon (²²²Rn) and thoron (²²⁰Rn) as a function of altitude measured from the level of the soil

Altitude (m)	Radon (%)	Thoron (%)
Level of soil	100	100
1	95	—
5	—	70
10	87	—
50	—	55
100	69	0.5
1000	38	—
7000	7	—

Table 8 Average radon and thoron content of the air in the surroundings of Paris and Brest

Locality	Activity, mBq/m ³	
	radon	thoron
Paris	7000	56
Brest	15500	126

air, that is, with increasing temperature the exhalation (emission) increases. However, above that temperature the relationship turns negative. It can be explained partly by the fact that the increase of exhalation is overcompensated by the increasing turbulent exchange which transfers the natural radioactive substance to higher layers, thus decreasing the concentration near to the ground. The radon concentration of the air near ground level is lower in the summer months than in winter (Yermolayeva-Makovskaya, Litver 1978).

The amount of radioactive substances being released from the soil into the air is determined even by more factors. On days without precipitation the exhalation of the wet soil is 76%, that of frozen surfaces is 75%, that of surfaces covered by snow is 26% in relation to the exhalation of dry surfaces. On days with precipitation, when the soil surface is wet, the ratio is only 55% (i.e., 72% of the 76% exhalation rate on

dry days), which is due to 'wash out' by the rain water and the material returns to the soil.

Because of the 'wash-out' effect of precipitation, there is a negative linear relationship between the natural radioactivity of the atmosphere (y) and the daily amount of precipitation (x). This relation can be described by the equation

$$y = -ax + b.$$

The natural radioactivity of the atmosphere is about 4800 mBq/m³ above the continents and is lower by two orders of magnitude above the oceans, about 44 mBq/m³. The majority of the radioactivity of natural origin is due to the radon content. Its average concentration above the continents, near to the ground soil (1–2 m) is about 3700 mBq/m³. However, in the closed space of caves the radon concentration may be many thousand times higher than the values measured in the open air, near to the soil.

According to the reports by Kawano and Nakatani (1964), Lockhart (1964), Hoster and Lockhart (1965), Mahalov and Chernisheva (1965), Yermolayeva-Makovskaya and Litver (1978), the radon concentration of the air above continents, has a value between 1100 and 11000 mBq/m³, above the oceans and ice fields a value lower by 1–2 orders of magnitude. For example, in the air above the Antarctic the concentration of radon is only about 70 mBq/m³ (Fontain et al. 1963; Lockhart et al. 1968). In Hungary in open air the average ²²²Rn concentration is around 5 Bq/m³. In winter and autumn the values are significantly higher than in summer and spring. In January the average activity of radon is ~10 Bq/m³, in June only ~2 Bq/m³.

Through the radioactive decay of ²³⁸U or rather radon, several radioactive isotopes are produced in the air (²¹⁸Po, ²¹⁴Pb, ²¹⁴Bi, ²¹⁰Pb, ²¹⁰Po) which are quickly absorbed on aerosol particles present in the air with a diameter of about 100–160 μm. As the number of aerosol particles is an order of magnitude higher than the number of produced radioactive atoms, on one aerosol particle usually only one radionuclide is absorbed. These radionuclides, adhering to the surface of the aerosol particles fall back to the earth either by sedimentation of a gravitational, electric or diffusive nature, or through rain or snowfall.

According to Francis et al. (1970), among the daughter elements yearly $8.1 \cdot 10^7$ Bq/km² ¹⁰Pb and $2.2 \cdot 10^7$ Bq/km² ²¹⁰Po arrives on the surface of Earth from the atmosphere through dry fall-out. The ²¹⁰Pb activity in rainwater and snow is 4–600 mBq/l, its ²¹⁰Po content is 7–50 mBq/l (Yermolayeva-Makovskaya, Litver 1978). The proportion ²¹⁰Po: ²¹⁰Pb in the rainwater and in the air is about 0.2. Most of the ²¹⁰Po and ²¹⁰Pb reaching the soil and later the different components of the biological chain process, arrive by dry fall-out, i.e. by sedimentation of a gravitational nature. The amount of wash-out by precipitation is significantly smaller. The ²¹⁰Pb activity of air above dry land is usually 0.1–1.5 mBq/m³, the ²¹⁰Po activity is 0.03–0.06 mBq/m³ (Jacobi 1965; Peirson et al. 1966).

The ²¹⁰Po: ²¹⁰Pb proportion in the air is a function of the altitude measured from ground level. It is about 0.2 in the lower layers of the air; however, at 10 km altitude it is about 1.0. Higher values of ²¹⁰Pb and ²¹⁰P concentration can be observed in the autumn–winter period, lower values in spring and summer.

According to a communication by Kauranen and Miettinen (1974), yearly about $(2.3-3.7) \cdot 10^{16}$ Bq of ^{210}Pb isotope reaches the atmosphere because of the decay of ^{222}Rn . According to measurements carried out in Helsinki, the specific ^{210}Pb activity of the aerosols was between 0.7 and 90.7 Bq/mg Pb, which indicates that the amount of inactive Pb in the air is extremely high. In 1968 from petrol engines alone—as the petrol contains lead tetraethyl in order to increase the octane number—184 thousand tons of lead was emitted into the environment. In 1971 in the FRG alone the lead emission of vehicles was 6700 tons (Enkelmann, Bayerlander 1973). The amount of lead emission further increased in the seventies, because of the growing number of cars. However, in the last few years the rate of emission has lessened to a certain degree because of the increasing consumption of petrol with low lead content (0.15 g/l) or even unleaded petrol.

It should be mentioned that in the last 30–40 years the ^{210}Pb concentration of our environment has increased significantly. There are basically two reasons. The first is that the consumption of fossil fuels has increased extraordinarily, and through the combustion of coal—as the humic acids of the coal absorb uranium and thorium with a very high enrichment factor ($\approx 10^4$)—uranium and its daughter elements enter the biological circulation through the atmosphere. In the buildup of the α -activity of the air a very important factor is the amount of radioactive substances in the air caused by coal burning (smoke, fly-ash). Thus the Th content of brown coal ash, for example, is usually 12–18 mg/kg, its U content is 10–15 g/kg (Loss 1975). The ^{222}Rn content of natural gas is rather high (about 370 mBq/l), thus through the combustion of natural gas a large amount of radon is passed into the air (UNSCEAR 1977). Another reason is the extensive use of artificial fertilizers, as, mainly through phosphates, a significant amount of uranium and other radioactive isotopes reach the soil and later the air (Pfister et al. 1975).

Some of the natural radioactive substances occurring in the atmosphere are brought about by the interaction between cosmic radiation and the atmospheric gases

Table 9 The more important atmospheric isotopes produced by cosmic radiation

Isotope	Half-life	Rate of atmospheric production atom/cm ³ s	Tropospheric concentration decay/kg min
^3H	12.3 years	0.25	$7 \cdot 10^{-2}$
^7Be	53 days	$8.1 \cdot 10^{-2}$	0.63
^{10}Be	$2.7 \cdot 10^6$ years	$4.5 \cdot 10^{-2}$	$7 \cdot 10^{-8}$
^{14}C	5568 years	1.8	7.6
^{22}Na	2.6 years	$5.6 \cdot 10^{-5}$	$6.7 \cdot 10^{-5}$
^{32}Si	500 years	$1.6 \cdot 10^{-4}$	$1.2 \cdot 10^{-6}$
^{32}P	14.3 days	$8.1 \cdot 10^{-4}$	$1.4 \cdot 10^{-2}$
^{33}P	25 years	$6.8 \cdot 10^{-4}$	$7.6 \cdot 10^{-3}$
^{35}S	88 years	$1.4 \cdot 10^{-3}$	$7.8 \cdot 10^{-3}$

or rather the different pollutant substances in the atmosphere (e.g., dust, volcanic fly-ash). Table 9 lists the most important radioactive isotopes produced in the atmosphere by cosmic radiation, their production rate and their concentration in the troposphere.

In many cases, without knowledge of the circumstances of the sampling procedure, it is impossible to determine whether an examined radioactive isotope was produced by cosmic radiation or can be classified among the radionuclides produced by nuclear explosion. Thus, for example, ^3H can be produced by cosmic radiation; however, the amount of tritium produced by nuclear tests and reaching the biosphere is much more important. Actually, the activity of the radioactive isotopes produced by cosmic radiation in the layers of the atmosphere near to the ground (the value of emission) is usually negligible in comparison with the activity of the derivatives of radon and thoron (Renoux, Madelaine 1973).

It is important to mention that the radioactive isotopes occurring in the atmosphere have an important influence on certain physical parameters of the atmosphere. For example, atmospheric electricity is characterized basically by the intensity of ionization—i.e., the number of produced ions in a unit of space within a unit of time—which is caused in about 80% of cases by the natural α , β and γ radioactivity of the atmosphere (Renoux et al. 1978).

2.1.2 Radioactive contamination of the atmosphere

Contamination of the atmosphere by the emissions of nuclear explosions is influenced not only by the explosive energy of the nuclear weapon tests, given in TNT equivalents, but in a decisive way by the mode of explosion. As to the latter, tests can be classified as stratospheric, atmospheric, surface, underground or underwater. In the case of underground tests, depending on the explosive energy, a cavity of a certain depth is formed by the explosion, thus the produced active substances do not contaminate the biosphere. In the case of stratospheric tests, or if the energy of nuclear explosion performed on the surface of earth or water is greater than 200 kt, the better part of the radioactive products arrive in the stratosphere. This is the reason why the stratosphere is called the world-scale collector of all fission products.

From the atomic cloud formed by the explosion, the radioactive substances fall back to the surface of the Earth as a function of the diameter of their particles. The fall-out can be classified as:

- local,
- tropospheric,
- stratospheric.

The local fall-out takes place in an area of some square kilometres around the epicentre within a few hours. The diameters of the fall-out particles are $>10\ \mu\text{m}$. This fall-out is called early fall-out (within 24 hours), in contrast to the late fall-out (after 24 hours), when the particle dimensions are $<10\ \mu\text{m}$. The local fall-out produces a contamination in the wind direction over an extended area, only within

the immediate environment of the test. The chief danger in the case of early fall-out is γ -radiation.

The tropospheric fall-out, otherwise known as regional fall-out, has particles belonging to the 1–10 μm size range. This fall-out may produce strong contamination over an area of many thousand square kilometres, as a function of the energy of the explosion, as well as that of the meteorological circumstances. The duration of the tropospheric fall-out can be from 1 day to several months. The fall-out is described by Stokes' law of deposition, as follows:

$$v = \frac{2gr^2(p-p_0)}{9\eta}$$

where:

v = speed of the falling object, i.e., the speed of deposition;

g = gravitational acceleration;

r = radius of the falling object;

p = density of the falling object;

p_0 = density of the air;

η = viscosity of the air.

In order to calculate the actual speed of deposition, such parameters as the distribution of the dimensions of the particles and the form factor must be known.

The percentage distribution of the particles falling out from the atomic cloud produced by a nuclear explosion on the surface of the Earth can be seen in Table 10. The altitude of an atomic cloud is determined by the energy of the explosion. In the case of an energy of 1 Mt, the cloud is pushed up higher than 12 km. The isotope proportion of radionuclides falling out in different distances from the explosion site is far from similar. This is because of the different particle size, as well as because of the fractionalization on aerosols caused by the different volatility of different radioactive isotopes (Israel et al. 1968).

In the case of high energy tests on the surface of the Earth or in water, as well as in stratospheric nuclear tests, the greater part of radioactive substances occurring in the atomic cloud reaches the stratosphere. The particles with larger diameter fall back onto

Table 10 Distribution of particles falling out from an atomic cloud during a nuclear test onto the surface of Earth

Particle diameter μm	Percentage distribution of fall-out as a function of the distance from the epicentre of the explosion			
	2 km	15 km	30 km	60 km
< 10	0	0	5	10
10–30	0	15	30	40
30–50	0	20	15	25
50–100	10	45	30	15
100–300	70	30	20	10
> 300	20	0	0	0

the surface of the Earth within a relatively short time. However, the particles with diameter smaller than $1 \mu\text{m}$, may remain in the higher layers of the atmosphere for decades. Brownian motion is already characteristic for these small particles. The fall-out from the stratosphere—the global fall-out—cannot be related directly to the site of the test and may occur practically at any place on Earth as a function of the meteorological circumstances. Thus the gravest danger produced by nuclear weapon tests, threatening the whole of mankind, is fall-out from the stratosphere.

The process of fall-out from the stratosphere to the surface of the Earth can be described well by an exponential mathematical model (Marey 1980). The limiting condition is that the distribution of the radioactive substances in the stratosphere is homogeneous and that the speed of fall-out is constant. In this case the differential equation describing the temporal change of activity of the radioactive substance in the stratosphere is as follows:

$$\frac{dQ(t)}{dt} = -\lambda Q(t),$$

where:

$Q(t)$ = amount of the radioactive isotopes present in the stratosphere at the time t ,
 $\lambda = \lambda_1 + \lambda_2$, where λ_1 is the radioactive decay constant, λ_2 is the speed of fall-out of the radionuclides from the stratosphere.

Solving the equation,

$$Q(t) = Q_0 e^{-\lambda t}$$

The degree of stratospheric fall-out for an optional time interval $(t_2 - t_1)$ is hence as follows:

$$Q(t_1) - Q(t_2) = Q_0 (e^{-\lambda t_1} - e^{-\lambda t_2}).$$

The radioactive isotopes in the stratosphere and the lower layers of atmosphere may gradually arrive on the surface of the Earth in three different ways:

- (a) through gravitational deposition in weather without precipitation (dry-out);
- (b) through rain-out within the clouds;
- (c) through wash-out under the clouds.

Thus the so-called dry-out is the process when the radioactive particles from the stratosphere through the troposphere arrive on the surface of the Earth with a dry fall-out. Wet fall-out includes both the terms rain-out and wash-out; however, the process of rain-out is more important as the radioactivity of the rainwater is produced mainly by the radioactive substances in the cloudy air. (It should be mentioned here that fall-out means the whole deposition, that is the sum of dry and wet depositions.)

In regions where there is much precipitation, the biggest part of the radioactive substances arriving on the surface of the Earth comes from wet fall-out, thus the level of contamination will be determined by the amount of precipitation (Gedeonov 1972). The following relation can be stated:

$$y = c \cdot x$$

where:

y = radioactive pollution of the Earth's surface in Bq/m^2 ;
 x = amount of precipitation in mm.

Thus the dimension of the constant c is $\text{Bq/m}^2 \text{ mm}$; its absolute value is determined by the geographical location.

Of course, the amount of radioactive isotope arriving by precipitation and dry-out at the surface of the Earth is determined to a great extent by geographical situation. For example, in the northern hemisphere the radioactive contamination is 2–3 times higher than the southern hemisphere, and maximum contamination is recorded between 30 and 60 degrees of latitude (Eisenbud 1963; Zier 1967; Malahov et al. 1975). There is a difference even between Europe and North America. For example, in America at the same latitude the contamination due to the nuclear tests is about 50% higher than in Europe.

The yearly maxima of atmospheric radioactive pollution, provided no new sources of contamination occur, can be expected in most regions of Europe from May to June (Gedeonov 1972). The reason for this is that the frequency of meteorological fronts in West and Central Europe is higher in May than in other months. According to Kopcewicz (1962), the increase in radioactivity is produced by radioactive aerosol particles from the highly polluted low layers of the stratosphere reaching the relatively weakly polluted troposphere in the "tropopause gaps" coexisting with the cold fronts. These radioactive substances are brought near to the surface of the Earth by the downward current before the cold fronts, then by wet or dry fall-out reach the surface.

The degree of wash-out of radioactive substances in the troposphere is a function of the amount, or rather of the frequency, of precipitation. Since during the initial phases of rain the air is practically cleaned up, the specific radioactivity of the

Table 11 The values of radioactive fall-out in the region of Bratislava between 1962 and 1966

Year	Total fall-out, Bq/m^2			^{90}Sr : total- β (%)
	Total β -activity	^{90}Sr activity	^{137}Cs activity	
1962	43 670	538	352	1.2
1963	43 570	723	1061	1.7
1964	9 160	667	640	7.3
1965	3 440	229	348	6.7
1966	1 600	104	173	6.5

Table 12 The values of radioactive fall-out in Berlin

Year	Total fall-out, Bq/m^2		^{90}Sr : total- β (%)
	Total β -activity	^{90}Sr activity	
1963	37 070	459	1.2
1964	8 732	237	2.7
1965	1 924	96	5.0
1966	1 110	59	5.3

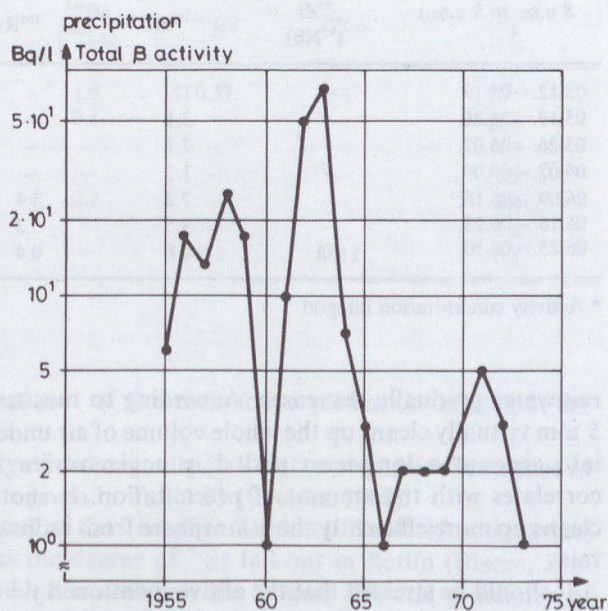
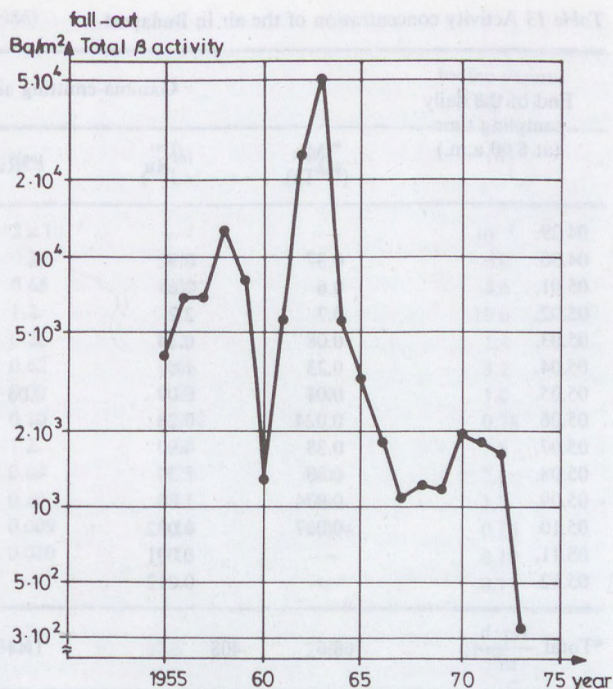


Fig. 11. Total β activity of dry and wet fall-out as well as the average total β radioactivity of the atmospheric precipitation in Budapest

Table 13 Activity concentration of the air in Budapest

End of the daily sampling time (at 8.00 a.m.)	Gamma-emitting aerosols (Bq/m ³)				
	⁹⁹ Mo (^{99m} Tc)	¹⁰³ Ru	¹⁰⁶ Ru	¹³² Te (¹³² I)	¹³¹ I
04.29.	—	—	—	7×10^{-5}	5×10^{-4}
04.30.	0.37	0.92	—	3.8	0.90
05.01.	0.6	1.6	—	6.5	1.9
05.02.	0.7	2.2	—	8.4	4.1
05.03.	0.08	0.34	—	1.1	0.75
05.04.	0.23	1.5	—	3.4	1.6
05.05.	0.01	0.09	0.06	0.22	0.42
05.06.	0.024	0.28	—	0.30	0.44
05.07.	0.38	4.9	—	4.2	3.7
05.08.	0.38	3.3	—	1.9	2.2
05.09.	0.074	1.8	—	0.75	0.75
05.10.	0.017	0.032	—	0.017	0.054
05.11.	—	0.031	—	0.015	0.051
05.12.	—	0.012	—	0.006	0.026
*Total $\frac{\text{Bq} \cdot \text{h}}{\text{m}^3}$	68.6	408	1.44	734.4	405.6

Weekly sampling periods (from 8 a.m. to 8 a.m.)	Gamma-emitting aerosols (mBq/m ³)				
	⁹⁵ Zr (⁹⁵ Nb)	¹⁰³ Ru	¹⁰⁶ Ru	^{129m} Te	¹³¹ I
05.12. — 05.19.	—	12	—	—	7.3
05.19. — 05.26.	—	3.1	—	—	1.6
05.26. — 06.02.	—	2.1	—	—	0.6
06.02. — 06.09.	—	1.3	—	—	0.2
06.09. — 06.16.	—	7.2	3.4	1.6	0.2
06.16. — 06.23.	—	9.5	5.2	—	0.65
06.23. — 06.30.	0.1	0.8	0.4	0.5	—

* Activity concentration integral

rainwater gradually decreases. According to measurements, a rainfall greater than 5 mm virtually cleans up the whole volume of air under the cloud. Of course if we take into account a long-term period in a given area, the frequency of precipitation correlates with the amount of precipitation. It should be mentioned that snowfall cleans up more efficiently the atmosphere from radioactive aerosol particles than does rain.

It should be stressed that the above-mentioned $y = c \cdot x$ relationship is not valid for dry, desert-like areas (Lavrenchik 1965; Shemi-zade 1968). For example, in the region of Tashkent, $74 \pm 20\%$ of the total fall-out consisted of dry-out. In regions where

						Iodine vapour (Bq/m ³)
¹³⁴ Cs	¹³⁶ Cs	¹³⁷ Cs	¹⁴⁰ Ba (¹⁴⁰ La)	¹⁴¹ Ce	¹³¹ I	
—	—	2 × 10 ⁻⁵	—	—	10 ⁻²	
0.16	—	0.30	0.10	—	4.0	
0.3	—	0.65	0.30	—	4.6	
0.8	—	1.4	—	—	10.0	
0.13	—	0.26	0.21	—	1.4	
0.34	—	0.62	0.66	—	3.2	
0.05	0.02	0.09	0.08	0.03	1.0	
0.056	0.02	0.10	0.055	—	0.78	
0.58	0.14	1.2	0.35	—	3.8	
0.33	0.09	0.69	0.27	0.03	7.4	
0.14	0.034	0.29	0.11	0.02	1.7	
0.006	—	0.009	0.11	0.004	0.16	
0.006	—	0.010	0.13	—	0.10	
—	—	—	—	—	0.11	
69.6	7.2	134.4	57.6	2.0	919.2	
						Iodine vapour (mBq/m ³)
¹³² Te (¹³² I)	¹³⁴ Cs	¹³⁷ Cs	¹⁴⁰ Ba (¹⁴⁰ La)	¹⁴¹ Ce	¹⁴⁴ Ce	¹³¹ I
12	1.5	3.5	1.9	0.57	—	36
0.1	0.8	1.5	0.8	—	—	12.4
—	0.5	1.1	—	—	—	4.7
—	0.3	0.7	—	—	—	0.8
—	0.2	0.4	0.2	—	—	1.4
—	0.24	0.5	—	—	—	2.6
—	0.3	0.6	—	0.1	0.15	0.1

precipitation is scarce, the largest part of the radioactive substances arrives at the surface by dry fall-out.

Information about the quantity of radioactivity (fall-out) arriving at the surface of the Earth from the atmosphere is given in Table 11 (Csupka 1969).

In order to facilitate comparison of data listed in Table 11, Table 12 summarizes the amount of fall-out as well as the degree of ⁹⁰Sr fall-out in Berlin (Blume, Zinder 1967). In addition, Figure 11 demonstrates the degree of total β-activity of rainwater and dry as well as wet deposition measured in Budapest. It can be seen that the maximum value was measured in 1963 and the data registered since 1965—

including the present-day contamination—demonstrate radioactivity smaller by an order of magnitude. However, due to a nuclear test of high energy or reactor disaster, the fall-out, the radioactivity of rainwater and aerosol particles, may increase significantly even if temporarily. For example, in Hungary the activity of the aerosol particles floating in the atmosphere in solid state usually has a value around 2–10 mBq/m³. However, in October 1976, as a consequence of the nuclear test performed on 26 September 1976 in China, a maximum value of 78 mBq/m³ was registered.

It is well known that the accident at the Chernobyl nuclear reactor in the USSR, which occurred on April 26 1986, caused extensive contamination locally and widely dispersed and deposited radioactive material in European countries and throughout the northern hemisphere. In almost all countries intensive monitoring was undertaken to evaluate the radiation levels. The average monthly airborne dust activity measurements since 1957 and the daily values at Mol (Belgium) are summarized in Figures 12 and 13 (Deworm 1987).

The different radioisotopes, ¹³²Te (¹³²I), ¹³¹I, ¹³⁷Cs, emitted in Chernobyl were first detected in Budapest in aerosol samples collected in the morning of April 29 1986. Figure 14 shows the time dependence of ¹³²Te, ¹⁰³Ru, and ¹³⁷Cs aerosol concentrations in Budapest, and the concentrations of the aerosols and ¹³¹I vapour as a function of time are given in Table 13 (Bíró et al. 1986). On examining the ¹³¹I distribution between the aerosol and vapour phase for the total time period, 31% and 69% of the time integral of the activity concentration can be related to the former and latter, respectively (Andrási et al. 1986) (see Figure 15). The external gamma dose rate and the time integral of ¹³¹I activity concentration in the air are given in Figure 16 (Láng 1986). These measurements were carried out at the Central Research Institute for Physics, Budapest.

After a nuclear test, within a relatively short period (not more than a few months), the majority of the fall-out activity is made up of isotopes of ¹⁴⁰Ba + ¹⁴⁰La, ¹³¹I, ¹⁴⁴Ce + ¹⁴⁴Pr, ¹⁰⁶Ru + ¹⁰⁶Rh, ¹⁴¹Ce, ⁹⁵Zr + ⁹⁵Nb, ⁹¹Y and ⁸⁹Sr. As can be seen in Table 14, as well as in Figures 7, 8 and 9, the proportion of activity of the isotopes ⁹⁰Sr + ⁹⁰Y and ¹³⁷Cs + ¹³⁷Ba, of primary importance in the buildup of long-term radioactive contamination of the atmosphere as well as the biosphere, becomes important only one year after the nuclear weapon test. After 10 years about 80%, and after 20 years about 90% of the contamination is caused by the activity of the above-mentioned isotopes.

As we have already said, the radioactive particles reach either the troposphere or the stratosphere, depending on the mode of explosion, or rather on its energy. The deposition in the air is determined by the starting layer-altitude of the fall-out. Knowing the aerosol concentration (*c*) and the rate of the fall-out (*D*), the speed of tropospheric deposition (*v*) can be calculated by the following empirical relationship

$$v = \frac{D}{c}$$

where:

D = degree of fall-out, Bq/m² · day;

c = aerosol concentration, mBq/m³.

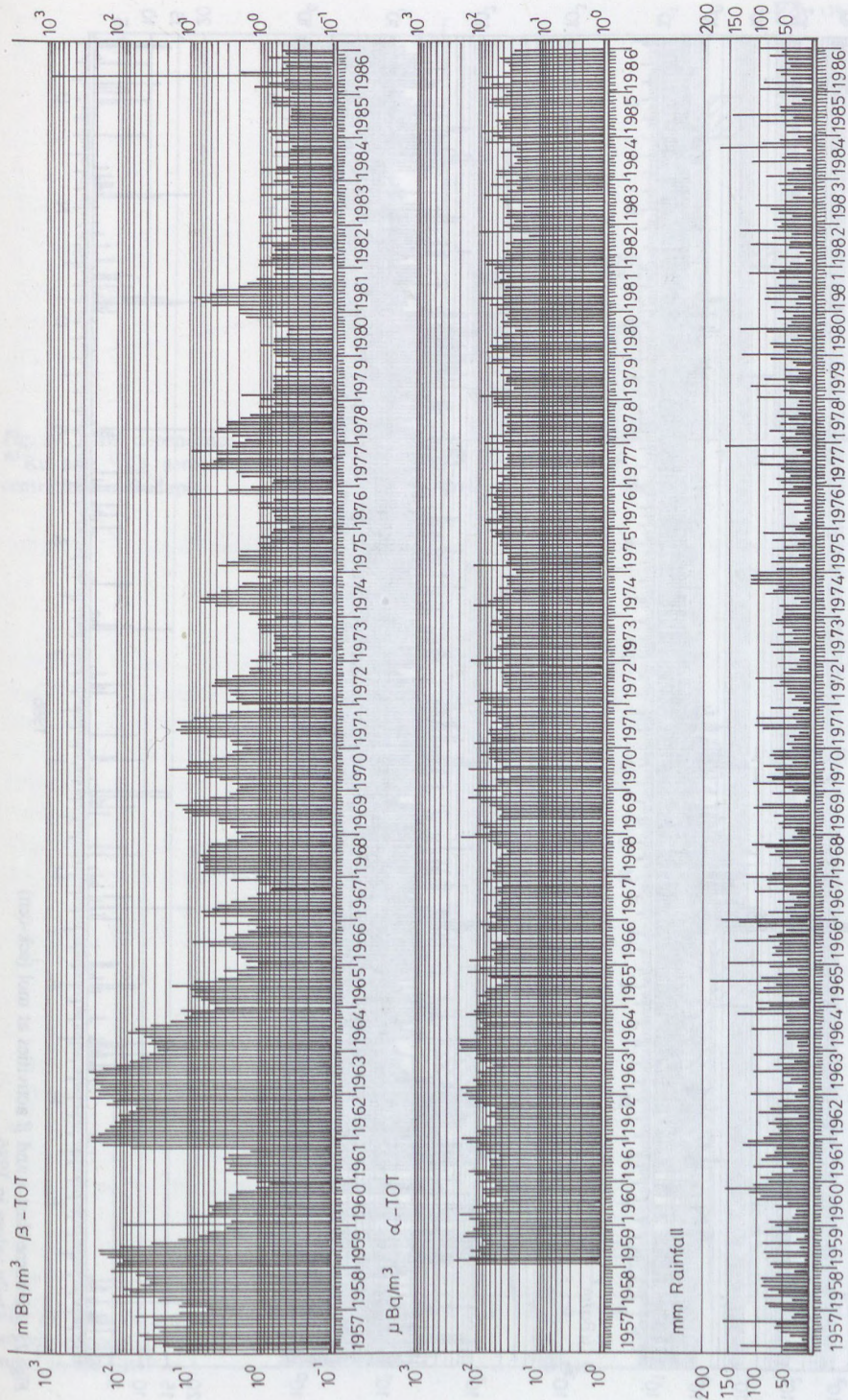


Fig. 12. Airborne dust α and β activities at mol (sck-cen)
Monthly mean values from 1957 to 1986

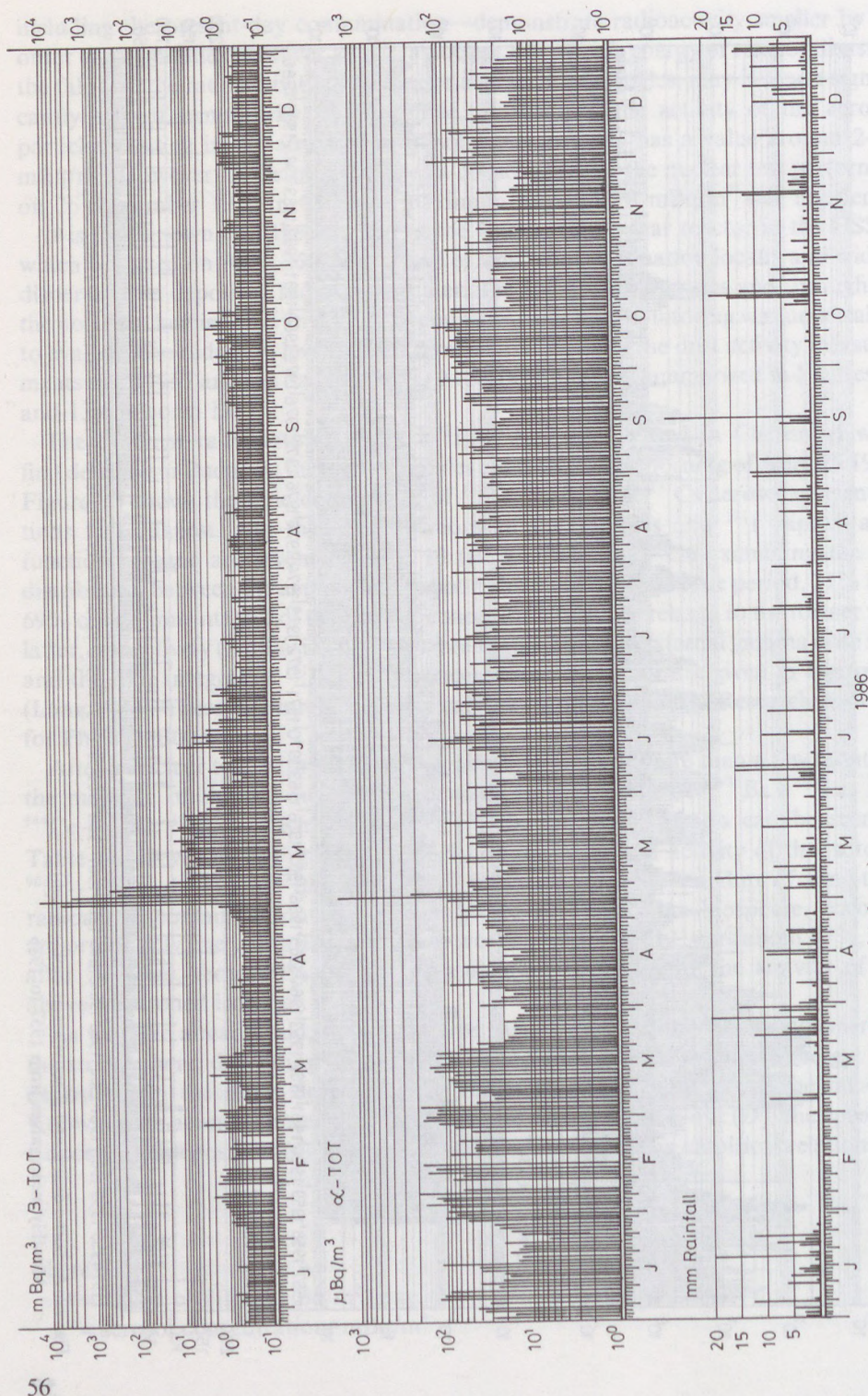


Fig. 13. Airborne dust α and β activities at mol (sck-cen)
Daily values in 1986

Fig. 14. Time dependence of ^{132}Te (^{132}I) ^{103}Ru and ^{137}Cs aerosol activity concentrations in Budapest

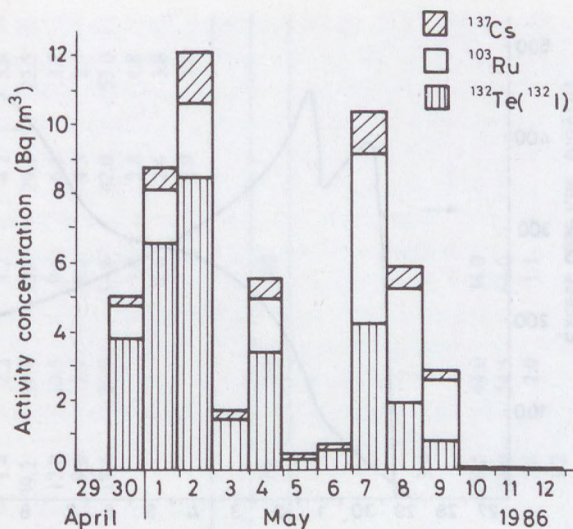
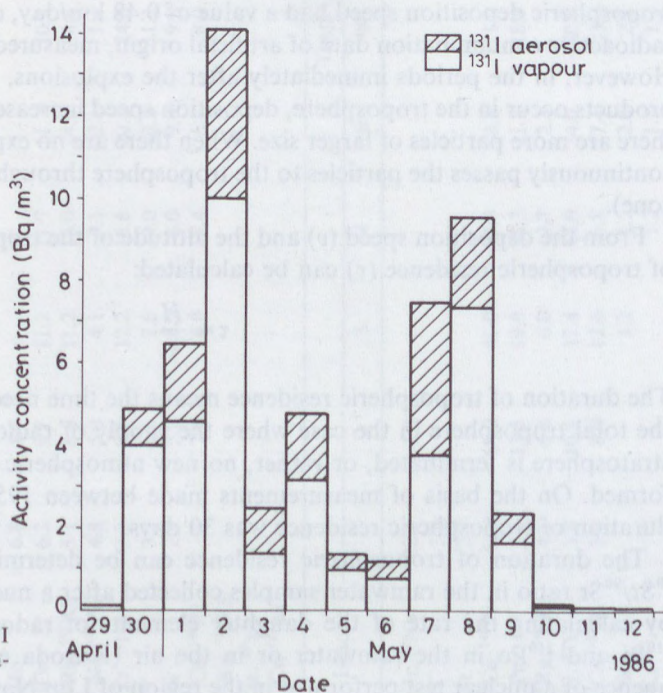


Fig. 15. Time dependence of ^{131}I activity concentration in air measured in the CRIP



on March 18, 1972, in the region of Fayetteville in the United States (36° N, 94° W) the rate of $^{89}\text{Sr}/^{90}\text{Sr}$ in the rainwater samples between April and July 1972 was higher than 10. However, in October and November the registered value was around 1.0 and in May 1973 the rate of $^{89}\text{Sr}/^{90}\text{Sr}$ activity was about 0.2. It can be demonstrated that ^{89}Sr had almost completely decayed after a period of little more than one year after the appearance of the pollution.

A relatively significant amount of radioactive isotope may also be released into the air as a consequence of the normal, disaster-free operation of nuclear power plants (Martin et al. 1974; Dibobes, Teverovsky 1975). However, the radionuclides reaching the troposphere in this way cause in most cases "only" local contamination, so their influence is negligible compared to the radiation load of areas further than 10–20 km distant. The air polluting emission figures of a research reactor with a power of 16 MW_t using light water both as moderator and coolant, working in the USSR, is given in Table 15 (Babanina et al. 1975).

Table 15 Air polluting radioactive emission of a research nuclear reactor with a power of 16 MW_t in Bq/h units

Radioactive emission, Bq/h			
gases		aerosol	
^{41}Ar	$(3.3 \pm 0.1) \cdot 10^{10}$	^{24}Na	$(8.1 \pm 0.7) \cdot 10^4$
$^{85\text{m}}\text{Kr}$	$(3.5 \pm 0.1) \cdot 10^8$	^{51}Cr	$(7.0 \pm 0.4) \cdot 10^2$
^{87}Kr	$(6.9 \pm 0.3) \cdot 10^8$	^{56}Mn	$(3.7 \pm 1.1) \cdot 10^4$
^{88}Kr	$(6.4 \pm 0.3) \cdot 10^8$	^{60}Co	$(3.5 \pm 0.2) \cdot 10^2$
^{133}Xe	$(5.6 \pm 1.9) \cdot 10^8$	^{65}Zn	$(6.7 \pm 1.9) \cdot 10^2$
^{135}Xe	$(7.4 \pm 0.4) \cdot 10^8$	^{89}Rb	$(8.9 \pm 4.8) \cdot 10^7$
^{138}Xe	$(1.4 \pm 0.1) \cdot 10^9$	^{89}Sr	$(5.2 \pm 0.4) \cdot 10^4$
		^{90}Sr	$(4.8 \pm 3.3) \cdot 10^2$
		^{91}Sr	$(7.0 \pm 0.7) \cdot 10^4$
		^{92}Sr	$(4.4 \pm 0.7) \cdot 10^4$
		^{99}Mo	$(5.6 \pm 0.7) \cdot 10^2$
		^{131}I	$(2.9 \pm 0.5) \cdot 10^3$
		^{132}I	$(1.7 \pm 0.6) \cdot 10^3$
		^{133}I	$(8.1 \pm 3.0) \cdot 10^3$
		^{137}Cs	$(1.5 \pm 0.1) \cdot 10^2$
		^{138}Cs	$(1.1 \pm 0.6) \cdot 10^8$
		^{139}Ba	$(7.0 \pm 0.4) \cdot 10^5$
		^{140}Ba	$(3.7 \pm 0.7) \cdot 10^3$
		^{140}La	$(2.5 \pm 0.3) \cdot 10^3$

Due to the Chernobyl disaster much more radioactive product was released into the atmosphere than by the normal operation of nuclear power plants. The daily emission and the radioisotopic composition of the releases are shown in Tables 16 and 17 (Abagyan et al. 1986).

A very important air polluting radioactive isotope of the nuclear industry must be mentioned: ^{85}Kr with a half-life of 10.8 years. With the fission of heavy nuclei, 36

kinds of noble gas isotopes are produced; however, all these elements, with the exception of ^{85}Kr which causes durable radiation pollution, are stable or have a short half-life. The main source of ^{85}Kr is not the nuclear tests but the reprocessing of burned-out fuel elements produced in reactors working for a long time. So in 1973 in the atmosphere of the Earth—which has a mass of about $5.3 \cdot 10^{21}$ g—there was about $1.96 \cdot 10^{18}$ Bq = 136 kg ^{85}Kr . Only about 2% of this amount was due to the

Table 16 Daily release, q, of radioactive substances to the atmosphere from the damaged unit (excluding radioactive inert gases)*. (Abagyan et al. 1986)

Date	Days after the accident	q, MCi**
26.04.	0	12
27.04.	1	4.0
28.04.	2	3.4
29.04.	3	2.6
30.04.	4	2.0
01.05.	5	2.0
02.05.	6	4.0
03.05.	7	5.0
04.05.	8	7.0
05.05.	9	8.0
06.05.	10	0.1
09.05.	14	~0.01
23.05.	28	$20 \cdot 10^{-6}$

* Release evaluation error $\pm 50\%$. It is composed of the error the dosimetric instruments, of the radiometric measurements of the radioisotopic composition of air and soil samples and of the error due to averaging the fall-out over the area.

** The values of q were calculated on 6 May 1986 taking into account radioactive decay. (At the time of the release on 26 April 1986, the activity was 20–22 MCi.) For the composition of the release, see Table 17.

nuclear tests, the majority was caused by the regeneration of fuel elements or, rather, by plutonium processing (Tertishnik et al. 1977). The development of the nuclear industry and the increase of ^{85}Kr emission is demonstrated by the fact that, for example, in Nevada State, USA, in 1970 the average ^{85}Kr content of the air was already higher than 440 mBq/m^3 .

The ^{85}Kr content of the atmosphere has increased significantly over recent years and decades. According to measurements carried out in Hungary, ^{85}Kr activity in 1954 was on the average only 27 mBq/m^3 ; in 1966 it was already 426 mBq/m^3 .

Table 17 Assessment of the radioisotopic composition of the release from the damaged unit*

Isotope**	Activity of release, MCi		Amount of activity released from the reactor by 06.05.86, %
	26.04.86.	06.05.86.***	
¹³³ Xe	5	45	Possibly up to 100
^{85m} Kr	0.15	—	Possibly up to 100
⁸⁵ Kr	—	0.9	Possibly up to 100
¹³¹ I	4.5	7.3	20
¹³² Te	4	1.3	15
¹³⁴ Cs	0.15	0.5	10
¹³⁷ Cs	0.3	1.0	13
⁹⁹ Mo	0.45	3.0	2.3
⁹⁵ Zr	0.45	3.8	3.2
¹⁰³ Ru	0.6	3.2	2.9
¹⁰⁶ Ru	0.2	1.6	2.9
¹⁴⁰ Ba	0.5	4.3	5.6
¹⁴¹ Ce	0.4	2.8	2.3
¹⁴⁴ Ce	0.45	2.4	2.8
⁸⁹ Sr	0.25	2.2	4.0
⁹⁰ Sr	0.015	0.22	4.0
²³⁸ Pu	$0.1 \cdot 10^{-3}$	$0.8 \cdot 10^{-3}$	3.0
²³⁹ Pu	$0.1 \cdot 10^{-3}$	$0.7 \cdot 10^{-3}$	3.0
²⁴⁰ Pu	$0.2 \cdot 10^{-3}$	$1 \cdot 10^{-3}$	3.0
²⁴¹ Pu	0.02	0.14	3.0
²⁴² Pu	$0.3 \cdot 10^{-6}$	$2 \cdot 10^{-6}$	3.0
²⁴² Cm	$0.3 \cdot 10^{-2}$	$2.1 \cdot 10^{-2}$	3.0
²³⁹ Np	2.7	1.2	3.2

* Evaluation error $\pm 50\%$. For explanation, see footnote to Table 16.

** Data on the activity of the main radioisotopes measured in the radiometric analysis.

*** Total release by 6 May 1986.

According to the communication of Tertishnik and Malahov (1974) and Tertishnik et al. (1977) in the USSR the average ⁸⁵Kr activity of the air was in 1971 503, in 1972 522, in 1973 540, in 1974 again 540 and in 1975 599 mBq/m³. Currently the ⁸⁵Kr activity of the atmosphere has a value about 1000 mBq/m³.

The considerable accumulation of ⁸⁵Kr in the atmosphere—97% of the total amount is in the air as krypton dissolves badly in water, and being an inert gas, does not form chemical compounds—can be dangerous not only because of the radiation load. The increase in the amount of ⁸⁵Kr may have an influence on the climate. The electricity of air—its degree of ionization—plays an important role in the formation of precipitation, and because of the growing ⁸⁵Kr concentration the conductivity of the air layers increases between the ionosphere and the surface of sea, possibly causing very substantial consequences (hardly foreseen in advance) in atmospheric meteorological phenomena. Of course, we must add that by the application of special

technologies the amount of ^{85}Kr getting to the atmosphere can also be significantly reduced (Nahutin et al. 1976; Ageykov, Valuyev 1976).

Let us now see some data about the radioactive contamination of the air. The amount of ^3H usually produced by cosmic radiation has been substantially increased by nuclear tests. Between 1962 and 1964, the rate of contamination being the maximum, the ^3H pollution of the atmosphere was higher by 2–3 orders of magnitude than the natural background. Since then the ^3H concentration has decreased significantly. Ninety-nine per cent of the ^3H content of the biosphere is contained in seawater and it occurs in the atmosphere as water vapour. In 1970 in Nevada the ^3H content of the air was between 5550 and 10 730 mBq/m^3 (RADIAT DATA 1974).

According to γ -spectrometric measurements of Vinogradov et al. (1974), in 1973 above the Atlantic the average concentration of ^7Be , ^{144}Ce and ^{137}Cs isotopes was 370, 22 and 2 mBq/m^3 , respectively. In 1974, in the USA, the average β -activity of the air was 5 mBq/m^3 , the average activity of ^{95}Zr , ^{106}Ru , ^{144}Ce and ^{140}Ba determined by γ -spectrometry was 34, 35, 16 and 17 mBq/m^3 , respectively (*Air and deposition* 1974). At the same time in Canada the average β -activity of the atmosphere was 9 mBq/m^3 .

During the fission of uranium the total production of ^{90}Sr (5.77%) and ^{137}Cs (6.15%) is nearly the same and their half-lives are also similar, thus the activity of ^{90}Sr and ^{137}Cs in the atmosphere is usually at the same level. According to a communication of Sherrill et al. (1975), the measurable $^{137}\text{Cs} : ^{90}\text{Sr}$ ratio in the atmosphere above the USA and in rainwater had a value between 0.75 and 1.9. When the radioactive contamination of the air is of stratospheric origin, i.e. it is not caused by new fission products, the majority of the radioactivity is due to the presence of ^{90}Sr and ^{137}Cs nuclides.

The amount of air-polluting plutonium isotopes (^{238}Pu , ^{239}Pu , ^{240}Pu , ^{241}Pu) produced by nuclear tests (as well as by the annihilation of artificial satellites in the high atmosphere containing plutonium source) is not negligible. Among these isotopes the ^{239}Pu and ^{240}Pu nuclides are important. As the largest part of 239 , ^{240}Pu contaminated the atmosphere, or rather the stratosphere, before 1963, most of it has already arrived on the surface of the earth by fall-out (Harley 1971). Before 1973 $1.2 \cdot 10^{16}$ Bq of ^{239}Pu arrived on the ground surface, and because at present the amount of ^{239}Pu in the stratosphere is two orders of magnitude smaller, the ^{239}Pu -pollution of the atmosphere is at low level (Wrenn 1974; Marey 1980).

Even the amount of different iodine isotopes leaking out from different nuclear facilities into the air may be significant. Some of the iodine and its compounds are volatile so the radioactive isotopes during the operation of nuclear power plants may leak from the fuel units through microfissures into the primary water-cycle, then into the air. Also during the production of isotopes radioactive iodine may be released into the atmosphere through the ventilation system. The yearly iodine emission of the Isotope Research Institute of the Hungarian Academy of Sciences is about $1.5 \cdot 10^{10}$ Bq, the projected ^{131}I emission of the Paks Nuclear Power Plant (Hungary) is $2.8 \cdot 10^{10}$ Bq, in the case of a possible breakdown $4.4 \cdot 10^{13}$ Bq.

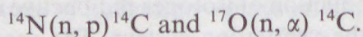
As we have already mentioned, the ^{14}C isotope, which plays a very important role in the radioactivity of the atmosphere and the living organisms, is, on the one hand, produced by cosmic radiation in a natural way; however, on the other hand, its presence is due to the nuclear weapon tests as well as to the contamination produced by nuclear power plants. Under the influence of cosmic radiation or atmospheric nuclear tests, in the ^{14}N (n, p) ^{14}C process the ^{14}C produced is oxidized to $^{14}\text{CO}_2$ gas and together with the other CO_2 molecules it participates in the carbon cycle of the Earth.

As the half-life of ^{14}C is rather short compared to the geological time scale, the cosmogenic ^{14}C is in a radioactive equilibrium state on Earth. The equilibrium proportion of the isotopes is $^{14}\text{C}/^{12}\text{C} = 1.17 \times 10^{-12}$. The amount of ^{14}C produced naturally on our planet is 82.7×10^8 GBq (5.1×10^4 kg ^{14}C), which is distributed in the hydrosphere, atmosphere and flora as 94.3%, 1.9% and 3.8%, respectively (Fairhall, Young 1970). Thus in the atmosphere ^{14}C of natural origin is only 1.57×10^8 GBq and the amount of ^{14}C produced yearly is only about 6.1 kg, i.e. about 10^6 GBq.

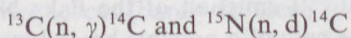
Incidentally the ^{14}C content of the atmosphere has not remained constant. During the last few thousand years the variation—about 5–10%—has been due to changes of the magnetic field (dipole momentum) around the Earth. However, since the beginning of the 20th century the $^{14}\text{C}/^{12}\text{C}$ proportion has been decreasing—this is the so-called Suess effect—which is due to the large-scale consumption of fossil fuels. Burning these fuels (coal, petrol), CO_2 virtually without ^{14}C is emitted into the atmosphere. This is the reason why in 1962 an international agreement was reached which prescribed that in age determination the reference level of ^{14}C -concentration should be wood samples taken between 1870 and 1890.

However, this effect—the decrease of the $^{14}\text{C}/^{12}\text{C}$ proportion—is overcompensated by the ^{14}C isotope production of nuclear tests. Thus in 1963—in the year of the Moscow Treaty on Banning Nuclear Tests—the ^{14}C concentration of the atmosphere was double that of the natural level. As the majority of nuclear tests were carried out in the northern hemisphere, in the 1960s the ^{14}C concentration was somewhat lower in the southern hemisphere than in the northern one. Since 1963 the ^{14}C contamination has slowly decreased, and since 1970 the level of ^{14}C concentration has been practically the same all over the world. The successive decrease in concentration is due in the first place to the exchange processes among the atmosphere and the other components of the biosphere, and in a significant part to absorption in seawater.

Incidentally, the radioactive contamination of nuclear power plants and reprocessing plants also has an influence on the buildup of the ^{14}C level. In nuclear power plants ^{14}C is produced first of all in the following nuclear reactions:



The nuclear reactions



are of secondary importance because of their very small cross-section.

The ^{14}C is mostly produced in the cooling water emitted into the environment of the nuclear power plants. The ^{14}C produced within the fuel units can be released during reprocessing. In the case of a reactor with pressurized water, ^{14}C occurs mostly in the form of hydrocarbons, in the case of the other reactor types and in the reprocessing plants in the form of CO_2 present in the effluent gas. Actually, at present the amount of ^{14}C produced by the nuclear industry is only a few per cent of the amount of ^{14}C produced naturally. In order to present a true picture, we have to mention that the amount of ^{14}C produced and emitted by a nuclear power plant of 1 GW capacity is about 260 GBq/year, and through the reprocessing of fuel rods another 470 GBq/year (Davis 1979). Otherwise there are reliable methods to retain and absorb the ^{14}C .

As we have mentioned, currently the tropospheric ^{14}C concentration shows a slow downward trend, in 1980 it exceeded the natural ^{14}C background level by about 27%. The average residence time of ^{14}C in the troposphere is about 9–10 years. Thus if in the future there are no further atmospheric nuclear tests, it can be expected that before the year 2000 90% of ^{14}C produced by nuclear tests will be deposited to the deeper layers of the oceans, so there will be no contamination any more (Devirc et al. 1982).

Several methods have been devised to describe the migration and the distribution of the radioactive materials emitted by nuclear power plants in the atmosphere. For instance, Iványi and Mersich (1981) developed such a model, based on the turbulent diffusion equation solved by numeric methods. The meteorological data needed for the calculation were available as the solution of a one-dimensional planetary border layer model. The authors demonstrated their model for the normal operation of nuclear power plants, as well as for a breakdown. The amount of emitted radioactive substances, the half-lives of the radioactive isotopes, the wash-out factor and the change of deposition speed are calculated based on the changing meteorological situation. The results may be very useful when decisions have to be made in a possible emergency situation (reactor disaster).

2.1.3 Summary

This subchapter has dealt with the natural and artificial buildup of the radioactivity in the atmosphere. Data on the levels of natural radioactivity, as well as on the radioactive pollution have been presented. The factors influencing the radioactivity of the atmosphere, the interaction between the atmosphere and the other components of the biosphere have been analysed. The most important statements are as follows:

1. The radioactive substances of natural origin occurring in the atmosphere are either produced by cosmic radiation (e.g. ^3H , ^7Be , ^{14}C), or they come from the soil or from the hydrosphere by thermal exchange (e.g. ^{222}Rn). Most of the natural radioactivity is due to radon. As the natural radioactive emanations are brought to the higher air layers mainly by thermal diffusion, the concentration of radon and thoron decreases as a function of altitude. The values measured near to soil level are significantly higher in autumn and winter than in spring and summer.

2. Most of the radioactive substances of artificial origin occurring in the atmosphere are the products of nuclear weapon tests. Within a relatively short time after the nuclear test (some weeks, perhaps months), the greater part of the atmospheric contamination is formed by the activity of isotopes with short half-lives (e.g., ^{140}Ba , ^{131}I , ^{106}Ru , ^{95}Zr , ^{89}Sr). However, later the activity of ^{90}Sr and the ^{137}Cs isotopes become dominant. The isotope ^{14}C should also be mentioned here. In 1963, when the contamination level reached its maximum, its level of concentration reached about 90% of the natural level. Later the ^{14}C contamination of the air decreased significantly, chiefly because of its absorption by seawater; thus in 1980 its level exceeded the natural level only by 27%.

3. The other sources of contamination include the various nuclear facilities; however, they usually cause only local contamination. An exception is the ^{85}Kr isotope. Its level of concentration is continually growing, which is due without doubt to the reprocessing of the burned-out fuel rods of nuclear power plants, or rather, to the ^{85}Kr emission into the atmosphere during these processes.

4. In May and June 1986, in many European countries the radioactive pollution of the atmosphere strongly increased as a consequence of the catastrophe in Chernobyl on April 26 1986. The damaged reactor was completely sealed on May 6 1986, thus no further radioactive substances are anticipated to be released into the environment. The atmosphere, mostly because of rainfall, was gradually cleaned and since July 1986 the contamination measured has been at levels similar to those of the earlier years.

5. The radioactive substances in the stratosphere may arrive on the ground surface in 3 different ways:

- (a) by gravitational deposition in dry weather (dry-out);
- (b) by rain-out occurring within the clouds;
- (c) by wash-out occurring under the clouds.

Thus the term dry-out means dry fall-out; rain-out and wash-out are the different kinds of wet fall-out. (The term fall-out stands for the total fall-out, the sum of the dry and wet fall-outs.) In places with much precipitation, most of the radioactive substances reaching the surface of the Earth from the atmosphere arrive by wet fall-out, thus the level of contamination is determined mostly by the amount or rather the frequency of the precipitation.

2.2 RADIOACTIVITY OF SURFACE WATERS AND INFLUENCING FACTORS

Radioactive substances in the hydrosphere, as in the atmosphere, can be classified into two groups: namely natural and artificial radionuclides. Among the natural radioactive substances we find the radioactive isotopes as well as their radioactive daughter elements being dissolved from the soil and rocks and thus transferred to the hydrosphere. Another group is the radionuclides produced mainly in the higher strata of the atmosphere by cosmic radiation and from there entering into the water. The

radioactive isotopes emitted by nuclear weapon tests as well as by the nuclear industry (reactors, isotope production) are classified into the group of artificial radioactive substances.

About 70% of the surface of the Earth is covered by water, thus most of the radioactive substances emitted into the air or produced there return through dry-out, rain-out or wash-out directly into the surface waters. However, an important portion of the active substances coming to the soil or to plants passes through the hydrosphere, and is leached out from the soil—possibly also by the way of the chain process of plant–animal–man–sewage water or partially following this route—and arrives in the rivers, lakes or in the sea. There they may accumulate, or rather may be assimilated into aquatic plants or animals and may leave the hydrosphere. Through the nutrition chain process the cycle of the biological circulation of the elements may start all over again.

The study of aquatic organisms is the task of hydrobiology. There are different elements to hydrobiology; water may be on the one hand flowing or stagnant, on the other hand fresh or saline. Oceanography is the study of the seas and oceans, limnology is the science of the lakes and rheology is the science of flowing waters, the rivers. The radioactivity of aquatic ecosystems is studied within a special field of hydrobiology, called hydro-radioecology.

Radioactive substances are in continuous movement, migration and exchange between the hydrosphere and atmosphere, as well as between the hydrosphere and the other elements of the biosphere. The radioactivity of the surface waters and the organisms within it is influenced in addition to the primary causes, by some other factors as well, for example, the volume, the depth of the recipient and the chemical composition of the water. The quality and quantity of the organisms living in the water, even the presence of a fishing industry in the region examined (for example, river, fish-pond) have significance. Namely, in the last case the amount of radioactive substances going from the hydrosphere to the other components of the biosphere in another way (thermal exchange, drinking water, irrigation water) may increase significantly. It is well known that fish accumulate very well different radioactive substances—just as they do some other toxic substances, for instance, heavy metals—in their bodies (Kulikov, Kulikova 1977). In addition, certain algae are able to absorb considerable quantities of some radionuclides from the water.

It should be mentioned that a certain amount of radioactive substance may be released into living waters because of the polluting effect of underground nuclear tests. Through migration during substance transport within the soil, mainly under the influence of groundwaters, a small amount of the radionuclides produced may come up to the surface and may cause contamination. However, the increase of radiation pollution occurring in this way is usually negligible compared with the natural background level of radiation.

The radioactivity of aquatic ecosystems, or rather its change with time, greatly depends on whether a sea, a lake or a river is in question. In the case of rivers, the mixture and transport of polluting substances are mostly determined by stream structure. The temporal and spatial changes of concentration are essentially the

function of

$$c=f(x, y, z, t)$$

where:

c is the concentration of the polluting component;

x, y, z are coordinates of space;

t represents time.

c depends on the nature of the source of pollution and the temporal changes of the water output and is described by the equations of motion of liquids and turbulent diffusion.

In the case of solid state pollutants, an important factor is the sedimentation rate of the particles, which is a function of the dimensions of the particles, as well as of the density difference. Through sedimentation the radioactive substances may settle into the ground sediment, the bottom mud. However, through mechanical (e.g. flood, dredging) and chemical processes (e.g. changes of chemical composition and pH caused by anaerobic chemical reactions within the bottom mud, chemical interaction because of the appearance of new polluting components) they may be again resolved and return to the hydrosphere.

In shallow lakes the mixing caused by wind also plays a significant role in influencing the quality and composition of the water, as well as the contamination level of the different radioactive and other substances. The water motion stirs up a large amount of sediment from the bottom of the lake and in addition to the different mineral substances, for example, even bottom-settled algae, may return to higher water layers where they have enough light for photosynthesis. Thus the chemical composition of shallow lakes changes much more quickly and to a greater extent than that of the sea.

Complex sedimentary, physico-chemical and biological processes take place between the chemical and radioactive substances dissolved in water and the substances within settled sediment. The organic compounds and the dispersed particles adsorb different radioactive substances on their large surface, thus through sedimentation the specific activity of the water may decrease significantly. Therefore in addition to the causative sources (the amount of radioactive substances released into the water) the radioactivity of the surface waters is influenced, not negligibly, by the adsorption of radioactive isotopes and other factors determining the sedimentation rate (e.g., the input of sewage water).

In this subchapter the natural radioactivity of the hydrosphere as well as the process of the buildup of radioactive contamination will be discussed, with reference to the factors which can influence the level of radioactivity. Finally, the radioactivity of the hydrosphere is not independent from the influences of the geochemical, biochemical, aerophysical, biological and other processes in the environmental compartments of the Earth, for the actual level of radioactivity is formed by these complex ecological interactions.

2.2.1 Natural radioactivity of the hydrosphere

The natural radioactivity of the hydrosphere is formed first and foremost by its continuous interaction with the lithosphere and the atmosphere (e.g. by dissolving radioactive substances from the lithosphere, or rather from the soil, and by the arrival of the radionuclides produced by cosmic radiation on the surface waters by wet or dry fall-out). An influencing, modifying factor is the presence of living aquatic organisms (algae, seaweeds, fish) as they build up their body partly from the mineral substances occurring in water, thus they also incorporate the radioactive isotopes. Of course the process is reversible, i.e., after their death with the decomposition of the organic material (biomass) the incorporated radioactive materials are returned to the water.

The radioactivity of a water, and the organisms living in it, is determined to a great extent by the fact of whether it is a flowing or stationary water. In the case of stagnant water, its depth is very important. The level of radioactivity measured in lakes is usually higher than the activity in flowing waters, because a significant part of the radioactive substances brought by the rivers into the lakes accumulate there. Similarly, the radioactivity of the organisms living in lakes is higher than that of those living in rivers. In the seas or oceans the case is not so clear-cut as the level of radioactivity also depends on depth.

The K-content of the water has a decisive influence on the natural radioactivity within the hydrosphere. The K-content of rivers and lakes depends on the geological environment, the soluble potassium-content in the soil and such anthropogenic factors as the sewage water load of communal origin or artificial fertilization. The average ^{40}K -activity of the water in the Danube has a value of around 70–90 mBq/l. However, the ^{40}K -activity of seawater is 2–3 orders of magnitude higher. This is due to the fact that, in addition to NaCl which is a dominant part of the salt content within the seawater, there is also a large amount of KCl. The ^{40}K -activity of seawater is 10–15 Bq/l.

In order to give an approximate idea of the radioactivity of water, first usually the so-called gross β -activity is given. This value—the activity of the dry material after evaporation—is the total activity of all non-volatile components and of course it contains in addition to the isotopes of natural origin even radionuclides like ^{90}Sr , ^{137}Cs causing contamination. In spite of the fact that the gross β -activity is not nuclide-specific, knowing the K-content of water it gives a clear-cut indication of the degree of possible contamination. The concentration of tritium is independent of the gross β -activity, as water with ^3H -content leaves during evaporation and the remnant does not contain any ^3H nuclides. The gross β -activity of Hungarian surface waters is presented in Table 18. In order to better judge the figures in the table, it must be mentioned that the gross β -activity of Danube water was usually 0.3–2.0 Bq/l in the 1970s and the 1980s. The often measurable relatively high values are due partly to the contaminating effect of the nuclear power plants built along the Danube, and partly to the influx of artificial fertilizers with potassium-content.

Among the isotopes of natural origin, the ^3H nuclide should be especially mentioned. The natural tritium activity of waters is about 0.6–0.8 Bq/l (Sáró, Duka-Zólyomi 1976).

Table 18 Average radioactivity of Hungarian surface waters

Year	Gross β -activity, Bq/l				
	Danube	Tisza (river in Eastern Hungary)	Rába (river in Western Hungary)	Sajó (small river in Northern Hungary)	Lake Balaton
1962	0.40	0.28	0.41	0.51	—
1964	0.33	0.96	0.78	—	—
1966	0.33	0.56	1.07	1.26	0.33
1968	0.11	0.52	1.63	1.43	0.26
1970	0.33	0.56	0.85	0.85	0.56

The radium, radon, ^{210}Pb , ^{210}Po content of surface waters is determined partly by the amount of uranium and radium dissolved in them, partly by the amount of radioactive substances coming from the air. If we take the average ^{226}Ra -content to be $3 \cdot 10^{-14}$ g/l (Yermolayeva et al. 1978), most of the ^{210}Pb and ^{210}Po content

Table 19 Natural radioactivity of waters

Rainwater Drinking water (tap water)	U-concentra- tion μg/l	²²⁶ Ra-	²²² Rn-	²¹⁰ Pb-	²¹⁰ Po-	Reference
activity, mBq/l						
Rainwater		4.6±0.3				Homma, Mura- kami 1977
Drinking water (tap water)		5.1±0.3				
Drinking water		3.7—37				Osmond 1964; Turner et al. 1961; Samuels 1964
Drinking water	0.3—50	26—1850				Belova, Lunkina 1967
Surface fresh water			37—3920			Belousov, Stuk- kenberg 1961; Hursh et al. 1965
Lakes in Finland				0.9—16.7	0.3—8.5	Kauranen, Miet- tinen 1971
Baltic Sea				0.3— 6.8	0.1—8.5	
Pacific Ocean				1.7— 4.8	0.9	Rama, Goldberg 1961; Holtzman 1964

of waters does not come from the derivatives of the ^{226}Ra decay, but originates from the air. Some data are summarized in Table 19 as regards the content of radioactive substances of natural origin in surface waters.

In order to be able to make a comparison, we have to mention that the radioactivity of mineral and medicinal waters may surpass significantly the active substance content of surface waters. The ^{210}Pb activity in certain radium-rich medicinal waters may be as high as 50 Bq/l (Andrews, Wood 1974). According to measurements made in Kovásznafürdő (Romania) the ^{222}Rn , ^{226}Ra and ^{40}K contents of medicinal waters are 0.3–6.1, 0.4–0.8 and 0.4–10.1 Bq/l, respectively.

When analysing these issues in relation to atmospheric radioactivity, we have already mentioned the fact that the better part of ^{14}C produced under the influence of cosmic radiation leaves the atmosphere by the continuous interaction among the atmosphere, hydrosphere, and a lesser degree, the flora. Through the exchange processes 94.3% of the ^{14}C produced is tied up in the hydrosphere, mostly in seawater in a dissolved state, in the form of $\text{H}_2^{14}\text{CO}_3$. The ^{14}C content of the hydrosphere is about 48 kg, equivalent to an activity of $7.8 \cdot 10^9$ GBq. The average residence time of ^{14}C in the surface layers of the oceans is 3–4 years, thus before the year 2000—provided nuclear weapon tests are discontinued—about 90% of the ^{14}C causing artificial contamination will be deposited in the deeper layers of oceans (Devirc et al. 1982).

2.2.2 Radioactive contamination of the hydrosphere

In addition to the radioactive isotopes of natural origin, the hydrosphere is also contaminated by several active isotopes of artificial origin. Among them, as we have already mentioned, ^{90}Sr and ^{137}Cs have special importance because of their radio-toxicity. Even the presence of the ^3H isotope causes a radiation load that is not negligible because at present its concentration, as we will see later, is 2–3 orders of magnitude higher as a consequence of the nuclear weapon tests and emissions of nuclear power plants than the level of natural tritium activity (representing the amount of ^3H produced under the influence of cosmic radiation).

The radiation load exerted by the radioactive contamination of the hydrosphere has both external and internal influences on the organisms. The external effect—e.g. the environmental radiation reaching the organism during a bath or taking a swim—is usually negligible compared to the internal effect caused partly by the activity of the water taken up by the organism, partly by the incorporation of radioactive substances through eating organisms living in the hydrosphere. So, radionuclides arrive in humans from the hydrosphere either directly or indirectly through the aquatic food chain. Incidentally, the radioactivity of drinking water is usually significantly lower than that of flowing (fresh) waters or surface waters. This is due to the different filtering, water-processing technologies. The radioactive substances falling out from the atmosphere or going directly into the hydrosphere are either in a dissolved or in a floating state. Of course their sedimentation, dissolution and migration goes on according to the same laws that apply to inactive substances (Frantz 1965; Hermann

et al. 1979). Some of the radioactive contamination reaches the bottom mud, some (the isotopes remaining dissolved) may accumulate in aquatic organisms. For instance, algae are able to accumulate certain radioactive isotopes; thus the examination of the radioactive contamination in algae indicates the severity of the aquatic pollution and the changes caused by the contamination. The considerable accumulation of certain radioactive isotopes within aquatic plants is demonstrated, for example by the measurements of Ruf (1973). He found that the ^{60}Co activity of Danube water had a value of 0.15–0.52 Bq/l; however, the mean ^{60}Co activity in aquatic plants was 3.3 Bq/l.

Two types of plant and animal organisms are suitable for indicating the radioactive contamination of the hydrosphere, or rather its temporal changes. This process is called radionuclide biological monitoring. The first type of indicator is the so-called actual-level monitor, which accumulates relatively quickly but also decomposes or evacuates quickly. For example, filamentous green algae are good biological monitors in flowing waters. The other type is the so-called integrating monitor. Its radionuclide content increases even at constant environmental level and stabilizes only after a very long time. These organisms, since the decomposition speed is low, accumulate the assimilated radioactive isotopes for a long time, thus their radioactivity gives information about the contamination level of a longer period (Ruf 1975; Holland 1980).

The radioactive contamination of surface waters, or rather the amount of radioactive isotopes passing on from them to the living aquatic plant and animal organisms is also influenced by the chemical composition of the water. To give an example, the possibility of ^{90}Sr uptake also depends on the inactive strontium and Ca content. The inactive alkaline-metal content of the water influences in a similar way the accumulation of ^{137}Cs and ^{134}Cs in the aquatic plants and fishes. So the ^{90}Sr uptake is lower from waters with high Sr or Ca content. The water of the Danube, for example, is of medium hardness; however, its degree of hardness at low output may be as high as 16 German degrees of hardness, although at high flows it may decrease to 8 German degrees of hardness. The salt concentration of the water is 160–370 g/m³, among the cations Ca^{++} occurs in the biggest proportion (an average of 66%).

Let us see some data about the radioactive contamination of surface waters. The ^{137}Cs and the ^{90}Sr activities of the water of the Baltic Sea are summarized in Table 20 (Stiro et al. 1978). Considering the data of the table which present the results of measurements made from 1973 to 1975, it can be stated that the deeper the water, the

Table 20 ^{137}Cs and ^{90}Sr activity of the water in the Baltic Sea

Depth (m)	^{137}Cs	^{90}Sr	$^{137}\text{Cs}/^{90}\text{Sr}$
	mBq/l		
0	15–26	18–33	0.8–1.1
50	15–30	18–30	0.8–1.1
> 50	15–33	11–26	1.2–2.0

higher the $^{137}\text{Cs}/^{90}\text{Sr}$ proportion of activities. The fact that the proportion of activities changes as a function of depth can be explained by the highest ^{90}Sr concentration measured near to the surface, and the ^{90}Sr activity gradually decreases with depth. At the same time the highest ^{137}Cs concentration can be measured not at the surface but at a depth of about 100–200 metres, and in the water layers immediately above the seabed the $^{137}\text{Cs}/^{90}\text{Sr}$ proportion of activities may attain the value of 5 (Dagrene et al. 1976). It is interesting to note that the deeper the water the higher the inactive Sr concentration. Near the surface the Sr concentration is usually 1.6–2.4 mg/l; at a depth of 400 metres it surpasses the value of 5 mg/l (Stankaytis, Stiro 1976).

In order to be able to judge the data in Table 20, we have to mention that according to the measurements of Trusov et al. (1975), the ^{90}Sr and ^{137}Cs content of the

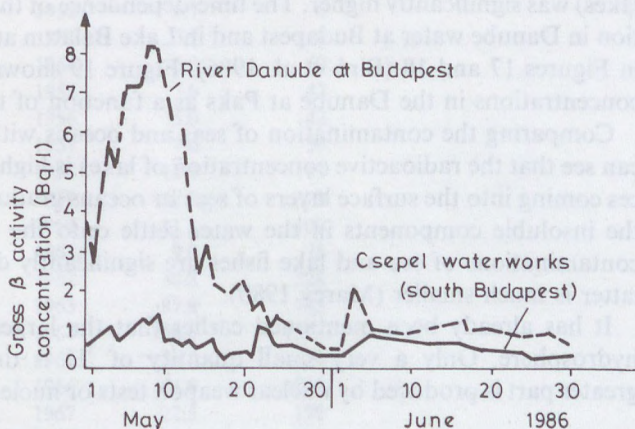


Fig. 17. Time dependence of the gross activity concentration in Danube water and the drinking water of Budapest

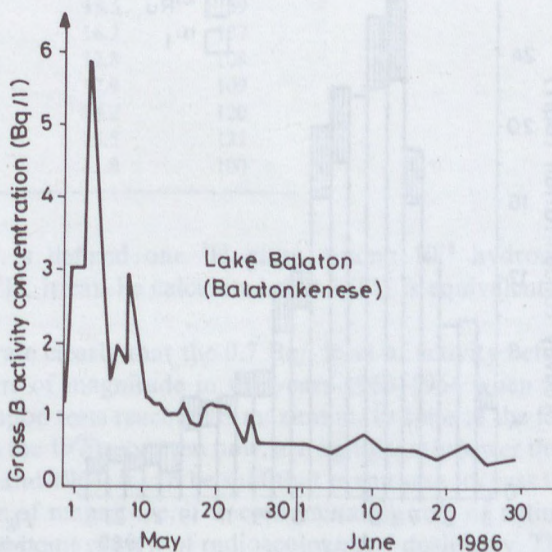


Fig. 18. Time dependence of the gross activity concentration in Lake Balaton at Balatonkenese

Baltic Sea is 23 ± 5 and 20 ± 4 mBq/l, respectively, the $^{137}\text{Cs}/^{90}\text{Sr}$ proportion is usually 0.7–1.0. According to Vakulovskiy et al. (1975), the mean concentration of ^{90}Sr and ^{137}Cs in the waters of the Baltic Sea is 26 and 33 mBq/l, respectively. The $^{137}\text{Cs}/^{90}\text{Sr}$ proportion is on the average about 1.2, and within the seawater there is about $(5-6) \cdot 10^{14}$ Bq ^{90}Sr and $(6-7) \cdot 10^{14}$ Bq ^{137}Cs . According to a report by Nakaoka et al. (1980), the ^{137}Cs activity in the Japanese Sea is only about 5–6 mBq/l, the ^{134}Cs concentration is less than 2 mBq/l.

The contamination level of seas and lakes is usually higher than that measured in rivers, where the accumulation rate of the radioactive isotopes is smaller. For instance, the ^{90}Sr activity of Danube water is usually only a few mBq/l, and its ^{137}Cs activity is on the average less than its ^{90}Sr activity. Of course in the “Chernobyl period” the contamination level of the Danube (and the other European rivers and lakes) was significantly higher. The time-dependence of the gross β -activity concentration in Danube water at Budapest and in Lake Balaton at Balatonkenese is presented in Figures 17 and 18 (Biró et al. 1986). Figure 19 shows the ^{131}I and ^{103}Ru activity concentrations in the Danube at Paks as a function of time (Biró et al. 1986).

Comparing the contamination of seas and oceans with the pollution of lakes, we can see that the radioactive concentration of lakes is higher. The radioactive substances coming into the surface layers of seas or oceans gradually reach the deeper layers, the insoluble components in the water settle onto the seabed. The ^{90}Sr and ^{137}Cs contaminations of sea and lake fishes are significantly different, usually that of the latter is much smaller (Marey 1980).

It has already been mentioned earlier that the largest part of ^3H occurs in the hydrosphere. Only a very small quantity of ^3H is due to cosmic radiation, its greater part is produced by nuclear weapon tests or nuclear power plants. As a matter

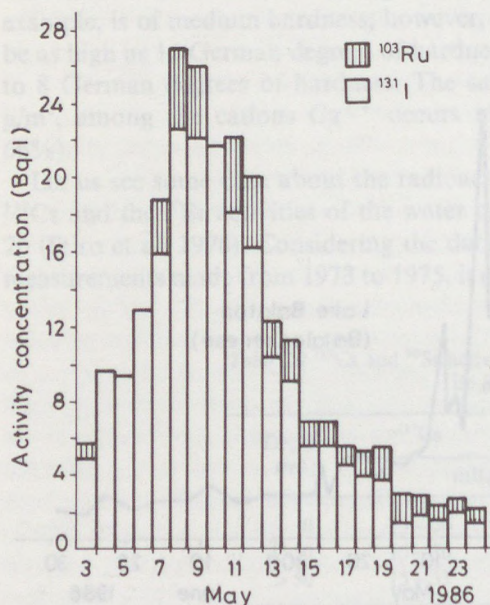


Fig. 19. Time dependence of ^{131}I and ^{103}Ru activity concentrations in the Danube at Paks

of fact, the migration of ^3H among the elements of the biosphere virtually corresponds to the water cycle, as ^3H occurs within the water (HTO), i.e., there is a continuous exchange between the atmosphere and the hydrosphere.

Data about the ^3H concentration are given in Tables 21, 22 and 23. Here the tritium activity is given not only in units of Bq/l but also in tritium units (TU), often

Table 21 Average tritium concentration of ground waters in Central Europe (Sáró, Duka-Zólyomi 1976)

Year	^3H -activity	
	Bq/l	TU
1951	0.7	6
1952	0.7	6
1953	0.7	6
1954	5.9	50
1955	5.0	42
1956	5.0	42
1957	7.0	59
1958	13.4	113
1959	38.2	323
1960	11.8	100
1961	9.0	76
1962	38.8	328
1963	87.9	743
1964	156.6	1323
1965	50.2	424
1966	24.8	210
1967	22.5	190
1968	23.7	200
1969	21.3	180
1970	18.5	159
1971	16.2	137
1972	12.8	108
1973	12.9	109
1974	14.2	120
1975	15.5	131
1976	11.8	100

used in the literature. One TU is defined one ^3H atom among 10^{18} hydrogen atoms. Knowing the half-life of ^3H , it can be calculated that 1 TU is equivalent to 0.1184 Bq/l.

The data in Table 21 demonstrate clearly that the 0.7 Bq/l level of activity before 1953 was surpassed by two orders of magnitude in the years 1963–1964 when the contamination due to nuclear weapon tests reached its maximum. In spite of the fact that the tritium levels measured in the 1970s, or even now, are significantly lower than the values characteristic for 1963 and 1964, it can be said that even these surpass the natural background by one order of magnitude. The contaminating role of tritium cannot be underestimated from the point of view of radioecology and dosimetry. This

issue should be stressed, because the ^3H emission of the nuclear power plants, of increasing importance in the energy supply system, is very significant (Gusev 1975; Gedeonov et al. 1975; Vakulovsky et al. 1978).

Let us compare the data in Tables 21, 22 and 23. In 1971 and 1972 the tritium concentration of natural surface waters in Hungary was 21–29 Bq/l. In 1977 and 1978 the ^3H activity of the Danube water at Gönyű, Budapest, Paks and Mohács was 12–16, 11–41, 12–21 and 12–25 Bq/l, respectively. The ^3H concentration in 1972 in

Table 22 Tritium-activity of the rivers and precipitation in the Soviet Union in 1975–1976 (Vakulovsky et al. 1978)

Study area	^3H -activity			
	riverwater		precipitation	
	Bq/l	TU	Bq/l	TU
Baltic states	7.7 ± 1.8	65 ± 15	7.1	60
Europe, North	11.6 ± 1.3	98 ± 11	7.9	67
Europe, Central part	11.1 ± 0.8	94 ± 7	12.4	105
North Siberia	19.7 ± 2.1	166 ± 18	18.8	159
South Siberia	17.8 ± 3.7	150 ± 31	18.0	152
Far East	18.9 ± 2.5	160 ± 21	17.2	145
Central Asia	15.0 ± 0.4	127 ± 3	11.6	98

Table 23 Average ^3H -concentration of the sea-water in 1974–1975 (Vakulovsky et al. 1978)

Sea	^3H -activity	
	Bq/l	TU
Baltic Sea	9.6	81
White Sea	6.3	53
Black Sea	4.5	38
Caspian Sea (lake)	5.2	44

the Baltic sea was 7.3 ± 2.7 Bq/l, its total tritium content was assessed to be $1.4\text{--}1.5 \cdot 10^{17}$ Bq (Vakulovsky et al. 1975). In 1976 the ^3H concentration of the water of the Danube, Tisza and Sava rivers in Yugoslavian territory varied between 10 and 23 Bq/l (Hadzisehovic et al. 1978). In 1977 in Hungary the tritium activity of the precipitation was on the average 15 Bq/l; the mean ^3H activity of the drinking water in Budapest was 21 Bq/l.

In Table 24 some comparative radiometrical data from the analysis of water samples taken from Hungarian flowing and still waters can be seen. Based on these data it can be stated that the radioactive contamination of the water in the Holt-Szamos (Dead-Szamos) was significantly higher than the radioactivity of the Hortobágy river or that of the fishponds in the Hortobágy plains. It is due to the fact that the Holt-Szamos is virtually a lake, with a minimal current, and the radioactive

substances accumulate through fall-out as well as through wash-in from the embankment and sewage waters. The radioactive contamination of the fishponds in the Hortobágy plains and that of the Hortobágy river was nearly the same, and significantly lower than that of Holt-Szamos. This is easy to understand, as the fishpond water is changed yearly, the water supply of the ponds is taken from the Tisza through the Hortobágy river.

According to the data of Table 24, only about 30% of the residual activity (the difference between potassium activity and total activity) is due to the metal ion fraction activity indicating radiostrontium pollution. It means that the pollution of the waters with ^{137}Cs and possibly with other isotopes is not negligible.

Table 25 summarizes a Hungarian survey, the results of water measurements from all over the country coordinated by the Radiological Control Network of the Ministry

Table 24 Average radioactivity of the Hortobágy River, fishponds in Hortobágy plains and Holt-Szamos (Dead-Szamos)

Sampling locality	Total- β	Potassium-activity	Residual-activity mBq/l	Metal ion-fraction
Hortobágy River	205	36	169	60
Fishponds	150	32	118	42
Dead-Szamos	496	155	341	73

Table 25 Results of water measurements carried out by the RCN* in 1980

Total β -activity	(mBq/l)	223 ± 209
^{90}Sr -activity	(mBq/l)	14 ± 19
^{90}Sr -activity	(mBq/g Ca)	278 ± 247

* Radiological Control Network of the Ministry of Agriculture and Food

of Agriculture and Food in 1980. Part of the water samples ($n=56$) were taken from surface flowing waters, another part from wells. There was a significant difference in the ^{90}Sr activity between the two types of samples, the activity of the surface waters being higher. This result is in agreement with another water investigation performed in Győr-Sopron county (Hungary), which demonstrated that the radioactivity of drinking water in the cities of Sopron and Győr is lower than that of surface waters. It is related to the fact that, on the one hand, the pollution of groundwater is usually lower than the contamination of surface waters, and on the other hand, the pollution of the water coming from waterworks built in the vicinity of large rivers is decreased significantly by filtration through the different layers of soil, as well as by various technologies.

We have already mentioned that the operation of the different nuclear reactors, power plants and facilities produces a relatively significant amount of radioactive

substances, some which arrive in the different elements of the biosphere. However, the contamination—with some exceptions, e.g., ^{85}Kr , which occurs in the atmosphere—appears in the immediate environment of the polluting source. To cite an example: measuring the ^{134}Cs , ^{137}Cs , ^{65}Zn and ^{60}Co activities in the water and different aquatic organisms in the environment of the Big Point power plant (USA), it was found that the activity levels were 3–10 times higher than before the emission (Nelson et al. 1972). Similar results were reported by Kaye (1969), who measured the activities of ^{54}Mn , ^{58}Co and ^{134}Cs in waters and in aquatic organisms coming from the nuclear power plants.

The amount of radioactive iodine isotopes (^{129}I and ^{131}I) emitted from the nuclear facilities into the air and water is also not negligible. During nuclear fission a significant amount of iodine isotopes is produced; it accumulates within the fuel rods and, through the micro-fissures of the fuel units, may escape into the primary water circle. The plants recycling the burnt-out nuclear fuel rods, the so-called reprocessing facilities, emit an important amount of ^{129}I isotope into the hydrosphere and atmosphere. The emitted radioactive iodine may be taken up by living organisms and accumulation in the thyroid gland of man may cause rather serious radiation load. Within a 5 km radius around the reprocessing centre in West Bell (USA), e.g., the ^{129}I content of the thyroid glands in animals—about $4 \cdot 10^3$ Bq/g iodine—was 3–4 orders of magnitude larger than the background level measured far away from the nuclear facilities (Zikova, Gneusheva 1976). In a river draining the sewage waters of a reprocessing plant an iodine activity of $0.2\text{--}2.2 \cdot 10^4$ Bq/g was measured.

During the operation of nuclear plants, nuclear reactors and isotope laboratories, the production of radioactive sewage waters must be expected, which can only be released into the recipients if the contamination is below the maximum permitted specific activity level.

In Hungary, the sewage water processing system of the reactor of the Budapest Technical University (operating since 1971) processes the sewage waters from 3 radiochemical laboratories and the “hot chamber”. If the specific activity of the sewage water is smaller than the maximum permitted to be released, the sewage water goes immediately into the Danube. If the specific activity is not more than 10 times higher than the maximum permitted activity, the radioactive sewage water goes first into a diluting basin, and only after the necessary dilution may it be drained into the Danube. If the contamination of the sewage water is higher than ten times the value of the acceptable level, the sewage water goes first to an ion exchange column line and then through the dilution basin into the Danube.

In addition to the stationary nuclear facilities, which emit from well-identified places, the polluting effect of nuclear power ships should be mentioned, too. However, many investigations have demonstrated that the radioactive emission of nuclear power ships is extremely low, seawater will be contaminated only insignificantly (Sivincev 1961; Miles et al. 1974).

2.2.3 Summary

In this subchapter the radioactivity of natural and artificial origin of the hydrosphere and the factors influencing its levels have been examined. The most important statements may be summarized as follows:

1. The radioactivity of the hydrosphere is formed in continuous interaction with the other elements of the biosphere. The most important interactions are that between the atmosphere and hydrosphere and that between the lithosphere (or rather the soil) and the hydrosphere. Radioactivity and its temporal change depends to a great extent on whether a sea, a lake or a river is in question. Other decisive factors include the volume of the recipient and the depth and chemical composition of the water as well as the hydrodynamical (rheologic) conditions.

2. Among the radioactive isotopes of natural origin, mainly potassium and uranium and radium (or rather their derivatives) occur. Of the radionuclides produced in the atmosphere as a consequence of cosmic radiation, and then from there entering the waters, ^{14}C is the most important. The natural radioisotope concentration of mineral and medicinal waters is usually significantly higher than that of surface waters.

3. The contamination of the surface waters resulting from nuclear explosions reached its maximum in 1963–1964; at present it is significantly lower. Of the radionuclides contaminating waters and accumulated within the aquatic organisms, ^{90}Sr and ^{137}Cs are the most important, because they cause lasting radiation load. In the environment of nuclear facilities, water may also be polluted with other radioactive isotopes, e.g., with radioactive iodine; however, it should be stressed that the hydrosphere-polluting effect of the nuclear power plants is effectively only local, except in the case of breakdown, and their global contaminating effect is small. However, the nuclear disaster at Chernobyl in 1986 has produced in most European countries a global contamination of the hydrosphere. Among the isotopes of artificial origin, ^3H should be especially mentioned. Its concentration level—chiefly because of the atmospheric nuclear tests—even today is more than one order of magnitude higher than the natural background level.

4. Other factors also influence the radioactivity of the hydrosphere, such as aquatic organisms (algae and fish may accumulate radionuclides) or factors related to agricultural activities such as the amount of artificial fertilizers washed into the surface waters, the quantity of sewage waters, the fishing activity, etc. The radioactive pollution of sea fish is usually lower than that of fish coming from freshwater lakes. This is due to the fact that the radioactive substances passing from the atmosphere into the water may accumulate more readily in lakes because of their shallower depth. The contamination of fish in rivers is usually significantly lower than that of fish from lakes, apart from the rivers which have a high-power nuclear facility in their environment.

2.3 SOIL RADIOACTIVITY

The soil plays an important, sometimes decisive, role in the absorption and accumulation of the different radioactive substances. The natural radioactivity of the soil depends on the geological circumstances. For instance, in Kerala State in India, there is an important amount of monazite with high thorium content in the soil, thus the intensity of the background radiation due to the natural radioactivity of the soil is by one order of magnitude higher than the average value on the surface of the Earth. Actually, similar examples may be found in other places, too. In Hungary, for instance, the granite stone in the Velence Hills has a high radium content. The α -activity of the soil in Hungary is usually 0.4–3.0 Bq/g, that of the granite in the Velence Hills is higher than 4 Bq/g.

It has already been mentioned that the amount of radioactive isotopes coming to the surface of the Earth by gravitational deposition or by precipitation is determined to a great extent by the geographical situation. To give an example, contamination in the northern hemisphere is higher than in the southern, and in North America the values (at the same latitude) are higher than in Europe. The level of ^{90}Sr contamination is highest between 30 and 50 degrees of latitude (Baranov et al. 1965).

In the buildup of radioactive pollution, the soil plays a mediating role, i.e. absorbs the fission products coming from the air, rainwater or surface water, which pass onto the living organisms in the biological cycle of the elements. During the development of the level of certain radioactive substances, e.g., radiostrontium, in the subsequent members of the biological chain process (plant–animal–man), the soil plays a basic role, as it is above all the contamination level of the soil that determines the radiation pollution of the members of the biological chain process.

The level of radioactive contamination is also influenced by the relief of the terrain. There is a difference in the absorption of radioactive substances between plains and mountainous regions (Raikov et al. 1970). It could be ascertained by examination of plants produced on hillslopes that because of the lower contamination level of the soil the uptake of radioactive substances by these plants was significantly lower than in the plains of the same region. This is partly due to the fact that the majority of the radioactive substances precipitated, mainly through heavy rainfalls, flow down to the deeper layers before being absorbed on the surface; thus a smaller amount of radioactive substance reaches the plants from the soil. Another reason is that most of the radioactive isotopes coming to the soil by wet or dry deposition are absorbed in the upper layers of soil. Because of the soil erosion observable on slopes of hills, which may occasionally be very severe, the upper layers of soil are carried away, and the eroded material with high contamination accumulates in the deeper pockets. According to measurements carried out in Oregon State (USA), on eroding slopes and hills the average ^{137}Cs activity of the soil surface was 300 mBq/cm²; where the eroded material was deposited the same measurement gave a mean value of 440 mBq/cm² (Brown et al. 1981). In the sites studied the amount of erosion was 0.6–2.7 ton/year/ha.

In addition to the sources of contamination, the radioactive contamination of the different soils is influenced decisively by the mechanical structure and chemical composition of the soil, as well as by the agrochemical and agrotechnical methods

applied. So, the radioactive substances arriving at the surface will be absorbed in different degrees, according to the mechanical structure of the soil. In the case of light soils, these substances will be washed quickly into deeper layers. However, plants are not able to take up and pass on to the animal or human organisms the active isotopes from the deeper layers. In fact, the radioactive contamination of the soil would not be interesting in itself if the radioactive isotopes in the soil did not reach the different members of the plant-animal-man biological chain. Of course, the radioactivity of the soil has also a direct influence on the plants, animals and humans living in the given environment, as the radioactive isotopes within the soil exert a direct radiation load. However, the radiation dosage sustained by the living organisms because of the radioactivity of the soil is usually much smaller than the inner radiation load caused by the incorporated isotopes. Here it should be mentioned that the natural γ -radiation of the soil is a function of its humidity, as higher moisture content decreases the radiation. As the radiation of soil is mostly due to the isotopes ^{40}K , ^{238}U and ^{208}Tl , and their concentration does not change significantly with time, on the basis of measurements, e.g. by airborne measurement, the moisture content of soil can be assessed (Carroll 1981).

The amount of radioactive substances reaching plants from the soil, is also influenced by the chemical composition of the soil. A dominant factor is the proportion of the inactive and active components of the given element, or rather the amount of a stable component chemically similar to the radioactive substance in question. Thus the application of certain agricultural methods (e.g., liming the soil) may contribute to the decrease of the active materials transferred from the soil to the plants. However, the effect may also be the opposite. It is well-known that extensive use of phosphate fertilizers in the soil significantly increases the concentration of radioactive isotopes of natural origin.

Last but not least, in the migration of radioactive substances the sequence of soil-plant-animal-man elements, or rather the direction of the chain is most important. This process may also be reversed, e.g. by the use of organic manure. Therefore in the buildup of radioactive contamination of the soil, transport in the opposite direction should be considered, too. In the migration of radioactive isotopes, even the direction from the soil to the atmosphere or from the soil through wash-out to the hydrosphere cannot be neglected.

In this subchapter, the processes of natural radioactivity buildup and radioactive contamination will be demonstrated, analysing factors like the mechanical and chemical structure of the soil, agrotechnical and agromechanical methods, amelioration of which has an influence on the level of radioactivity.

Of course, the soil is in continuous interaction with the other elements of the biosphere, and the actual level of the soil radioactivity, through the influence of ecological factors, is determined by the quantitative and qualitative features of these very complex interactions. The question of modelling the migrations of the radioactive isotope within the different soils is discussed in detail by Anokhin (1974).

2.3.1 Natural radioactivity of the soil

The level of natural radioactivity in soils is a function of the geological circumstances, agricultural production (use of fertilizers, irrigation, etc.) and other environmental and meteorological parameters. We should mention that, in the environment of coal-burning power plants, the ^{232}Th and ^{226}Ra content of soil may increase, as some of the fly ash of the power plant is deposited onto the surface. With the exception of ^{210}Po which sublimates at 400°C , the natural radioactivity content of the coal is concentrated in the fly ash. According to measurements carried out in Hungary in 1973 and 1974, in the fly ash of the thermal power plant in Pécs, the mean activity of ^{228}Th , ^{226}Ra and ^{40}K was 0.24, 0.31 and 0.85 Bq/g, respectively.

A significant part of the natural radioactivity of soils is caused by the ^{40}K isotope of natural potassium. This isotope emitting β and γ rays occurs everywhere in natural potassium with a frequency of 0.0119%, thus the share of ^{40}K activity within the radioactivity of soil is a clear-cut function of the potassium content of the soil. As already mentioned, the average potassium content of soils is about 1–2%; however, in the case of soils formed from granite it can be as high as 3–4%. Thus the fertilization of soils with potassium increases the radioactivity of the soil.

Rubidium, which has similar chemical features to that of potassium, occurs within the soils, too. Of the natural rubidium, 27.85% is radioactive ^{87}Rb . As the average rubidium content of soils is as low as $6 \cdot 10^{-3}\%$, the ^{87}Rb activity of soils is usually smaller by a few orders of magnitude than that of ^{40}K . The salts of rubidium, like the salts of potassium and sodium, dissolve well in water; thus they may go on through the plants to the animal and human organisms.

The radon content of the upper layers of the soil is a function of the season, or rather of temperature. Maximum concentration occurs in summer, its minimum is in winter (or rather in autumn). This phenomenon is due to the temperature gradient. In summer the gases flow inward and not outward (into the atmosphere), from the soil; thus the radon remains within the soil. This is the reason why in the upper layers the maxima can be measured. In colder months, when the medium air temperature is lower than the average temperature of the corresponding soil layers, the process is reversed; at this time minima can be measured. To the above-mentioned changes usually a "narrow" winter peak is added, which is due to the freezing of the surface layers of the soil, which in turn contributes to the decrease of radon flow-out (Somogyi et al. 1981).

The radon concentration greatly depends on depth. For instance, at a depth of 2.5 metres the radon concentration of the air within the soil is an order of magnitude higher than at the depth of 0.5 m (Kraner et al. 1964; Yermolayeva-Makovskaya, Litver 1978). In addition to the geological factors, the radon concentration of the upper layers is also determined by the gas permeability of the soil. On the basis of measurements carried out in the European part of the Soviet Union, it was established that the radon concentration in sandy soils was only about one-fifth of the value measured in clayey soils because of the poor gas permeability of the latter. In the case of sandy soils the speed of radon exhalation (emission) is usually 150–300 Bq/m²

day (Sisigina 1965). The ^{210}Pb and ^{210}Po content of soils is 15–300 and 5–220 mBq/g, respectively (Yermolayeva-Makovskaya, Litver 1978).

Table 26 summarizes the values of natural radioactivity (fission series of ^{238}U and ^{232}Th) on the basis of measurements of soil samples ($n=35$) collected in Hungary by the Radiological Control Network of the Ministry of Agriculture and Food, and analysed by γ -spectrometry in the National Research Institute for Radiobiology and Radiohygiene. In both series the last members—Bi and Pb or rather Bi, Pb and Tl—are virtually in equilibrium because of their short half-lives.

The differences in the activities of the samples according to the place of origin can be evaluated on the basis of the standard deviations of the results. According to the data of Table 27, five classes can be established considering the values of standard deviations.

The uranium content of the soils is a function mostly of the geological and geomorphological situation of the region. The average uranium, thorium and radium content of soils is $1 \cdot 10^{-4}\%$, $6 \cdot 10^{-4}\%$ and $8 \cdot 10^{-11}\%$, respectively. According to the measurements carried out in Hungary, the uranium content of the soils (podsol,

Table 26 Natural radioactivity of soil samples taken in 1980

Radionuclide		Activity, mBq/g soil
^{238}U derivatives	^{40}K	602 ± 146
	^{234}Th	85 ± 36
	^{226}Ra	111 ± 64
	^{214}Bi	45 ± 29
	^{214}Po	49 ± 32
^{232}Th derivatives	^{228}Ac	40 ± 12
	^{212}Bi	39 ± 16
	^{212}Pb	39 ± 11
	^{208}Tl	40 ± 9

Table 27 Classification of Hungarian soils based on their radioactive isotope content of natural origin

Radionuclide	Activity, mBq/g soil				
	low	lower than medium	medium	higher than medium	high
^{40}K	< 373	$373-529$	$529-675$	$675-821$	> 821
^{234}Th	< 31	$31-67$	$67-103$	$103-140$	> 140
^{226}Ra	< 16	$16-79$	$79-143$	$143-206$	> 206
^{214}Bi	< 3	$3-31$	$31-59$	$59-87$	> 87
^{214}Pb	< 4	$4-34$	$34-64$	$64-94$	> 94
^{228}Ac	< 24	$24-34$	$34-46$	$46-56$	> 56
^{212}Bi	< 15	$15-31$	$31-47$	$47-63$	> 63
^{212}Pb	< 21	$21-33$	$33-45$	$45-57$	> 57
^{208}Tl	< 26	$26-35$	$35-45$	$45-54$	> 54

open country soil, windblown sand, meadow clay) is usually 0.0–3.0 mg/kg. However, certain muds and marsh soils have a much higher uranium content, about 10 mg/kg. According to a communication of Nadkarni (1974), the average thorium concentration of the soils is 7 mg/kg. According to Loss (1975) clay stones with a relatively low thorium and uranium content have 10–15 and 4–6 mg/kg thorium and uranium content, respectively. Gil et al. (1981) examined the ^{226}Ra absorbing capacity of different kinds and layers of soil, and established that the Ca and Ba content determines decisively the degree of absorption.

Arhipov et al. (1981) conducted an extensive investigation in order to determine the total, acid-soluble, and the exchangeable (water-soluble) ^{238}U , ^{232}Th , ^{226}Ra , ^{210}Po and ^{210}Pb content in the cultivated soils of the Soviet Union. It was established that the amount of water-soluble active substances, which the plants are able to take up, is generally not higher than 10%. At the same time it was found that, except for ^{232}Th , the other radionuclides are soluble in acidic media and occasionally, as a function of the chemical characteristics of the soils, as much as 70% of the total amount is dissolved. In highly acidic soils the adsorption of these isotopes is low; thus the plants may more easily take them up. Incidentally, the pH value of the cultivated soils, except that of the sodic soils, is between 4 and 8.

2.3.2 Radioactive contamination of the soil

In addition to the natural radioactive substances occurring in the soil, many radioactive isotopes of artificial origin pollute it. They are emitted into the environment by nuclear tests and by nuclear power plants and nuclear facilities. Among the isotopes causing long-lived radioactive contamination in the soil ^{90}Sr and the ^{137}Cs have outstanding importance. The better part of these isotopes reach the surface of the soil from the atmosphere by dry or wet fall-out, where they either are fixed or pass on down to the lower layers of soil. The latter movement is a function of the mechanical and chemical features of the soil, or rather of the agrotechnical and agrochemical methods used.

Not considering the desert and semi-desert regions most of the radioactive isotopes reach the surface by wet fall-out (rain-out, wash-out); thus the level of radioactive contamination of the soil is determined first of all by the amount of precipitation. As already mentioned, for a region with normal or high precipitation, the following relationship can be used as a good approximation

$$y = c \cdot x$$

where:

y = radioactive pollution of the Earth surface;

x = amount of precipitation.

According to this equation, there is a linear relationship between the radioactive contamination of an area and the amount of precipitation. The above relation—on the basis of metal ion fraction activity indicating radiostrontium pollution—was also borne out by measurements carried out in Győr–Sopron county in Hungary from

1968 to 1973. We have established that there is a correlation between the radioactive contamination of an area and the amount of precipitation. The level of contamination is also influenced to a significant degree by the years and seasons (meteorological and climatic parameters) as well as by local (pedological) parameters.

To describe the temporal changes of radioactive pollution in the soil, an exponential mathematical model may be well applied, considering that, as a consequence of the radioactive decay and the transport processes within the soil, its radioactivity decreases. However, at the same time new radioactive substances may arrive on the surface of the soil from the atmosphere, which increase the contamination level. The change in the radioactivity of the soil is described by the following differential equation:

$$\frac{dQ(t)}{dt} = -\lambda Q(t) + \lambda_3 m(t),$$

where:

$Q(t)$ = radioactive isotope content of soil at time t ;

$\lambda = \lambda_1 + \lambda_2$, λ_1 being the radioactive decay constant and λ_2 being a constant indicating the speed of the release of the radioactive substance from the soil;

λ_3 = a constant indicating the fall-out speed of the radioactive substance from the atmosphere;

$m(t)$ = the degree of radioactive fall-out from the atmosphere.

In order to characterize the contamination level of Hungarian soils, data referring to the ^{90}Sr content in different types of soil samples measured between 1972 and

Table 28 Average values of ^{90}Sr activity in soils of arable fields and meadows

Year	Number of samples	Cultivation method	^{90}Sr activity	Bq/kg
1972	17	arable land	5.54	
	17	meadow	7.71	
1973	17	arable land	5.21	
	18	meadow	6.66	
1974	18	arable land	5.20	
	19	meadow	5.99	
1975	18	arable land	4.75	
	19	meadow	5.89	
1976	17	arable land	2.70	
	17	meadow	4.16	
1977	18	arable land	3.77	
	19	meadow	4.20	
1972–1977	105	arable land	4.53	
	109	meadow	5.77	
	214	arable land + meadow	5.15	

1977 are presented in Table 28. The survey completely covered Hungary and included all more important types of soil (sand, meadow soil, chernozem (black soil), brown forest soil, solonch (salt soil)). According to measurements in the period of survey, ^{90}Sr pollution had a value between 1.9 and 16.7 Bq/kg, in the samples, which were always taken from the upper 0–5 cm soil layer. In the 214 soil samples, 65% had an activity of 3.7–11.1 Bq/kg, 30% had a value lower than 3.7 Bq/kg, and in 5% the activity value was higher than 11.1 Bq/kg. The highest activity values were measured in meadow, solonch and forest soils with high clay or adobe content.

In order to characterize the contamination level, Table 29 summarizes the measurement data of soil samples ($n=94$) examined in 1980 by the Radiological Control Network of the Ministry of Agriculture and Food. The samples were taken from the upper 0–10 cm layer of the soil, from all over Hungary. It can be seen that the ^{90}Sr and ^{137}Cs activity is one order of magnitude lower than the ^{40}K activity, which makes up for most of the total β -activity of the soils. It can be stated, especially in the case of ^{90}Sr , that there are significant differences among the levels of contamination in different regions, that is to say, the standard deviation of the data is very high. The most polluted regions are in Transdanubia (the part of Hungary that lies between the Danube, the Drava and the western borders of Hungary); the least polluted regions are those between the Danube and the Tisza rivers. These differences are due to the different precipitation conditions.

Table 29 The results of soil investigations carried out in 1980 by the RCN

Total β -activity	Bq/kg	705 $\pm$ 211
^{90}Sr activity	Bq/kg	21 $\pm$ 24
^{90}Sr activity	mBq/g Ca	621 $\pm$ 1530
^{137}Cs activity	Bq/kg	16 $\pm$ 7

Table 30 Average ^{90}Sr and ^{137}Cs contamination of arable land soils in Czechoslovakia

Year	Exchange- able Ca mg/g	^{90}Sr activity		Exchange- able K mg/g	^{137}Cs activity		$^{137}\text{Cs}: ^{90}\text{Sr}$
		Bq/kg	Bq/g Ca		Bq/kg	Bq/g K	
1964	4.47	8.88	1.99	1.56	8.99	5.76	1.01
1965	4.41	6.07	1.38	1.46	13.10	8.97	2.16
1966	4.55	4.51	0.99	1.95	10.55	5.41	2.34
1967	3.08	6.25	2.03	1.15	15.61	13.58	2.50
1968	2.69	15.58	5.79	1.18	12.03	10.19	0.77
1969	7.03	9.92	1.41	0.68	16.35	24.05	1.65

The radioactive contamination of the soils in Hungary, due, just as in other European countries, to the nuclear tests, had their peak value in the years between 1962 and 1964.

In order to characterize the $^{137}\text{Cs}: ^{90}\text{Sr}$ proportion in soils, Table 30 summarizes the average results of a survey carried out by Csupka (1975) in Czechoslovakia from

1964 to 1969. The exchangeable Ca and K content (taken up by plants) and the ^{90}Sr and ^{137}Cs activity, as well as their proportion, were measured in soils sampled in arable lands from the 0–25 cm layer. It can be seen that the $^{137}\text{Cs}:^{90}\text{Sr}$ proportion varies between 0.77 and 2.50, and its average value is about 1.5. This result corresponds with the data of Boltneva et al. (1977), who found that the value of

Table 31 Activity of soil samples in the autumn of 1986, in Hungary

Activity-component	Bq/kg
Total β -activity	718 ± 198
^{90}Sr	8 ± 6
^{137}Cs	135 ± 100

$^{137}\text{Cs}:^{90}\text{Sr}$ proportion is about 0.55 ± 0.12 in the case of grey forest soil and chernozem, and 0.62 ± 0.15 in grey-brown soils and sandy soils. The $^{90}\text{Sr}:^{137}\text{Cs}$ proportion of fall-out due to nuclear tests is similar, with a value of about 1.2–1.6.

As was reported earlier, as a consequence of the Chernobyl disaster, very high contamination levels were registered in many European countries. Figure 20 shows the ground surface contamination in Hungary (Andrási et al. 1986). It can be seen that the distribution of radioactive contamination is rather heterogeneous. The difference between the most and least contaminated areas is significant (by one order of magnitude). Some data are given in Table 31 to characterize soil contamination in Hungary in 1986. It is stressed that in May, 1986, in some places the total β -activity of the investigated soil samples reached 5000–8000 Bq/kg.

As a consequence of the operation of the increasing number of nuclear power plants, nuclear facilities and isotope laboratories, a growing amount of radioactive waste is produced. In order to deposit it with safety, it is imperative that it should not contaminate the environment for several hundred years to come. Up to now safe storage was effectuated considering above all the geological and hydrological circumstances, e.g. storage in old salt mines. Of course, when selecting the proper place for an isotope depository, circumstances such as the distance from the nearest settlement, future development plans for the region and, last but not least, the possibility of enlargement must be considered. Incidentally, from the point of view of public health, centralized waste disposal is preferable.

An often-used method of solid state nuclear waste storage is deposition in the upper layers of the soil. As in such cases the contaminating effect of radioactive substances appears through the ground water, there are bylaws prescribing the distance of deposition from the highest level of the ground water. For example according to Soviet bylaws the bottom level of a storage must not be nearer than 4 metres above the highest level of ground water (Belicky, Orlova 1968).

The radioactive substances from the waste stored within the soil may come, because of the leaching effect of the precipitation water, through the unsaturated zone to the saturated zone, i.e., into the ground water. This may in turn directly pollute the

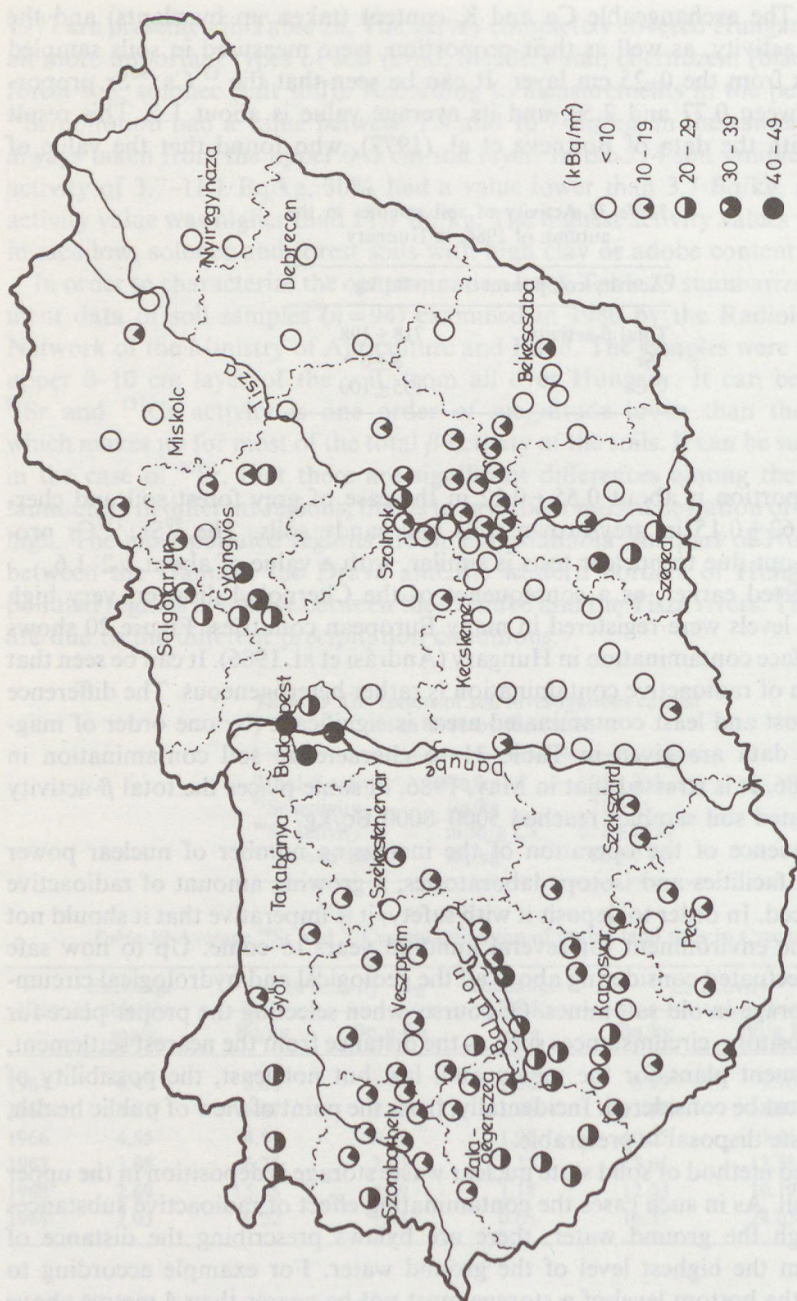


Fig. 20. Ground surface contamination corrected for 15 May 1986 measured at different locations in Hungary

surface waters, the sites of the waterworks and also the plant-animal-man biological chain. The selection of a suitable site for radioactive waste storage, or rather the evaluation of safe storage, is based on considerations of the distance between the ground water level and the waste storage, the adsorption characteristics and the water permeability of the soil, the slope of the site, its distance from surface waters, or rather from inhabited areas. In order to protect the surface waters, the radioactive waste storage site must be as far as possible from lakes and rivers. In the case of soil with coarse-grained sand or gravel, the ^{90}Sr dissolved in ground water may migrate within a year to more than 50 metres. The issue of the ^{90}Sr migration between the ground and surface waters is discussed in detail, e.g. by Rovinsky et al. (1976).

The behaviour of radioactive substances within the soil is influenced by many factors, primarily pedological, hydrological and meteorological. The main interactions between soil and radioactive isotopes are as follows: filtration, adsorption, ion exchange, chemisorption and biosorption. Filtration is a process by which the polluting substances being in a colloid state in the water, become fixed according to the mechanical structure of the soil. The smaller the grains forming the soil, the higher the efficiency of the filtration. Adsorption means a surface fixation of a physical nature. Clay particles fix very well the different active substances. Ion exchange is a chemical process whereby certain stones, rocks—above all because of their capacity of cation exchange—are able to fix the radioactive cations (possibly even anions) dissolved in the water and in exchange they release free inactive ions into the water. These zeolites set free calcium and sodium instead of radiostrontium and radiocesium, respectively, which are in turn fixed. Chemisorption is another chemical process when part of the radioactive substances dissolved in the water enter into chemical interaction with the rock or soil, thus producing hardly soluble compounds. Finally, biosorption is a process whereby the micro- and macro-organisms living within the soil or water assimilate the radioactive substances into their body.

It has already been mentioned that, from the point of view of safe (to the environment harmless) storage of the radioactive substances, the distance between the bottom of store and the ground water level, i.e., the unsaturated zone is very important. The migration of radioactive substances in the unsaturated zone is influenced decisively by the water-permeability of the soil layer. If the interaction between the soil and the radioactive isotopes is not considered, it is clear that the maximum spreading speed of the radioactive isotopes is a function of the water permeability of the soil. Of course, the interaction between the soil and the radioactive substances decreases this speed. Just to impress upon the reader the significance of this issue, we mention that the water-permeability of gravel soil, medium-grained sand, small-grained sand or peat soil and clay is 100–1000, 10–100, 1–10, 0.1–1 and less than 10^{-4} metres/day, respectively.

The movement of radioactive substances in the soil is determined by the interaction between the soil and the radioactive isotopes, the infiltration rate of precipitation water and the flow speed of the ground water. There is a significant difference between the characteristic migration in the unsaturated zone above the level of ground water and in the saturated zone containing ground water. The migration speed in the unsaturated zone is much slower. The time needed by a radioactive substance to cover

a distance H can be calculated as follows:

$$t = \frac{HK_f}{v},$$

where:

K_f = factor of time transformation;

v = speed of ground water.

The value of K_f is given by the following relationship:

$$K_f = 1 + \frac{1-f}{f} p K_d,$$

where:

f = the porosity of the soil;

p = the density of the soil;

K_d = the quotient of distribution.

The values of f , p and K_d can be measured among laboratory circumstances. The quotient of distribution is the rate of concentration of the radioactive elements in question in the soil and in the solution in contact with the soil, after reaching equilibrium. The equation is:

$$K_d = \frac{(C_0 - C)V}{C \cdot m}$$

where:

C_0 = the specific activity of the solution in the initial state;

C = the specific activity of the solution at equilibrium;

V = volume of the solution;

m = mass of the sorbent (soil).

The value of the quotient of distribution, which refers to the sorptive capacity of the soil, depends on the pH value of solution, the chemical composition as well as the mechanical structure and temperature of the soil. The various radioactive isotopes behave differently from the point of view of their sorption on the soil. For instance, ^{90}Sr and ^{137}Cs usually fix well, ^{106}Ru not so well. The highest value of fixation is in the alkaline range (pH = 10–11); however, it is not negligible even in the acidic range (pH < 7). Maximum ^{90}Sr sorption is at pH = 10, at pH = 5 it is only about 20%. At the same time ruthenium is virtually not fixable in acidic medium, but plutonium is adsorbed well in a wide pH range (pH = 2–9).

There is an inverse relation between the quotient of distribution of radioactive isotopes and the relative migration speed related to the speed of the ground water:

$$V_{\text{rel}} = \frac{V_i}{V_w} = \frac{1}{1 + D},$$

where:

V_{rel} = relative migration speed of the radioactive substance;

V_i = migration speed of the radioactive substance;

V_w = speed of the ground water;

D = a constant characterizing the soil (dimensionless, modified quotient of distribution).

In Hungary a large capacity radioactive waste-processing and storage facility was built in the region of Püspökszilág. The facility constructed under the sponsorship of the State Atomic Energy Committee and operating since 1977 under the supervision of the Sanitarian and Epidemiological Station of the capital, is still able to meet the capacity requirements. The site was selected because of its extraordinarily advantageous pedological situation: the upper 30 m layer of the soil has very low water permeability—it is in 10^{-8} cm/s order of magnitude—and the clay mineral, the montmorillonite content is on the average 24%. This clay mineral plays an important role in the sorption of the radioactive isotopes.

The majority of studies on the radioactive contamination of the soil examine the concentration and distribution of the strontium and cesium isotopes which have primary importance in the buildup of the long-lived radioactive contamination of the soil. Much less is known of the other contaminating radionuclides; however, still there are numerous data in the literature. For example, Bochkarev et al. (1964) communicated data about the migration and distribution in soil of radiocerium. Antipov-Karatayev et al. (1966) discussed the polluting effect of ^{106}Ru , ^{144}Ce and ^{147}Pm radionuclides, their accumulation in the soil and plants. Besides ^{90}Sr and ^{137}Cs , Yudintseva and Gulyakin (1968) also presented information about ^{60}Co , ^{91}Y , ^{95}Zr , ^{106}Ru and their uptake from the soil and their distribution among the different parts of plants. Tihomirov et al. (1981) analysed the behaviour of radioiodine in the soil-plant system, as a function of the genetic properties of the soil.

2.3.3 The relationship between the mechanical structure of the soil and its radioactive contamination

The mechanical structure of the soil has a significant influence on the fixation and migration of radioactive isotopes coming to the surface of earth. In the sorption of strontium, the amount of clay minerals has paramount importance. At higher clay content ^{89}Sr and ^{90}Sr are fixed better, their movement is slower, only a small amount is washed out and settled into the deeper layers of the soil when there is heavy rain. However, the fixation level of strontium on sandy soil as well as its adsorption on the particles of the soil is less, thus its vertical mobility is greater. Among advantageous hydrological circumstances, in the case of soils with light mechanical structure, radiostrontium gradually arrives in the deeper layers of the soil and after a certain period it cannot be reached by plants (Duhamel 1969). Another important factor is soil dispersity. It is related to the humus content, which influences the Ca, Mg and Sr uptake of the plants from the soil. Fine dispersed soils usually have a higher humus content.

The mud fraction of the soil also has an influence on ^{89}Sr and ^{90}Sr absorption. The montmorillonite clay minerals are strong absorbers; micas, iron and aluminium hydroxides and kaolinite may fix strontium to a lesser degree (Gulyakin, Yudintseva 1968). In the case of clay minerals, the degree of sorption is determined mainly by the montmorillonite content. The value of the above defined K_d distribution quotient can be seen in Table 32 as a function of the quartz-sand and clay proportion. It can

be stated that by the increase of the clay content of the sorbent, the distribution quotient in relation to strontium increases significantly.

In the case of sodic soils, mechanical structure plays a less important role in the sorption of fission products because of the specific hydrodynamic and chemical properties of the soil. However, the mechanical structure usually determines the hydrodynamics of a given soil to a great extent, which in turn is related to the movement of radioactive isotopes in the soil (to leaching). Namely, the speed of leaching is determined by the hydrodynamics of the soil (Melnikova, Kudela 1966). In hard soils this process is rather slow because the downward water movement is relatively slow, thus the amount of leached out radioactive substances is smaller.

As we have already mentioned, at the beginning of the 1970s in Hungary an extensive survey was carried out to study the radioactive pollution of different types of soils from various parts of the country. The data of average ^{90}Sr pollution in the most important Hungarian soil types is summarized in Table 33. As can be seen, sandy soil was significantly less polluted than the adobe.

Table 34 shows the vertical distribution of ^{90}Sr as a function of the mechanical structure of the soil (Squire et al. 1958). The data represent the percentage proportion

Table 32 The change of strontium distribution quotient as a function of the quartz-sand/clay proportion in the sorbent

Relative amount of quartz-sand (%)	100	99	98	95	90	80	50	0
Relative amount of clay (%)	0	1	2	5	10	20	50	100
Value of K_d distribution quotient	0.9	1.8	2.7	5.5	10.1	21.7	86.2	207.0

Table 33 Average ^{90}Sr pollution in Hungarian soil types in 1971–1972

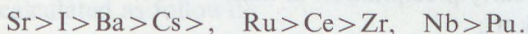
Soil type	^{90}Sr contamination Bq/kg
Sandy soil	4.81
Meadow alluvial soil	5.70
Forest soil	6.40
Meadow soil	6.48
Chernozem	7.73
Solonec	8.95

Table 34 The ^{90}Sr penetration depth (%) in different soil types

Depth (cm)	Clay soil pH=4.6	Sand soil pH=6.6	Lime marly soil pH=7.6
0.0–2.5	50	63	37
2.5–5.0	20	22	24
5.0–7.5	15	13	19
7.5–10.0	10	2	16
10.0–12.5	5	0	3
12.5–15.0	0	0	1

of the amount of isotopes placed on the surface and fixed in the soil layer of a given thickness.

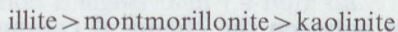
Bochkarev et al. (1964) and Tyuryukanova et al. (1964) have measured the migration of ^{90}Sr and ^{144}Ce in different soils. They have established that both isotopes are practically absorbed in the upper 5 cm layer of soil, on unbroken soils they may arrive at the layers deeper than 20–25 cm only in insignificant amount. Examining the speed of motion of these two isotopes, it was found that cerium moves much slower in the soil than strontium, i.e., it is better absorbed. The rank order of some radioactive elements as important pollutants of the soil is as follows, based on the probability of their uptake by plants (Nishita et al. 1961):



Of course, fixation in the soil is just the reverse of uptake, i.e., Pu is absorbed best and the Sr is absorbed worst. ^{239}Pu is fixed to a great extent on the mineral components of the soil (clay, limestone, etc.), but less so, in soils with high humus content (Nishita, Hamilton 1981). Incidentally, organic substances block the fixation of many radioactive isotopes, e.g. ^{137}Cs , in the soil; thus the plants can more easily take them up in soils with a high organic matter content.

The mobility of ^{137}Cs in the soil is lower than that of ^{90}Sr , therefore it is less readily absorbed by the plants. It is due to the fact that ^{137}Cs binds strongly to the clay minerals with sponge structure (Marckwordt, Lehr 1972). The amount of ^{137}Cs coming from the soil into the plants correlates with the content of organic substances in the soil (McHenry et al. 1973). In soils which have in their upper layers organic substances in high concentration, the cesium uptake may be higher just because the organic substances inhibit the fixation of ^{137}Cs on clay minerals. There is a clear correlation also between the grain diameter of the mechanical fractions of the soil and the amount of ^{137}Cs taken up by the plants from the soil.

Examining the absorption of cesium on clay mineral substances it was found that cesium is so strongly fixed on clay minerals that plants cannot take up much cesium from the soil. However, K^+ ions in high concentration may significantly decrease the sorption of Cs^+ in the soil. According to measurements the sequence of ^{137}Cs absorption is as follows:



In Table 35 the penetration depth of ^{137}Cs in different soil types can be seen (Squire 1958). As in Table 34, the readings of the vertical distribution are expressed as the percentage of the amount put on the surface. Comparing Tables 34 and 35 it is clear that ^{137}Cs does not penetrate into the soil as much as ^{90}Sr does, and independently from the type of soil, only an insignificant amount goes deeper than 5 cm.

According to Gedeonov (1972), the total amount of ^{137}Cs (Q) arriving on the surface of unbroken soil can be given with the following formula:

$$Q = A + B + C,$$

where:

$A = ^{137}\text{Cs}$ activity measured in the 0–1 cm layer of the soil;

B = the amount of ^{137}Cs accumulated by plants;

C = ^{137}Cs activity in the deeper layers of the soil.

According to Gedeonov $A \approx C$, and $B \approx 0.1 A$. Hence,

$$Q \approx 2.1 A.$$

Gedeonov (1972) also reports a relationship between the ^{137}Cs contamination and the amount of precipitation as follows:

$$A = a + b \cdot h,$$

where:

h = amount of yearly precipitation in mm;

$a = 3.3 \cdot 10^8 \text{ Bq/km}^2$;

$b = (2.6 \pm 0.7) 10^6 \text{ Bq/km}^2$.

On the basis of this relationship it can be calculated that the amount of ^{137}Cs effecting the surface by dry fall-out, i.e., by gravitational fall-out without precipitation,

Table 35 The ^{137}Cs penetration depth (%) in different soil types

Depth (cm)	Clay soil pH=4.6	Sand soil pH=6.6	Lime marly soil pH=7.6
0.0— 2.5	98	98	43
2.5— 5.0	2	2	21
5.0— 7.5	0	0	16
7.5—10.0	0	0	15
10.0—12.5	0	0	4
12.5—15.0	0	0	1

is equivalent to the contaminating effect of a yearly precipitation of 130 ± 20 mm. So, the radioactive substances hit the ground surface in arid regions mostly by dry fall-out and in regions with greater amounts of precipitation by wet fall-out.

2.3.4 The relationship between the chemical composition of the soil and its radioactive contamination

As we have already mentioned, the level of contamination of the soil is important for us first of all because of the following question. What is the influence of soil contamination on the plants, i.e., at what rate are the radioactive substances transferred from the soil to the plants and later to the subsequent members of the biological chain? The degree of radioactive contamination of plants with Sr isotopes is influenced among the chemical components of the soil by the calcium content in a decisive way. The reason for it is that the chemical behaviour of strontium is very similar to that of calcium—both being alkaline earth metals—and considering the matter flow from the soil to plants both Ca and Sr form a common transport system. It can be

said that calcium is an inactive carrier of ^{90}Sr . The issues of the relative mobility of Ca and ^{90}Sr in the soil are discussed in detail by Pavlotskaya and Zatsepina (1970).

The concentration of radioactive substances taken up by plants from the soil and later by the animals and humans, changes as a function of the original concentration. In the different metabolic processes so-called discrimination factors can be determined, which actually tell us the change of specific concentration of the radioactive isotope in question, i.e., what proportion of the radioactive pollution could go over to the next member of the biological chain process. The amount of a radioactive substance is always given as a ratio to the amount of an inactive carrier, in the case of ^{90}Sr to Ca. For the passage of ^{90}Sr from the soil into the plants, the plant/soil discrimination factor can be calculated as follows:

$$D = \frac{{}^{90}\text{Sr}/\text{Ca in plants}}{{}^{90}\text{Sr}/\text{Ca in soil}}$$

It can be seen that the ^{90}Sr content of the plants is determined to a great extent by the Ca content of the soil. According to Nadkarni (1974), the average Ca content of the soils is 0.98%. The Ca content of soils with no carbonates is 0.1–1.2%; however, in soils with high carbonate or gypsum (calcium sulphate) content it can be 8–12%, in extreme cases even as high as 20%.

This is the reason why when evaluating the ^{90}Sr contamination level of the soils, the ^{90}Sr concentration expressed as a ratio of the Ca content should also be considered. Eventually in the matter transport from soil to plants it is not the absolute amount of ^{90}Sr , but the relative ^{90}Sr activity given as a fraction of the Ca content which plays a determining role. In Tables 29 and 30 the ^{90}Sr contamination values of the soil are expressed as a ratio to the Ca content, as well.

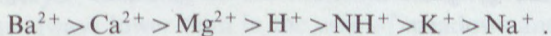
A survey was carried out in Hungary in the early 1970s to determine ^{90}Sr activity expressed as a fraction of Ca content. Based on these findings, Hungarian soils can be classified into 4 groups as follows:

- a) the $^{90}\text{Sr}/\text{Ca}$ proportion is less than 3.7 Bq/g Ca;
- b) the $^{90}\text{Sr}/\text{Ca}$ proportion is between 3.7 and 5.6 Bq/g Ca;
- c) the $^{90}\text{Sr}/\text{Ca}$ proportion is between 5.6 and 7.4 Bq/g Ca;
- d) the $^{90}\text{Sr}/\text{Ca}$ proportion is higher than 7.4 Bq/g Ca.

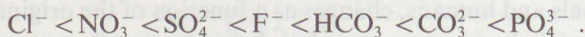
Among the soil samples examined 11.8% belonged to group a, 41.2% to group b, 32.3% to group c and 14.7% to group d with the highest contamination level.

The amount of ^{90}Sr absorbed by plants is influenced even by the pH of the soil, the amount and the chemical composition of the water-soluble salts. The higher the salt content of the soil, the less the ^{90}Sr sorption in the soil (Dergunov et al. 1981).

In the interaction of the radioactive substances and the soil, the inactive salt concentration of the groundwater, or rather the soil solution, plays an important role. The increase of sodium and potassium, or calcium and magnesium decreases the sorption of radioactive isotopes in the soil. For example, the sorption of ^{90}Sr in montmorillonite is inhibited by different cations in the following order:



Considering the sorption of strontium, the value of K_d distribution quotient is determined significantly by the quality of anions in the solution. At identical anion concentrations the value of distribution quotient increases in the following order:



Of course, the amount of radioactive strontium passing on from the soil to the plants is also a function of the inactive strontium content of soils. According to Yuditseva and Gulyakin (1968), the Sr content of different soils has a value between 0.1 and 2.8 g/kg soil. According to Lobanov (1971) and Nadkarni (1974), the average Sr concentration of soils is 0.03% and 214 mg/kg, respectively. As the Sr content is usually higher in peat soils, the amount of ^{90}Sr taken up by plants from such soils is less than in soils with higher mineral content (Ivanov, Shagalova 1969).

The uptake of ^{137}Cs and ^{134}Cs by plants—because of the extremely strong sorbent effect of clay minerals—is much smaller than in the case of radiostrontium, in spite of the fact that the ^{137}Cs contamination of soils is usually higher than that of ^{90}Sr . This phenomenon is highly characteristic for regions which were polluted because of the Chernobyl disaster. The radiostrontium content increased a little, but the pollution with radioactive cesium isotopes increased by 1–2 orders of magnitude. As the cesium is chemically similar to potassium, both being alkaline metals, the discrimination factor in relation to cesium, characteristic for the matter transport from soil to plant, can be described as follows:

$$D = \frac{{}^{137}\text{Cs}/\text{K in plants}}{{}^{137}\text{Cs}/\text{K in soil}}$$

It seems that the Cs uptake from the soil is influenced by the K content of the soil in a decisive way. Incidentally, the K content of the soils is about 1–5%; according to Nadkarni (1974) the mean content is 1.62%. According to the pot experiments of Menzel (1959), the possibility of uptake of univalent cations (^{86}Rb and ^{137}Cs) from the soil is an inverse function of the exchangeable K content of the soil, i.e., the lower the K content, the higher the Cs uptake.

The ^{137}Cs transport from soil to plant is related to the inactive Cs content of the soil. According to the measurements of Fredriksson and Eriksson (1958), the Cs^+ adsorption capacity of the soil is relatively low and the Cs^+ sorption is limited to specific points in the crystal-lattice of the clay minerals. Where these specific points have already been saturated by inactive cesium, the soil can fix only a smaller amount of the contaminating ^{137}Cs , so the amount of radiocesium uptake by plants increases.

This is the reason why, in contrast to strontium, the adsorption of $^{137}\text{Cs}^+$ ions in the soil cannot be considered as a simple exchange adsorption, as in cesium sorption (a physical and chemical sorption) the physico-chemical features of certain clay minerals as well as the characteristics of the crystal-lattice are involved. The average Cs concentration of the soils is according to Lobanov (1971) and Nadkarni (1974) $5 \cdot 10^{-4}\%$ and 5.9 mg/kg, respectively. In the lithosphere the average Cs concentration is 3.2 mg/kg and most of the soils contain 1–10 mg/kg cesium (Fried, Broeshart 1967).

In addition to the K and Cs content of the soil, the ^{137}Cs uptake of the plants is influenced by the ion exchange capacity, the pH and the moisture content of the soil (Magnaval 1973; Hebel 1973). This last factor is connected to the fact that the plants take up ^{137}Cs from solution rather easily. However, this is not so from the soil (Yudintseva, Gulyakin 1968). In fact, during ion uptake from a nutrient solution ion antagonisms, ion competition and the concentration of the solution all have a role; but in the case of soil, solubility is of the greatest importance.

2.3.5 The influence of cultivation and agrochemical methods on the radioactivity of the soil

The means of cultivation in a given agricultural region plays a determining role in the accumulation of radioactive pollution. There is a significant difference in the vertical distribution of the radioactive contamination in unbroken lands, meadows, pastures and regularly cultivated agricultural lands. In uncultivated land the largest part of ^{90}Sr is accumulated in the upper 5 cm layer of soil (Tyuryukanova et al. 1964). The vertical distribution greatly depends on the hydrological circumstances, the mechanical characteristics and genetic type of the soil, i.e., the speed of downward movement of ^{90}Sr is a function of the physical and chemical characteristics of the soil (see Table 34). For instance, chernozem accumulates more ^{90}Sr than meadow soil does, thus the migration in chernozem is smaller (Tyuryukanova, Pavlotskaya 1967). Raikov et al. (1970) studied the vertical distribution of ^{90}Sr in different mountainous soils in Bulgaria. It was found that in the upper 0–3 cm layer the highest level of ^{90}Sr activity was 89 mBq/g soil, in the 3–10 cm layer it was only 26 mBq/g soil.

In arable land the radioactive contamination in the upper 20 cm layer is nearly the same, because of the regular ploughing of the soil, and the activity of the isotopes causing radioactive contamination gradually decreases with depth. According to Bose (1970), in regularly cultivated areas the upper 0–15 cm layer has 1–3 times more ^{90}Sr than the 15–50 cm layer. In the case of heavy soils, only a very small amount of ^{90}Sr penetrates deeper than 50 cm; however, in light soils ^{90}Sr is gradually leached into the deeper layers. The data in Table 28 also demonstrate that there is a significant difference between the radioactive contamination of cultivated and uncultivated land. In cultivated areas in the upper 20–30 cm layer the level of contamination is similar because of mechanical mixing but in uncultivated areas it decreases gradually from the surface with depth. The decrease can be described by an exponential formula:

$$C(x) = C_0 e^{-Kx}$$

where:

$C(x)$ = concentration of the radioactive isotope at depth x ;

C_0 = surface concentration of the radioactive isotope;

K = constant characteristic for the interaction between the radioactive isotope and the soil.

From the data in Table 28 showing the radioactivity of soil samples taken from the upper 0–5 cm soil layer, it is easy to see that in the case of arable lands the ^{90}Sr contamination was always lower than in the case of meadows and pastures.

The information related above is more or less also valid for the other radioactive isotopes coming from the atmosphere to the ground surface. In cultivated lands the distribution of radioactive substances in the upper 20–30 cm layer of soil is almost homogeneous. The contamination of the highest layer in arable fields is smaller than the level of contamination characteristic for uncultivated areas. According to Magnaval (1973), the ^{137}Cs contamination of the upper soil layer in pastures is about three times higher than in arable land.

Hardy et al. (1977) examined the ^{137}Cs , ^{90}Sr , ^{239}Pu , ^{240}Pu , ^{241}Am and ^{238}Pu content of soils and in different plants. The levels of concentration of these isotopes in cultivated land, as a function of depth are shown in Table 36.

The amount of radioactive substances taken up by the plants can be decreased by different agricultural methods, e.g., by liming (i.e., by the increase of Ca content of the soil which decreases the Sr/Ca proportion). Thus the ^{90}Sr uptake can significantly decrease. However, the efficiency will decrease after some time, which is due to the leaching of the agents. At the same time the increase of the salt content of the soil (water-soluble, K, Na, Ca, etc. content) may promote the desorption of radiostrontium fixed in the soil earlier (Dergunov et al. 1981).

By artificial fertilization (addition of large quantity of potassium), decreasing the Cs/K proportion, the ^{137}Cs uptake of plants from soil can be also slowed down. However, this has no practical importance even at a contamination level significantly higher than the present one, because most of ^{137}Cs —perhaps even 95%—is taken up not by the roots but directly through the leaves (Rohleder 1972). In the same way radiostrontium may also be taken up; however, the level of ^{90}Sr contamination in plants is determined usually by the amount of ^{90}Sr taken up from soil.

The flow of the isotopes from the soil to plants is also influenced by the applied artificial fertilizers (Suslova, Nikolayeva 1981). Nitrogen fertilizers influence for example the radiocesium uptake, ammonium (NH_4^+) ions facilitate the flow of ^{137}Cs from the soil to plant. Nitrate (NO_3^-) ions have no such effect. According to Jackson

Table 36 ^{137}Cs , ^{90}Sr , ^{239}Pu , ^{240}Pu , ^{241}Am and ^{238}Pu activity as a function of depth of the soil

Depth of soil (cm)	Radioactivity of soil, mBq/g dry soil				
	^{137}Cs	^{90}Sr	$^{239}\text{Pu} + ^{240}\text{Pu}$	^{241}Am	^{238}Pu
0–5	17.7 ± 1.6	2.01 ± 0.16	0.28 ± 0.02	0.057 ± 0.008	0.008 ± 0.002
5–10	18.3 ± 1.6	1.91 ± 0.16	0.29 ± 0.02	0.049 ± 0.004	0.011 ± 0.002
10–15	15.3 ± 1.7	1.82 ± 0.17	0.24 ± 0.03	0.050 ± 0.004	0.005 ± 0.002
15–20	12.2 ± 1.2	2.65 ± 0.16	0.25 ± 0.02	0.081 ± 0.004	0.010 ± 0.002
20–25	16.5 ± 1.6	2.82 ± 0.32	0.21 ± 0.02	0.056 ± 0.004	0.015 ± 0.003
25–30	9.7 ± 1.2	1.66 ± 0.16	0.13 ± 0.02	0.035 ± 0.003	0.005 ± 0.001
30–35	0.7 ± 0.3	0.33 ± 0.16	0.02 ± 0.02	0.004 ± 0.001	0.001 ± 0.001
35–40	0.3 ± 0.3	0.32 ± 0.16	0.02 ± 0.02	—	0.001 ± 0.001
40–45	—	0.32 ± 0.16	0.02 ± 0.02	—	0.001 ± 0.001
45–50	—	0.32 ± 0.16	0.02 ± 0.02	—	0.001 ± 0.001
Average 0–30	14.9 ± 3.3	2.16 ± 0.49	0.23 ± 0.06	0.057 ± 0.016	0.009 ± 0.004

et al. (1968), the presence of much nitrogen promotes Sr uptake. The phosphate fertilizers inhibit the accumulation of ^{90}Sr since strontium with phosphate makes strontiumphosphate, which is hardly soluble in water, thus only a small amount of strontium reaches the plants (Prohorov et al. 1970). The ^{90}Sr uptake of pea and wheat can be moderated not only by phosphate but also by adding sodium silicate to the soil (Moiseyev et al. 1969).

The ^{90}Sr uptake of plants decreases with increasing pH of the soil, because the increased alkalinity results in most of the radiostrontium being in the form of SrCO_3 which is hardly soluble in water. Ivanov et al. (1970) demonstrated a close negative correlation between the pH value of the soil and the ^{90}Sr content measured in plants.

In certain cases the radioactive substances can be leached into the deeper layers of the soil through inundation. This is possible mostly in soils with light mechanical structure. This method is useful when an agriculturally important area is polluted (for instance, because of a nuclear reactor disaster) and the soil must be decontaminated depending on the pedological characteristics. Thus sodic (high Na content) soils, for example, can be leached with water. The ^{90}Sr content of the soil can also be diminished by CaCl_2 and KCl solutions, often used for melioration (Dergunov et al. 1981). Through the leaching of the soil the radioactive isotopes penetrate deeper than the root zone, i.e., they arrive at a depth from where the plants are unable to take them up. However, under natural conditions (normal amount of precipitation) this downward vertical migration is very slight. According to the statement of Marey (1980), the yearly amount of ^{90}Sr and ^{137}Cs going out of root zone is a maximum of 2.5 and 0.7%, respectively.

With artificial fertilizers a significant amount of radioactive substance of natural origin is added to the soil: in addition to ^{40}K , among others uranium, thorium, ^{226}Ra , ^{210}Pb and ^{210}Po (Menzel 1968; Tso et al. 1968; Drichko et al. 1975; Pauly et al. 1976; Philipp et al. 1979). Among the artificial fertilizers the activity of uranium, or rather its daughter elements, is sometimes extremely high. The uranium content in raw phosphates coming from sea deposits (e.g. in Florida) is sometimes so high that it is extracted through the processing (Pfister et al. 1975). Table 37 shows the ^{226}Ra , ^{210}Pb and ^{210}Po contents in certain types of artificial fertilizers (Yermolayeva-Makovskaya, Litver 1978), while Table 38 summarizes the uranium,

Table 37 ^{226}Ra , ^{210}Pb and ^{210}Po content of artificial fertilizers

Sample	Provenance	Activity, Bq/g		
		^{226}Ra	^{210}Pb	^{210}Po
Phosphorite flour	Voskresensk	0.48 ± 0.05	0.33 ± 0.02	0.49 ± 0.04
Double superphosphate	Voskresensk	0.13 ± 0.02	0.12 ± 0.01	0.14 ± 0.01
Apatite flour	Voskresensk	0.03 ± 0.01	0.03 ± 0.01	0.03 ± 0.01
Double superphosphate	Uvarovsk	0.64 ± 0.07	0.67 ± 0.04	0.83 ± 0.07
Ammophos	Nevinnomisk	0.10 ± 0.02	0.09 ± 0.01	0.11 ± 0.01
Nitro-ammophos	Nevinnomisk	0.02 ± 0.01	0.02 ± 0.01	0.02 ± 0.01

Table 38 Average radioactivity of phosphate fertilizers applied in the FRG in 1973–1974

Type of phosphate fertilizer	Proportion of use (%)	Activity, Bq/g P ₂ O ₅			
		U	²²⁶ Ra	Th	⁴⁰ K
Superphosphate	1.9	2.87	2.89	0.09	0.75
Thomas slag	28.0	0.04	0.04	0.01	0.02
PK fertilizer	25.3	2.50	2.28	0.08	37.42
NP fertilizer	5.5	3.28	1.12	0.10	0.15
NPK fertilizer	31.3	3.42	2.04	0.11	38.64
Other	8.0	2.38	1.72	0.09	0.40
Total of phosphate fertilizers	100.0	2.14	1.48	0.07	21.62

²²⁶Ra, thorium and ⁴⁰K activities of phosphate fertilizers used in FRG in 1973–1974 (Pfister et al. 1976).

The natural radioactivity of the soil increases significantly with the application of artificial fertilization. For instance, 1000 kg of artificial phosphate fertilizer applied to 1 ha area in the USA doubled the natural radioactivity of the soil as regards the heavy radionuclides (Grashenko et al. 1982). Actually the accumulation of the natural radionuclides in the soil can be described as a double process: on the one hand the radioactive substances are added to the soil with the fertilizers, on the other hand they are removed from the soil with the harvesting of the crop. The following equation can be stated:

$$\frac{dA}{dt} = A_m - \lambda A,$$

where:

A = the amount of the radionuclide in question in an area of 1 ha;

A_m = the amount of the radionuclide in question added to an area of 1 ha through artificial fertilization;

λ = the portion of radionuclide contamination in an area of 1 ha, which is removed by harvesting of the crop during one year. The value of λ is about 10^{-2} /year.

Solving the equation, the product shows that the accumulation of radionuclides has an exponential character as a function of time. Incidentally, investigations show that the uranium, thorium and radium content of dry plant materials (e.g., wheat, barley, rice, pea) is about 10^{-2} – 10^{-4} times that of the soil (Grashenko et al. 1982).

Last, but not least, we have to mention that in addition to the different agrotechnical and agrochemical methods, even the amount and the species of the produced plants have an influence on the radioactive contamination of the soil. On the one hand the spatial movement of the radioactive substances is influenced by the plants mostly because they have an important role in the evapotranspiration of water. Thus they have an influence on the water movement, which determines the horizontal and vertical movement of the water-soluble radioactive substances. On the other hand the plants are able to take up the radioactive substances through their roots, and trans-

port them; thus they redistribute them. Just to have a picture, the roots of tall trees are 5–7 metres deep. Fruit trees, grapes and alfalfa have 3–4 metres long roots. Sugar beet, sunflower, red clover, hemp and maize have 2–3 m deep roots, the cereals, rape, pea, bean, potato and vegetables have 1–2 m deep roots and only the roots of Gramineae, onion, lettuce etc., are shorter than 1 metre long. It is worth mentioning that inter-varietal accumulation differences within species may be as great as interspecific differences.

The radioactive substances taken up by plants from cultivated land are removed by harvest. Of course, this amount depends on the species of plant and its yield as well as on pedological and meteorological parameters. It is a matter of fact that the harvest decreases the radioactive contamination of the soil; however, this effect, considering the total amount of radioactive isotopes in the soil, is usually not too significant. Bose (1970) published detailed data, as regards the different food and fodder plants, of the proportion of the amount of ^{90}Sr being in the soil and taken up by the plants. According to the investigations carried out between 1966 and 1968, over 3 years, the yearly amount of ^{90}Sr removed by harvesting was only about 0.1–0.9% of the total ^{90}Sr content in the soil. It should be mentioned that a certain part of this amount is returned to the soil as organic manure. However, from the above-mentioned facts it is evident that the amount of radioactive isotopes reaching the soil in the form of organic manure is insignificant relative to the total amount of radioactive substances within the soil in the root zone.

In summary we state that in the decontamination of soil from the radioactive substances the following routes have to be taken into account: decay of the radioactive isotope, leaching from the soil, flow-away on the surface of the soil, radionuclide uptake of the plants and soil erosion. These factors, or rather their relative efficiency, are discussed in detail, for example, by Haghiri (1969).

2.3.6 Summary

In this subchapter the formation of the natural and man-made radioactivity of the soil is explained. The relationships between the radioactive contamination and the mechanical structure of the soil and its chemical characteristics, as well as the agrotechnical and agrochemical methods used, are discussed. The more important points are as follows:

1. The natural radioactivity of the soil depends on geological circumstances. However, man-made radioactive pollution due to atmospheric nuclear tests, i.e. radioactive contamination, is determined mostly by geographical and orographic factors (e.g. the relief of terrain). Except in desert and semi-desert areas, the bulk of the radioactive isotopes arrives from the atmosphere carried by precipitation (rain-out, wash-out) to the surface and only a smaller part is deposited by dry-out. As a consequence of the Chernobyl reactor disaster the contamination of the soil increased in most parts of Europe. In 1986 the ^{137}Cs and ^{134}Cs contamination of the soils increased by 1–2 orders of magnitude in several countries, as compared to earlier years.

2. In the sorption of the radionuclides reaching the soil surface as well as in their

migration towards the deeper layers of the soil, the mechanical structure, the distribution of grain diameter and the chemical composition of the soil all play an important role. In light soils with loose mechanical structure (e.g., sand), the speed of migration of the radioactive substances is higher than in heavy soils (e.g., clay), where the radioactive substances seep down towards the deeper layers much more slowly and they remain even after several years within the upper few centimetre section of the soil.

3. The long-lasting radioactive pollution of soil is largely due to the activity of ^{90}Sr and ^{137}Cs isotopes. The proportion of the ^{137}Cs : ^{90}Sr activities is usually between 1 and 2. However, this proportion is not characteristic for the areas contaminated as a consequence of the Chernobyl disaster, where, as we have mentioned earlier, the ^{137}Cs contamination increased significantly, but the radiostrontium contamination only slightly increased. Thus in 1986 in numerous European countries the proportion of ^{137}Cs and ^{90}Sr activity attained a value between 10 and 100. In addition to the radiocesium and radiostrontium isotopes, several other radionuclides, e.g. ^{95}Zr , ^{106}Ru , ^{140}Ba and ^{139}Pu , contaminate the soils; however, they are adsorbed so much onto the soil particles that only an insignificant portion of those isotopes may be incorporated into the plants.

4. There is a significant difference between the vertical distribution of the radioactive contamination of the soil in cultivated and uncultivated areas. In meadows and pastures the radioactive isotopes arriving on the soil from the atmosphere by dry-out or brought down by the precipitation, accumulate in the upper layers of the soil and their concentration decreases exponentially as a function of depth. Incidentally ^{90}Sr moves faster than ^{137}Cs . At the same time in arable lands the level of contamination is nearly constant in the upper 20–30 cm layers, due to the mechanical mixing by ploughing.

5. The applied methods of soil melioration have an influence on the radioactivity of the soil, as well as on the amount of radioactive substances taken up by the plants. On the one hand, in loose-structured soils with inundation we can leach the radioactive substances into the deeper soil layers. On the other hand, liming, i.e., decreasing the Sr:Ca proportion in the soil causes a decrease in the amount of radioactive strontium taken up by the plants. At the same time the application of artificial fertilizers may increase the radioactivity of the soil, as the phosphates are rather rich in uranium as well as in natural radioactive elements belonging to its decay sequence.

2.4 Radioactivity of plants and influencing factors

The radioactivity of the phytosphere, i.e., of plants, is formed by continuous and dynamic interactions with the other members (the hydro-, litho- and atmospheres) of the biosphere discussed earlier in detail, i.e., with the water, the soil and the air. Among them, in most cases, the soil plays a dominant role, as the mineral material from which the plants build up their bodies, is mostly taken up from the soil through the roots. Thus in the first place the physico-chemical parameters of the soil determine the amount of radioactive isotopes accumulated in the plants. Of course, as we have

discussed in detail in the previous chapter, the plants may also modify the concentrations and the proportions of the radioactive isotopes in the soil.

However, radioactive substances may arrive in the plant tissues not only from the soil but also directly from the air through the leaves. For instance, the ^{14}C isotope is taken up by plants in the form of CO_2 through the leaves and is assimilated by the process of photosynthesis into their tissues mainly in the form of carbohydrates. Matter transport through the leaves also plays a decisive role in ^{137}Cs contamination, being similar to the ion uptake of cells in the leaves and roots with identical kinetic constants and ion selectivity. Using the method of isotope indication, it has been demonstrated that all the elements going from the soil through the roots into the plant may also directly enter into the plant through the leaves.

A relatively significant amount of radioactive substances may accumulate in the tissues of a plant without any apparent developmental or other malformation. Plants are able to accumulate in them such large quantities of radioactive substances without any apparent or detectable malfunction, which renders their consumption by an animal (or human) hazardous. In some way the situation is similar to the case of many micro-elements, where, due to the differences between the plant and animal physiological processes, e.g., a concentration of boron which is far from toxic for plants, for animal or human organisms may cause serious illness (boron-enteritis) manifested by clinical symptoms.

According to the survey made by Yudintseva and Gulyakin (1968), there is no observable change in plant organisms when the ^{90}Sr and ^{137}Cs content of the tissues is less than 30–40 Bq/g. Just to give a proper picture we have to mention that this value of activity is higher by 2–3 orders of magnitude than the ^{90}Sr and ^{137}Cs activities measurable in the present contamination situation, and it is also higher by 1–2 orders of magnitude than the gross β -activity (see also the data in Tables 39, 41, 42, 50 and 51).

Of course, a very high dosage (inhibition range) has a negative influence on the development and growth of plant. In such cases the yield decreases, the influence of radiation causes disturbances in the carbohydrate and nitrogen metabolism, e.g., the proportion of non-protein nitrogen increases compared to the total nitrogen content. There are significant differences among some species as regards radiosensitivity, e.g., wheat is much more sensitive than oats. Incidentally, the reproductive (generative) organs of plants are much more sensitive than the vegetative parts. We should mention that perhaps before the range of inhibition (bionegative influence) there is a low-dose stimulation (biopositive influence) range. This topic will be discussed later in relation to the Chernobyl disaster in Chapter 4.

The radioactivity of the vegetation, just as with the other components of the environment mentioned earlier, is from two sources. Part of the radioactivity is of natural origin (depending on the geological circumstances of the area, as well as the radionuclides produced by nuclear weapon tests and nuclear facilities (power plants, isotope laboratories) which contaminate the environment. The radioactivity of the plant tissues (including agricultural products and food) is determined by several factors. The radioactivity of a plant is the result of close interactions with its environment as well as its metabolic processes which in turn are influenced by such ecological

factors as climatic and meteorological parameters, water supply, temperature, mode of cultivation, etc. Temperature and the number of hours of sunshine influence the metabolic processes of the plants, which in turn correlates with the uptake and accumulation of radioactive isotopes. Ca metabolism, together with the radiostrontium uptake of the plants, is influenced by the light intensity. Within given limits of illumination, the Ca content expressed in terms of dry material grows proportionally.

In this subchapter the buildup of radioactivity in plants (radionuclides of both natural and artificial origin) is discussed. The factors (e.g., vegetation period, species of plant) which determine the degree of radioactive contamination of the soil-plant system will also be dealt with. Eventually the methods by which the radiation pollution of agricultural products and vegetable food can be decreased will be explained.

Knowing the discrimination factors between the soil and the plants based on the soil activity, we are able to estimate the expectable concentration of the different radionuclides in plant tissues. However, knowing the level of radioactivity in plants is not important in itself. Its significance lies in the fact that the radioactivity of plants chiefly determines the radioactive contamination of the further elements (animals and man) of the biological chain. To be specific, the level of radioactivity measurable in animal and human organisms is due almost exclusively to the radioactivity of vegetable foods and fodders. The amount of radioactive substances taken in by the organisms with drinking water and through inhalation is usually significantly smaller. This is the reason why under given circumstances the processes, methods and technologies of decontamination decreasing the level of radioactive contamination of vegetable foods and agricultural products are of paramount importance.

2.4.1 Natural radioactivity of plants

Among the naturally radioactive elements occurring in plants, potassium, or rather the ^{40}K isotope found in natural potassium with a relative rate of 0.0119%, should be mentioned. As potassium is an essential macroelement for plants, all plants contain this element. Potassium is taken up from the soil; the degree of uptake is determined in some way by the water-soluble K^+ concentration of the soil or rather its exchangeable K-capacity. Plants, depending on their species, contain potassium only between relatively narrow limits of concentration, just as in the case of the other essential macroelements. However, this statement is not valid for the trace elements (mostly the non-essential microelements like Cs). The same plant species may accumulate quantities of the same microelement different by orders of magnitude, depending on environmental factors.

Under the present contamination circumstances (not considering 1986, the year of the Chernobyl disaster), the activity of ^{40}K is responsible for most plant radioactivity, i.e., the share of activity of the other isotopes causing contamination is significantly smaller. This statement is supported by data in Table 39, which summarizes the results of a survey carried out in Hungary in 1980 by the Radiological Control Network of the Ministry of Agriculture and Food, during the course of which the

so-called indicator plants (spinach, lettuce and sorrel) were examined. It can be seen from the table that about 80–90% of the gross β -activity is due to the activity of the ^{40}K isotope. Incidentally, the metal ion-fraction activity indicating radiostrontium contamination of the sample actually contains the metals of the I–IV groups. We have to note that, although the measurement of metal ion-fraction activity provides a quick way of obtaining information about the level of artificial radioactive contamination, this technique is not nuclide-specific. With this method it is not possible to determine the activity and the effect of radiation load of each and every contaminating isotope. For this reason it is necessary to carry out a nuclide-specific analysis. The table also shows that the average standard deviation of the K-concentration in the samples was $\pm 22.4\%$, i.e., the K-content of the different plants varied only within these limits due to the changes of the environmental factors. Actually, this variation determines even the 25.1% average standard deviation of the gross β -activity.

Every plant, like all living creatures, contains carbon, and in this way also contains the radioactive ^{14}C isotope. Knowing the carbon content of a plant and the $^{14}\text{C}/^{12}\text{C}$ proportion at equilibrium, the specific ^{14}C activity of each plant can be determined. Plants take up the ^{14}C isotope in the form of CO_2 from the air, thus they are good indicators of the concentration changes of ^{14}C in the atmosphere (Wilson 1961). The ^{14}C isotope is built into the plants' organic materials by photosynthesis. Trees accumulate ^{14}C in the cellulose of their annual growth rings, therefore it is possible to obtain information about the given years and about the average atmospheric concentration of ^{14}C characteristic for the vegetation period by examining the annual rings. It must be pointed out, however, that the ^{14}C activity of plants gives a reliable picture of the actual atmospheric $^{14}\text{C}/^{12}\text{C}$ proportion only if there was no important CO_2 source in the immediate environment of the sample examined. Near to busy motorways or thermal power plants the CO_2 content of air is poorer in the ^{14}C isotope because during the combustion of fossil fuels, practically ^{14}C -free CO_2 is produced. Incidentally, as a consequence of the so-called isotope-effect, the $^{12}\text{C}/^{13}\text{C}/^{14}\text{C}$ proportion during the process of photosynthesis does not remain perfectly the same, the isotopes with lower mass are accumulated in a somewhat higher proportion.

Table 39 Summary of results of investigation of spinach, lettuce and sorrel samples carried out by the RCN in 1980

Sample	Number of samples	Season	Activity, mBq/g dry material			^{40}K : gross β -activity (%)
			gross β -activity	^{40}K	metal ion-fraction	
Spinach	56	Spring	2304 ± 579	1994 ± 579	52 ± 29	87
	63	Autumn	2442 ± 693	2017 ± 531	58 ± 41	83
Lettuce	53	Spring	2208 ± 665	1881 ± 339	45 ± 23	85
	54	Autumn	2102 ± 351	1723 ± 273	54 ± 24	82
Sorrel	52	Spring	1560 ± 390	1413 ± 370	42 ± 23	91
	63	Autumn	1712 ± 428	1394 ± 311	52 ± 34	81
Average coefficient of variation (%)			25.1	22.4	57.0	

In Hungary the atmospheric (and thus biospheric) ^{14}C concentration is continuously decreasing, as shown by the examinations of annual rings and leaves of trees growing in the vicinity of the city Debrecen and the town of Paks. In 1980, the ^{14}C contamination due to the atmospheric nuclear weapon tests and the emissions of nuclear reactors represented only 27% of the natural ^{14}C level produced by cosmic radiation. Incidentally, most of the ^{14}C content of the air, through exchange processes, comes into the hydrosphere, and it occurs mostly in the seawater in dissolved form. The phytosphere contains only about the 3.8% of the total ^{14}C resource of the Earth, i.e., the ^{14}C activity accumulated within the plants is about $3.1 \cdot 10^8$ GBq, which is equivalent to a mass of 1.9 kg (Fairhall, Young 1970).

In addition to the ^{40}K and ^{14}C isotopes, plants contain other natural radioactive isotopes as well, taken up from the soil or immediately from the air through the leaves. According to a communication by Belova and Lunkina (1967), the ^{226}Ra activity of vegetables is around 0.1–0.4 Bq/kg, their uranium content is between $1.9 \cdot 10^{-6}$ and $2.6 \cdot 10^{-6}$ g/kg; Homma and Murakami (1977) a ^{226}Ra activity of 0.4–0.6 Bq/kg measured in different plants.

Table 40 gives data on the ^{210}Pb and ^{210}Po content of various plants (Yermolaeva-Makovskaya, Litver 1978). It can be seen that the $^{210}\text{Pb}/^{210}\text{Po}$ proportion is different in the different plants. In plants of short vegetation period the $^{210}\text{Pb}/^{210}\text{Po}$ proportion is relatively low—there is no time for the formation of a radioactive equilibrium. However, in perennial plants the proportion approaches or reaches the value of 1.

Table 40 ^{210}Pb and ^{210}Po activity of plants

Plant	Activity, Bq/kg		$^{210}\text{Po} : ^{210}\text{Pb}$
	^{210}Pb	^{210}Po	
Trefoil	30.7	8.5	0.28
Willow leaf	33.3	7.8	0.23
Birch leaf	42.2	11.8	0.28
Grass	4.7	1.9	0.41
Roots of grass	30.3	34.7	1.14
Soviet cigarette	32.6	18.5	0.57
Bulgarian cigarette	22.6	16.7	0.74
Korean cigarette	91.0	82.1	0.90

The majority of the α -activity of the plants, about 80–90%, is due to the activity of ^{210}Pb , and its daughter element ^{210}Po (Mayneord et al. 1960; Hill 1960; Frindik 1963). It could be detected that in grasses the specific activity of ^{210}Pb relative to that of the natural Pb isotopes was almost 20 times higher than the specific activity of ^{210}Pb measurable in the soil. It means that, on the one hand, there is no correlation between the Pb concentration in the plants and in the soil, respectively. On the other hand, during the buildup of the concentration of the ^{210}Pb , ^{210}Bi and ^{210}Po isotopes in plants, the uptake through the roots is usually of secondary importance, because the majority of the radionuclides enter the plant tissues through the

leaves (Burton, Stewart 1960; Berger et al. 1965; Chadwick, Chamberlain 1970; D'Souza, Mistry 1970; Watters, Hansen 1970). Incidentally, the mechanical structure of the soil strongly influences ^{210}Pb and ^{210}Po uptake through the roots. In sandy soils the lead and polonium uptake through the roots may be significant (Volkova et al. 1966). Among the various plants, the ^{210}Po content of tobacco has been the most thoroughly investigated, because during smoking most of the polonium is inhaled (Tso et al. 1964, 1966, 1968; Nikolova 1972; Pyrkayastha, Bhattacharyya 1975).

A small part of ^{210}Po in the plants is produced by the decay of the ^{226}Ra content of the plant. Examining different varieties of tobacco, Tso et al. (1966) have determined that this amount is smaller by 3 orders of magnitude than the total amount of ^{210}Pb .

According to investigations carried out in Hungary, the ^{210}Po activity of tobaccos and cigarettes is 11–30 mBq/g dry matter; the ^{210}Po activity of tinned foods is about 11–45 mBq/kg.

A relatively significant amount of ^{210}Pb and ^{210}Po is accumulated in aquatic plants. For instance, the ^{210}Pb activity of different organisms living in the sea is about 10^4 times higher than in the same amount of seawater (Kauranen, Miettinen 1971). The ^{210}Pb and ^{210}Po activity of planktons is 18–74 and 51–204 Bq/kg, respectively (Hursh et al. 1965). Incidentally, the ^{210}Pb and ^{210}Po activities of phytoplankton are lower than those of zooplankton (Shannon, Cherry 1967).

The ^{87}Rb isotope occurs in plants too. Rubidium is chemically very similar to potassium, it belongs to the alkaline metals. Rubidium is not an essential microelement for the plants, which contain only quantities of mg/kg order of magnitude of this element. For instance, the rubidium concentration of potato powder, used as standard reference material in activation analysis (IAEA V-4 sample), is 6.2 ± 0.2 mg/kg (Schelenz 1977). Because of the long half-life of ^{87}Rb , its specific activity has a relatively low value. The ^{87}Rb activity represents only a very small part of the radioactivity of plants (a maximum of few per cent).

2.4.2 Radioactive contamination of plants

An important, sometimes practically the only factor responsible for the radioactive contamination of plants is the presence of radioactive isotopes of strontium and cesium. Among the radioactive isotopes of strontium, ^{90}Sr , and, among the radioactive cesium isotopes, ^{137}Cs , may cause long-lasting contamination. Of course, besides radioactive strontium and cesium, plant tissues also contain the stable isotopes of these elements, and their uptake is influenced by the presence and concentration of the inactive isotopes too. It is worth taking a closer look at strontium and cesium, these typical microelements.

As we mentioned earlier, strontium does not belong to the very rare elements, the average Sr content of soils being about $3 \cdot 10^{-2}\%$. The physiological role of strontium in plants is not clear; it seems that it is an essential microelement for many species of plants. For instance, it was found that after sprinkling the leaves of peach trees

suffering from chlorosis with dilute solution of strontium chloride, the symptoms disappeared. Incidentally, the analysis of these leaves demonstrated that the strontium content in their ashes was about one order of magnitude smaller than its concentration in healthy peach tree leaves. According to a communication by Kolo-miytseva and Gabovich (1970), the strontium content of vegetables and green plants is about 4–80 mg/kg of fresh, original material. According to Kovalevsky (1969) the Ca : Sr proportion of plant ashes is 80 : 1; on the average, however, in soils with high Sr content it may decrease to below 10 : 1.

According to our measurement (Szabó 1979, 1980, 1984), the Sr content in the different investigated plants (sorrel, spinach, lettuce, lucerne, bean, wheat, maize) had a value of 27–82 mg/100 g dry material. Under natural circumstances the Sr concentration in plants is about 1–4% of their Ca concentration. However, in the case of plants growing on soils with high Sr content, the Sr : Ca proportion was about 10–18%. It means that Sr, having a very similar characteristics to Ca, may substitute a relatively high proportion of the Ca content of the plants.

Strontium in suitably dilute solution is able to stimulate the growth of certain plants; for example, it has a beneficial effect on the growth of wheat and maize. The yield of bean may be increased significantly by spray-fertilization with Sr. However, above a certain concentration level strontium decreases the growth of the plant. According to our investigations carried out with wheat, maize and bean, a concentration of 1 g Sr/100 g soil has a strong toxic effect on the growth of all the three plants. In case of wheat and maize, the phytotoxic effect could already be detected above a soil concentration of 0.1 g Sr/100 g; however, it seems that bean is less sensitive to a relatively high Sr content.

As Sr and Ca have a common transport system, the amount of Sr, or rather ^{90}Sr taken up from the soil, is a function of the exchangeable Ca content of the soil. When the absorbable Ca content of the soil is low, the amount of ^{90}Sr taken up from the soil may be much higher than the ^{90}Sr contamination of plants grown on soils with high Ca content. Actually, the plants may be classified on the basis of their relation to calcium in two groups: there are calciphilous (fond of calcium) and calciphobic (calcium avoiding) plants. Among the higher plants there are relatively few calciphobic ones, such as lupin, chestnut and the peaty bog plants. Among the definitely calciphilous plants the family Papilionaceae are the best known; usually they also need much phosphorus. Expressed in dry matter, the radiostrontium contamination of calciphilous plants is usually significantly higher than that of the calciphobic plants, which is due to the clearly higher Ca content. The pH of the soil also influences the affinity of the plants to calcium, as well as the degree of its uptake.

The concentration of cesium in nature is much lower than that of strontium. As mentioned earlier, only about $5 \cdot 10^{-4}\%$ can be considered as the average cesium content of the soil. As far as we know, cesium is not an essential microelement for plants. As with strontium, the growth of certain plants can be stimulated with very dilute CsCl solution, however, in higher concentrations (> 1 g Cs/kg soil) it has a negative influence on growth.

In plant tissues Cs usually occurs in concentrations lower than 0.1 mg/kg (Nadkar-ni 1977). According to our photometric measurements, the occurrence of Cs in the

plant ash is around 0.1 mg/g, thus the Cs concentration is about one order of magnitude lower than the strontium concentration. Cs added to the soil in a concentration of 100 mg/100 g of soil demonstrated a toxic effect in the investigated plants (bean, wheat, maize). In spite of the fact that cesium is taken up by plants only in a very small amount, there is already a phytotoxic effect also in the case of a relatively low Cs content of soil. It means that Ca cannot replace potassium at all.

We have many data on the radioactive contamination of plants, both cultivated and uncultivated. In almost every country of the world, stations for measuring radioactivity were established after the beginning of the nuclear weapon tests, where the concentrations of the most important contaminating components are continuously monitored. In Hungary, systematic measurements started in 1959.

Of course most data refer to the concentration of the ^{90}Sr and ^{137}Cs isotopes which play a decisive role in long-lasting radioactive contamination. Based on the data of Marey (1980), Table 41 summarizes the results of measurements carried out on certain plants in the Soviet Union between 1963 and 1973. Looking at the data in the table, we can establish that in the plant samples investigated the ^{137}Cs content was higher than the ^{90}Sr concentration. According to the measurements of Bose (1970) performed between 1965 and 1968, the $^{137}\text{Cs}/^{90}\text{Sr}$ proportion in grass, potato, rye and wheat was 1.0–1.4, 1.1–3.3, 1.8–5.2 and 1.3–3.7, respectively.

The data in Table 41 also substantiate that after the contamination maxima in the years 1963 and 1964, the degree of radioactive pollution has decreased. This decrease is due without any doubt to the Moscow Nuclear Ban Treaty (1963). The radioactive contamination of the atmosphere, the surface waters and the soil reached its peak between 1961 and 1963; the maximum contamination in plants was recorded between 1961 and 1964 and that in animal and human organisms between 1963 and 1965, respectively (Sivintsev et al. 1966; Scheel et al. 1967; Miserez 1967; Ocker 1968; Gedeonov et al. 1975; UNSCEAR 1977).

During the period of maximum contamination, the activity of the metal ion-fraction characteristic for radiostrontium pollution was of the same order of magnitude as the natural activity, and in certain cases it was significantly greater than the natural one. In order to be able to compare the data given in Table 39, we have to mention that, for example, the average metal ion-fraction activity of lettuce in the autumn period of 1961 in Hungary was 2.07 Bq/g dry material, and the average activity of the metal ion-fraction of spinach in the spring of 1963 was 1.58 Bq/g dry material. In the years 1963–1964, in certain samples the measured activity of metal ion-fraction was as high as 4 Bq/g dry material. These values were about one order of magnitude higher than the present level of contamination.

The degree of radioactive contamination measured in plant samples after the peak between 1961 and 1964 has decreased significantly and actually it has fluctuated from the end of 1960s until 1985 around a relatively constant low level of pollution. The smaller or greater differences among the various years were due, in the first place, to the different meteorological circumstances, as well as to the different soil conditions of the sample sites. However, at the end of 1976, as a consequence of a nuclear weapon test performed in China in September, 1976, another transitional peak in contamination level was observed in the atmosphere, and its influence could also be observed in

Table 41 ^{90}Sr and ^{137}Cs contamination of different plants in the Soviet Union between 1963 and 1973

Year	Radioactive contamination, Bq/kg							
	Potato		Cabbage		Tomato		Apple	
	⁹⁰ Sr	¹³⁷ Cs	⁹⁰ Sr	¹³⁷ Cs	⁹⁰ Sr	¹³⁷ Cs	⁹⁰ Sr	¹³⁷ Cs
1963	0.18±0.01	2.33±0.37	0.41±0.05	2.04±0.30	0.67±0.11	—	0.89±0.11	—
1964	0.43±0.03	4.07±0.26	0.56±0.04	2.59±0.19	0.41±0.05	1.48±0.19	0.67±0.15	2.81±0.67
1965	0.33±0.02	2.52±0.11	0.47±0.02	2.11±0.09	0.29±0.04	1.04±0.15	0.33±0.03	2.00±0.19
1966	0.28±0.02	1.26±0.07	0.46±0.03	1.18±0.07	0.22±0.01	0.83±0.41	0.17±0.01	0.83±0.06
1967	0.25±0.01	0.93±0.03	0.43±0.06	0.85±0.11	0.21±0.01	0.70±0.04	0.15±0.01	0.65±0.03
1968	0.25±0.01	0.65±0.05	0.38±0.01	0.48±0.03	0.17±0.01	0.33±0.02	0.19±0.01	0.37±0.03
1969	0.22±0.01	0.60±0.04	0.29±0.03	0.44±0.05	—	0.36±0.04	0.20±0.03	0.34±0.04
1970	0.27±0.01	0.65±0.10	—	0.43±0.04	—	—	—	—
1971	0.26±0.01	0.56±0.06	—	0.34±0.04	—	—	0.28±0.01	0.41±0.03
1972	0.21±0.03	0.40±0.03	0.55±0.09	0.50±0.06	0.20±0.04	0.26±0.09	0.10±0.03	0.20±0.07
1973	0.23±0.02	0.40±0.03	0.48±0.05	0.37±0.07	0.26±0.03	0.33±0.01	0.17±0.03	0.21±0.09

Table 42 Average radioactivity of sorrel samples taken in Győr-Sopron County between 1968 and 1980

Year	Activity, mBq/g dry material			Metal ion-fraction activity mBq/g Ca
	Gross β -activity	^{40}K	Metal ion-fraction	
1968	1795	1536	66	4 218
1969	1647	1521	78	4 107
1970	1765	1576	159	10 360
1971	1173	932	74	5 513
1972	1735	1476	118	10 027
1973	1695	1369	59	4 218
1974	1384	1236	70	4 366
1975	1798	1628	70	5 291
1976	1336	1003	56	8 991
1977	1854	1565	155	14 689
1978	1780	1173	184	10 915
1979	1840	1644	64	5 356
1980	2230	1940	59	3 941
Average	1694 $\pm$ 267	1431 $\pm$ 281	88 $\pm$ 45	7071 $\pm$ 3831
CV (in %)	15.8	19.6	51.1	54.2

plants. This is also demonstrated by the data in Table 42, as the activity of metal ion-fraction referred to the Ca content between 1968 and 1980, in 1977 reached its peak. However, the contamination level of ^{90}Sr and ^{137}Cs causing long-lasting pollution did not increase significantly, because the activity of the radioactive isotopes with short half-lives (e.g., ^{89}Sr , ^{140}Ba , ^{131}I , ^{95}Zr) dominate in the relatively rapid (early) fall-out. The data in Table 42 also demonstrate that, considering the present contamination situation, most of the gross β -activity is caused by the activity of the ^{40}K isotope. It can also be seen that the standard deviation of the gross β -activity (15.8%) is determined mainly by the standard deviation of ^{40}K (19.6%). The activity of the metal ion-fraction has a value about an order of magnitude smaller than the activity of ^{40}K . In spite of fact that its coefficient of variation is rather high ($> 50\%$); thus it does not have a strong influence on the yearly changes of the gross β -activity.

In Figure 21, the average metal ion activity of sorrel, lettuce and spinach, selected as indicator plants, is shown on the basis of measurements carried out by the Radiological Control Network of the Hungarian Ministry of Agriculture and Food between 1960 and 1980. The peaks in contamination levels in 1962–1963 as well as in the autumn of 1976 can be clearly observed.

In Table 43, the results of measurements of ^{90}Sr and ^{137}Cs activities of spinach, lettuce and sorrel samples carried out by the Radiological Control Network of the Hungarian Ministry of Agriculture and Food in 1980 are given. It can be seen that the level of contamination by ^{90}Sr and ^{137}Cs was about the same.

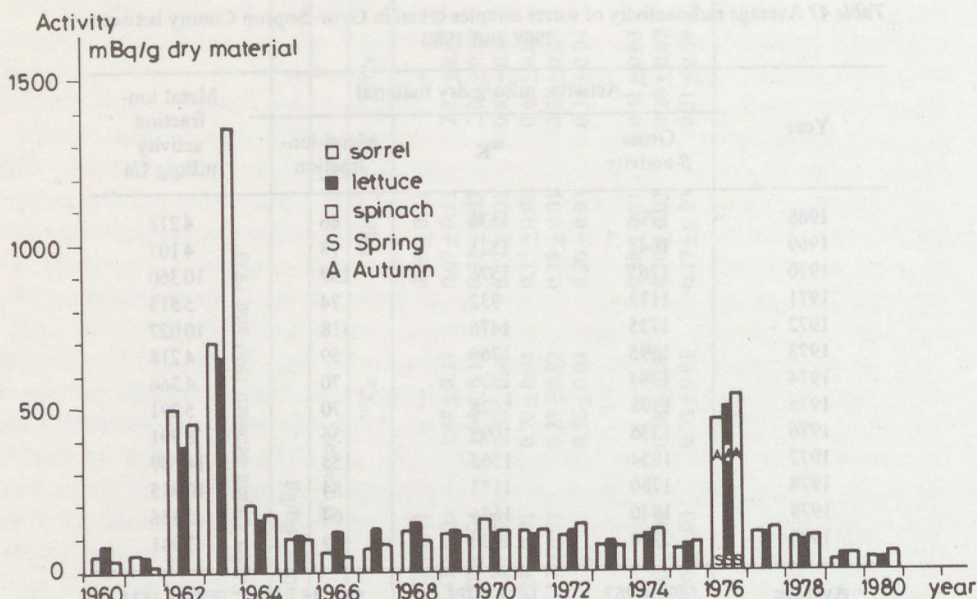


Fig. 21. Average metal ion-fraction activity of indicator plants between 1960 and 1980 in Hungary

The majority of radioactive contamination in plants is usually due to the ^{90}Sr and ^{137}Cs isotopes. However, in the case of early fall-out it may happen that a significant part of contamination is due to radioactive isotopes with a relatively short half-life (^{89}Sr , ^{91}Y , $^{95}\text{Zr} + ^{95}\text{Nb}$, $^{106}\text{Ru} + ^{106}\text{Rh}$, $^{140}\text{Ba} + ^{140}\text{La}$, ^{141}Ce , $^{144}\text{Ce} + ^{144}\text{Pr}$, ^{147}Pm , ^{151}Sm , etc.). However, in such cases the plants take up the radioactive isotopes either directly through the leaves, or most of the contamination is due to surface dust pollution; therefore it may be washed down by water. When the plant takes up the radioactive substances from the soil, the migration from the ground surface to the root zone as well as the matter transport through the roots into the plant tissues needs a relatively long period of time; thus within this period the activity of the isotopes of short half-life will significantly decrease. This is the reason why only a very small amount of isotope of short half-life may arrive in the plants from the soil.

Within the first few weeks after a nuclear explosion (or perhaps a reactor disaster), the effects of the radioactive iodine isotopes (^{131}I , ^{133}I , ^{135}I) are the greatest radiation danger. These isotopes occur only in the tropospheric and in the early local fall-outs, thus they represent no danger for the growth and later the maturation of plants. Their contamination is to be found almost exclusively on the surface of the plants. According to the systematic radiometric measurements carried out in the USA, imported cocoa, tea and coffee contains the ^{131}I isotope only in concentrations below detection limit (Simpson et al. 1974). The same can also be said of the activity of the ^{106}Ru isotope.

In Table 44 data are given on the radioactive contamination of fruits (apple, cherry, pear, raspberry, plum, etc.) and vegetables (bean, peas, lettuce, potato, etc.) in Hun-

gary in 1986. The very large standard deviation values prove the heterogeneous distribution of radioactive contamination as a function of locality, time and species.

For comparison, in Table 45 the radioactive contamination intervals measured in vegetable food in Belgium in May, 1986, are shown (Deworm 1986). It should be mentioned that in May, 1986, as a consequence of the Chernobyl accident, the total β -activity of the different plant samples reached in some cases the 5000–10 000 Bq/kg contamination level.

The presence of ^{239}Pu and certain actinides should be mentioned, which contribute only to a very small portion of the radioactivity of the plant tissues. Usually the different biological materials contain these radioactive isotopes in a concentration range of 0.18–3.7 mBq/kg original material (Kracke, Bunzl 1978). Table 46 contains data about the activities of ^{239}Pu , ^{240}Pu and ^{241}Am in some plants (Hardy et al.

Table 43 Results of measurements of ^{90}Sr and ^{137}Cs activity in samples of spinach, lettuce and sorrel carried out by the RCN in 1980

Sample	^{90}Sr activity		^{137}Cs activity	
	mBq/g dry material	mBq/g, Ca	mBq/g dry material	mBq/g, K
Spinach	22 ± 13	1330 ± 893	27 ± 26	478 ± 337
Lettuce	20 ± 6	1440 ± 200	27 ± 8	600 ± 737
Sorrel	11 ± 5	875 ± 451	21 ± 12	665 ± 363

Table 44 Radioactive contamination of fruits and vegetables in Hungary in 1986

Radioactive isotope	Fruits		Vegetables	
	No. of samples	activity Bq/kg	No. of samples	activity Bq/kg
^{131}I	536	35 ± 52	284	38 ± 185
^{134}Cs	277	17 ± 18	275	15 ± 29
^{137}Cs	287	31 ± 31	283	20 ± 53
Total- β	12	93 ± 15	138	139 ± 101

Table 45 Radioactive contamination of plant food in Belgium, May 1986

Sample	Activity, Bq/kg						
	^{131}I	^{132}Te	^{103}Ru	^{137}Cs	^{134}Cs	^{136}Cs	$^{140}\text{La-Ba}$
Strawberry	17–23	2–4	0–5	0–6	2–4	2–3	2–4
Radish	20–44	2–4	3–21	3–10	1–8	1–2	1–2
Lettuce	20–300	1–25	3–59	2–36	1–22	0–4	0–14
Spinach	26–1080	1–180	5–420	8–144	3–25	0–11	1–21
Tomato	1–6	1–2	0–4	0–4	1–2	0–1	0–3

1977). It can be seen that the activities of $^{239}\text{Pu} + ^{240}\text{Pu}$ and ^{241}Am are at least two orders of magnitude smaller than the contamination level of ^{137}Cs and ^{90}Sr .

In addition to the above-discussed isotopes, every plant (like all living creatures) also contains a significant amount of hydrogen, as well as tritium. The tritium content of plants depends mainly on the composition of the water, the values of ^3H con-

Table 46 ^{137}Cs , ^{90}Sr , $^{239}\text{Pu} + ^{240}\text{Pu}$ and ^{241}Am activity of vegetables referred to kg original material

Crop	Activity, mBq/kg original material			
	^{137}Cs	^{90}Sr	$^{239}\text{Pu} + ^{240}\text{Pu}$	^{241}Am
Corn	111 ± 4	59 ± 4	0.81 ± 0.11	0.09 ± 0.02
Potato	130 ± 7	33 ± 4	0.28 ± 0.02	0.06 ± 0.01
Pea	85 ± 30	200 ± 22	0.07 ± 0.07	0.07 ± 0.07

centration measured in plants are similar to those in the surface and ground waters and precipitation. In the section on the radioactivity of the hydrosphere, we have already discussed in detail that ^3H is produced by cosmic radiation in amounts not negligible through a completely natural process. However, the amount of tritium produced by nuclear weapon tests, as well as emitted by nuclear facilities, is much greater.

We have already mentioned that the radioactive isotopes migrate into the plant tissues not only from the soil through the roots, but also directly through the leaves, and for certain isotopes this is the predominant mode of uptake. The temperature and humidity of the air also play a role. In the case of strontium, the amount of ^{90}Sr taken up from the soil is usually greater; however, in certain plants as much as 40% of the total ^{90}Sr activity may be taken up through the leaves (Yudintseva, Gulyakin 1968). For ^{137}Cs , uptake through leaves is predominant, often almost the whole ^{137}Cs content of a plant is taken up through the leaves (Benes et al. 1971; Rohleder 1972).

Field experiments demonstrated that the $^{90}\text{Sr}:\text{Ca}$ proportion in the leaves is lower than in the stem. Hence, it is evident that the majority of ^{90}Sr is taken up from the soil through the roots and during further transport, because of the different diffusion and migration speed of Ca and Sr, the $^{90}\text{Sr}:\text{Ca}$ ratio somewhat decreases. When the plant takes up a significant share of ^{90}Sr directly from the air, the $^{90}\text{Sr}:\text{Ca}$ proportion increases. The completely opposite trend was found in the case of ^{137}Cs , where the $^{137}\text{Cs}:\text{K}$ proportion in the leaves was many times higher than in the stem. This is clearly due to the fact that most of the ^{137}Cs in the plant enters through the leaves. Similarly to the ^{137}Cs isotope, most of the radioactive isotopes of short half-life contaminate the plants only by ion-uptake through the leaves. The plants are practically unable to take up these isotopes from the soil as the soil absorbs them well. By the time they reach the root zone, which is determined by the mechanical structure and chemical composition of the soil (as well as the agrotechnical methods applied), on the one hand they are mostly decayed, and on the other

hand the plants take up the ions selectively from the soil. As these are isotopes of chemical elements not essential for plants, because of ion competition, they are taken up by the root cells only in an extraordinarily small amount. For example, 90% of ^{95}Zr is taken up through the leaves (Rohleder 1972). In the case of leaf cells, although the kinetic constants and the features of ion selectivity are similar to those characteristic for the ion-uptake mechanism of root cells, the competition among different ions during uptake by leaf cells is not so strong, and actually these radioactive substances may reach the plant tissues without delay, i.e., well before their radioactive decay. Since a certain part of the isotopes of short half-life does not exist in any ionic form and forms no water-soluble compounds, these isotopes do not enter into the inner organs of the plant and actually they represent only a direct surface contamination.

On the basis of Tables 39, 41 and 43, containing the data of Ocker (1968), Bose (1970) and Schulze-Wollgast (1969, 1972), as well as the results of our own measurements of the radioactive contamination of vegetables and fodder plants, it can be said that the ^{90}Sr and ^{137}Cs contamination of different plants, as a consequence of fall-out from nuclear explosions, is of about the same order of magnitude, or the ^{137}Cs activity is not more than several times higher than the ^{90}Sr activity. It is well known that ^{90}Sr and ^{137}Cs fall-out is of the same order of magnitude, and on average the $^{137}\text{Cs} : ^{90}\text{Sr}$ proportion of the soils has a value around 1.6. However, at the same time the discrimination capacity of the plants for strontium and cesium is very different. While the discrimination factor in relation to strontium, describing the ion uptake from the soil,

$$D_{\text{Sr}} = \frac{{}^{90}\text{Sr/Ca in the plant}}{{}^{90}\text{Sr/Ca in the soil}}$$

usually has a value of 1, the discrimination factor in relation to cesium,

$$D_{\text{Cs}} = \frac{{}^{137}\text{Cs/K in the plant}}{{}^{137}\text{Cs/K in the soil}}$$

has a value of about 0.01. Since the K content of the plants is not more than several times the Ca content, the logical consequence of the above-mentioned facts would be a significantly higher ^{90}Sr content of the plants than the level of ^{137}Cs contamination if ion uptake occurred only from the soil. As we have seen, under field conditions, or rather for cultivated plants, this is not so. This is the reason we conclude that during ^{137}Cs contamination of a plant it is the ion uptake through the leaves that plays the decisive role and not that from the soil.

2.4.3 The correlation between different species and growth phase of plants and the level of radioactive pollution

As we stressed earlier, the soil/plant discrimination factor for the Sr-ion uptake is as follows:

$$D_{\text{Sr}} = \frac{{}^{90}\text{Sr/Ca in the plant}}{{}^{90}\text{Sr/Ca in the soil}} = \frac{\text{Sr/Ca in the plant}}{\text{Sr/Ca in the soil}}$$

Its value is ≈ 1 , thus similar $^{90}\text{Sr}/\text{Ca}$ proportions can be expected in the plants as in the soil, because the decisive majority of radiostrontium is taken up by the plant from the soil. Table 47 shows the values of discrimination factors for some plants, indicating the accumulation of ^{90}Sr and ^{137}Cs isotopes taken up from the soil (Yudintseva, Gulyakin 1968).

Table 47 Values of discrimination factors characteristic for the different plants

Plant species	Plant/soil discrimination factor	
	^{90}Sr	^{137}Cs
Wheat	0.6–1.6	0.004–0.02
Pea	0.6–1.2	0.001–0.02
Oats	0.6–1.0	0.001–0.004
Trefoil	0.3–0.9	0.004–0.008
Beet	0.4–2.2	0.007–0.04

Because of the similar chemical properties of Sr and Ca, Sr is always expressed as a ratio of the Ca content. Thus it is quite natural that, expressed in dry matter, there may be significant differences between the ^{90}Sr contamination levels of plants growing under similar agricultural conditions and having nearly the same discrimination capacity. For example, comparing the ^{90}Sr radioactivity of spinach and lettuce, it can be stated that spinach has a higher activity expressed in units of the mass of its dry matter. However, if the ^{90}Sr activity is calculated in relation to the Ca content, it becomes clear that the two values are almost the same as spinach has a significantly higher Ca content than lettuce. For the same reason legumes accumulate 3–7 times more radiostrontium in relation to their dry matter than cereals do.

The ^{90}Sr uptake of a series of different plant species decreases in the following order (Yudintseva, Gulyakin 1968); pea, trefoil, beet, potato, oats, wheat, flax. According to Haghiri (1970), maize do not accumulate as much ^{90}Sr as the Gramineae species do. The different contamination level of various plants, under identical soil and meteorological circumstances, is due to the different capacities to discriminate, i.e., to the selective ion-uptake capabilities of the different plant species. The selective ion-uptake capability of plants is influenced by the following factors:

- the ion in question is absorbed by the roots of the various plant species in different degrees;
- the tissue structures of the various plant species being different, the speed of transport of the ions absorbed by the roots is quite different;
- the metabolic end-products of the plants have an influence on the transport of certain ions, sometimes forming complexes (chelates) with other metal ions;
- the equilibrium of the bi-directional transport (from the roots to the leaves and vice versa) is different in each plant.

A comparison of the radioactivities of the grains and the straw of cereals gives interesting results. The data in Table 48 clearly show that the radioactive substances

concentrate more in the stem of plants than in the grain, i.e., the reproductive parts are less contaminated than the vegetative organs.

In order to illustrate the transfer of ^{90}Sr relative to that of Ca from straw to grain in the crops, the quotient

$$\frac{{}^{90}\text{Sr}/\text{Ca in the grain}}{{}^{90}\text{Sr}/\text{Ca in the straw}}$$

has been computed in Table 49 from the data of Haak and Lönsjö (1975).

In agreement with the literature reviewed this quotient ($\text{DF}_{\text{grain}/\text{straw}}$) is smaller than 1, which shows that Sr is less mobile than Ca during their passage from straw to grain. As found by Fredriksson and Eriksson (1970), the $\text{DF}_{\text{grain}/\text{straw}}$ is smaller in peas than in barley, which is due to the physiological differences between the two species.

Among the various grains, rice contains the least ^{90}Sr (Ocker 1968; Kodaira, Tsumura 1973). In addition to the possible differences in discrimination capabilities of the various species, the low concentration of ^{90}Sr in rice is first of all due to the fact that rice is grown on irrigated land; thus most of the strontium gradually reaches the deeper layers of the soil, from which the plant is unable to utilize it.

The distribution of radioactive pollution, as shown in Table 48, is not homogeneous among the different parts of the plant. It can be seen that, e.g. comparing the Sr : Ca proportion, it decreases through ion transport from the vegetative parts to the generative (reproductive) parts (Goltsev, Aleksahin 1969). However, the different Ca content plays a more decisive role because of the common ion transport system. First of all, ^{90}Sr accumulates in the vegetative organs because of their higher Ca content. Thus the radioactive contamination expressed as dry matter of hay is significant-

Table 48 Radioactivity of some corn straws

Sample	Activity, mBq/g dry material		Metal ion-fraction : gross β -activity %
	Gross β -activity	Metal ion-fraction	
Wheat grain	111	7	6
Wheat straw	459	152	33
Barley grain	167	11	7
Barley straw	440	93	21
Oats grain	174	22	13
Oats straw	666	41	6

ly higher than that of grain fodders. Also the ^{90}Sr contamination of wheat leaves expressed in dry matter is higher by about one order of magnitude than the value measured in the grains.

In addition to the determination of the relative concentration of the radioactive isotopes, the discrimination capability may also be studied through the measurement

of the concentrations of the stable isotopes, i.e. of the inactive concentrations of the elements. The quotient in relation to the discrimination capability of the plants

$$D'_{\text{Sr}} = \frac{{}^{90}\text{Sr}/\text{Ca in the plant}}{{}^{90}\text{Sr}/\text{Ca in the soil}}$$

will be equal to the value of the quotient

$$D''_{\text{Sr}} = \frac{\text{Sr}/\text{Ca in the plant}}{\text{Sr}/\text{Ca in the soil}}$$

if the mineral (and radioactive) substances are taken up by the plants exclusively from the soil. However, because the ${}^{90}\text{Sr}$ isotope may be taken up even through the leaves, e.g. from precipitation, the value of the ${}^{90}\text{Sr}/\text{Ca}$ quotient may be higher than the Sr/Ca quotient (supposing of course that the inactive strontium is taken up

Table 49 $DF_{\text{grain/straw}}$ in crops during 1961–68

a. Effect of topsoils

Soil no.	Barley		Peas	
	Subsoil I	Subsoil II	Subsoil I	Subsoil II
1	0.83	0.63	0.57	0.59
2	0.73	0.68	0.65	0.66
3	0.88	0.69	0.60	0.60
4	0.74	0.75	0.67	0.59
5	0.86	0.73	0.52	0.64
6	0.90	0.75	0.66	0.64
7	0.85	0.76	0.59	0.58
8	0.86	0.70	0.74	0.64
9	0.95	0.80	0.85	0.67
10	0.85	0.78	0.67	0.59
11	0.82	0.73	0.64	0.52
12	0.90	0.77	0.77	0.63
Mean 1–12	0.85	0.73	0.66	0.61

b. Effect of years

Years	Barley		Peas	
	Subsoil I	Subsoil II	Subsoil I	Subsoil II
1961	0.61	0.44	0.56	0.31
1962	0.48	0.45	0.35	0.35
1963	1.05	0.98	0.90	0.71
1964	0.74	0.60	0.62	0.69
1965	0.91	0.92	0.65	0.63
1966	1.02	0.88	0.80	0.76
1967	0.94	0.60	0.71	0.64
1968	1.02	0.99	0.80	0.75

by the plants practically only through the roots from the soil, and only in a negligible amount through the leaves). This is the reason why the discrimination factors cannot be calculated from the ^{90}Sr concentrations measured in soils of arable fields and in plants growing there. For example, in Hungary in 1980 the average ^{90}Sr activities of the soils and the various agricultural and wild plants was 620 and 800–1600 mBq/g Ca, respectively. However, it does not mean that the value of discrimination factor for strontium in plants was between 1.3 and 2.6.

In order to compare the data in Tables 39 and 43, in Table 50 the data of the activities of wild-plant samples investigated by the Radiological Control Network of the Hungarian Ministry of Agriculture and Food in 1980 are listed. The comparison demonstrated that there was no significant difference between the contamination levels of the different cultivated vegetables and the wild plants.

As we have mentioned before, the plants discriminate quite well when they take up Cs from the soil. Of course, cesium is bound strongly to the soil; thus only a very small amount of it is in the soil solution. The value of the plant/soil discrimination factor in the case of Cs-ion uptake from the soil,

$$D_{\text{Cs}} = \frac{{}^{137}\text{Cs}/\text{K in the plant}}{{}^{137}\text{Cs}/\text{K in the soil}} = \frac{\text{Cs}/\text{K in the plant}}{\text{Cs}/\text{K in the soil}}$$

is about 0.01. As a consequence, most of the cesium in plants is taken up through the leaves and not from the soil. Expressed in dry matter, the ^{137}Cs uptake of different plants increases in the following order (Moiseyev et al. 1972): wheat, barley, pea, millet, oats, sugar beet, bean.

It is interesting that in the Arctic region the ^{137}Cs contamination of plants, and consequently that of animals and humans, is several times higher than in the temperate zones (Ramzayev et al. 1973; Marey 1980). This may be attributed to the fact that lichens and other Arctic plant species accumulate ^{137}Cs in a much higher degree, and even the amount of ^{137}Cs coming from soil into the plants is significant.

We have carried out several experiments studying the radioactive contamination of various plants. The most important statements may be summarized as follows (Szabó 1976, 1977, 1979, 1980):

1. For Sr, the plant/soil discrimination factor (D_{Sr}) has a value 0.92 ± 0.24 , depending on the type of soil (alluvial meadow soil, brown forest soil), the species of

Table 50 Results of investigations on wild plants carried out by the RCN in 1980

Sample	Number of samples	Activity, mBq/g dry material			^{90}Sr -activity		^{137}Cs -activity	
		gross- β	^{40}K	metal ion-fraction	mBq/g dry material	mBq/g Ca	mBq/g dry material	mBq/g K
Nettle	59	1113 ± 287	656 ± 206	67 ± 31	39 ± 17	1192 ± 571	22 ± 18	1028 ± 1077
Mugwort	49	978 ± 248	767 ± 215	38 ± 22	15 ± 8	1560 ± 968	13 ± 10	642 ± 684
Morning glory	36	985 ± 333	732 ± 219	44 ± 21	16 ± 11	1338 ± 842	19 ± 16	803 ± 882

plant (sorrel, spinach, lettuce, lucerne, bean, wheat, maize), calculated from the radioactivity data as well as strontium measurements carried out by flame photometry. It proves that the plant organism does not discriminate in the Sr-Ca system, i.e., in the case of ion uptake from the soil the Sr/Ca proportion in the soil and in the plant is the same.

2. According to our measurements and calculations, the plant/soil discrimination factor referred to cesium is $D_{Cs} = 0.014 \pm 0.003$. That means that the plants discriminate very well for cesium, and as cesium is bound strongly to the soil particles, the Cs/K proportion in the plants may be lower by about two orders of magnitude than the proportion measured in soil if ions are taken up from the soil by the plant.

3. As regards the distribution of the ^{90}Sr content taken up from the soil, we established that, compared to the total ^{90}Sr content of the plant, $82 \pm 4\%$ is found in the above-ground parts of the plant in wheat, $85 \pm 3\%$ in maize, and $86 \pm 4\%$ in bean. Thus only a relatively small percentage of ^{90}Sr taken up from the soil remains in the root; most of it accumulates in the stem and in the leaves. According to Yuditseva and Gulyakin (1968), if the uptake is from the soil, the proportion of ^{90}Sr in the above-ground parts of different plants is between 77 and 92%, and in the underground parts of plants varies from 8 to 23%.

The distribution of ^{137}Cs is different from that of ^{90}Sr . We have found that most of ^{137}Cs remains in the root; only a relatively small percentage reaches the above-ground parts. In maize, wheat and bean $58 \pm 7\%$, $61 \pm 5\%$ and $68 \pm 4\%$, respectively, remains in the root. According to Yuditseva and Gulyakin (1968), the proportion of ^{137}Cs in the above-ground parts of different plants is between 18 and 78%, and in the underground parts of plants it varies between 22 and 82%, if the element is taken up from the soil.

Of course, the proportion of the above-ground and the underground parts of a plant may change during its growth and development. Even the amount of available water has an influence on the distribution of radioactive isotopes, or rather on their proportion in the above and underground parts of the plant. When there is plenty of water, the mass of the root is smaller; thus a higher proportion of radioactive substances concentrates in the above-ground parts of the plants.

Comparing the Sr:Ca and Cs:K concentration proportions in the above-ground and underground parts of the plants, it can be seen that both are smaller in the stem than in the root. It documents the fact that the relative concentration of Sr and Cs decreases during matter transport from the root to the leaves. This is probably due to the different atomic mass of Sr and Ca, as well as Cs and K, respectively, which in turn causes the lower mobility of the Sr and Cs ions.

4. The influence of the K, Na and Ca content of soil on the ^{90}Sr uptake by the plants has also been examined. We found that increasing the Ca content of the soil significantly decreased ^{90}Sr uptake. The ^{90}Sr uptake can also be decreased by the increase of the inactive strontium content of the soil. According to our measurements, increasing the K or inactive Cs content of the soil decreases the ^{137}Cs uptake from the soil; however, as we have said earlier, under natural conditions this is of no practical importance: in the field most of the ^{137}Cs contamination is through direct ion uptake by the leaves.

Table 51 Average radioactivity of fruits and vegetables

Sample	Activity, mBq/g dry material			Metal ion-fraction total- β %
	total- β	^{40}K	metal ion-fraction	
Apple	266	229	26	10
Pear	259	226	19	7
Strawberry	625	514	48	8
Grape	448	437	4	1
Raspberry	348	289	11	3
Grey clitocybe (mushroom)	1528	1321	130	9
Cucumber	78	70	4	5
Capsicum (paprika)	1003	895	81	8
Carrot	607	411	26	4
Cabbage	818	792	19	2
Pea	278	215	11	4

5. Between 1971 and 1974 a large-scale survey was carried out in order to measure the radioactive contamination of certain foodstuffs of plant origin of great importance for human nutrition, but not included in the programme of the Radiological Control Network of the Hungarian Ministry of Agriculture and Food. The results can be seen in Table 51. We found that the radioactive contamination of the foodstuffs of vegetable origin was of low level and the activity of the metal ion-fraction characteristic for the contamination level represented only 1–10% of the gross β -activity. Similar proportions were also found during the radiometric investigations of other plant samples (tree leaves, sugar beet, tobacco, cereals, etc.). According to our measurements, the contamination of rye was higher than that measured in wheat or rice.

6. Because of the Chernobyl disaster, in 1986 significantly higher contamination could be measured in numerous countries than in previous years. However, the level of radioactive contamination in vegetable foodstuffs in 1987 "returned" to the order of magnitude characteristic for years before the disaster.

A further important factor for the buildup of radioactive contamination in plants is the developmental phase of the plant, in which substantial radioactive pollution is sustained (Pelshenke et al. 1967). That is to say, young plants accumulate more radioactive substances (expressed as dry matter) than the more mature ones do. As the transport of the ions taken up by the root is not homogeneous within the plant, and the radioactive isotopes can also be taken up through the leaves, the contamination levels of parts of the same plant with different location and age may be fairly different. For example, ^{90}Sr accumulates first of all in the vegetative organs; thus only a smaller amount reaches the generative organs (Menzel et al. 1961).

The behaviour of younger and older plants is also different as regards ^{137}Cs uptake. Young wheat and sunflower take up ^{137}Cs isotopes sprinkled on their leaves 2–4 times faster than the older ones (Yuditseva et al. 1972). Incidentally, the uptake of radioactive nuclides, irrespective whether the uptake is through the root or through

the leaves, continues till the end of the vegetation period, and usually the contamination level reaches its maximum in the mature plant.

In certain cases, an important task of agricultural radioecology is to forecast the level of radioactive contamination in plants. Gromov et al. (1982) have elaborated a model, and by its application the expectable contamination level can be forecast with high probability considering various agrometeorological parameters. In order to prepare a proper forecast, in addition to the contamination levels of the crop in earlier years, the contamination and moisture content of the soil, the amount of precipitation, the temperature, the yield, the amount of fertilizers used, even the discriminatory capabilities of the different plants as regards the radioisotope uptake from the soil are considered (Gulyakin, Yudintseva 1973). Some questions of agricultural radioecology are also discussed, for example by Batschinski (1979).

2.4.4 Factors influencing radioactivity of the soil-plant biological chain

In subchapter 2.3, where the issue of soil radioactivity and the migration of radioactive isotopes within the soil were discussed, I stressed that because the plants take up a significant part of their radioactive substances from the soil, the radioactivity of the soil and the vegetation should be discussed as one system. It is justified by the continuous interactions characteristic of the soil-plant system, by the complex and dynamic relationship which is the result of the processes between the soil and the plants living on (within) it. This relationship in turn determines the migration and transport of the radioactive isotopes, the level of the radioactive contamination of the soil and vegetation, continuously changing as a function of time. The interaction manifests itself, for example, by the fact that not only the soil determines the level of radioactivity and the contamination of vegetation but the vegetation also has an influence, e.g. by modifying the water take-up of the soil on the migration and the vertical (and horizontal) distribution of the radioactive substances within the soil. For an ecologist the radioactivity of the soil or the change in the level of contamination is not very interesting in itself. It is important to find out what are the connections with the changes in other parts of the biosphere and finally what is its influence on humans.

Earlier we discussed several factors, for example, the geological, orographical and topographical conditions, the meteorological parameters, the mechanical structure and the chemical composition of the soil, the applied cultivation methods and the plant species. All of these may play a decisive role in the buildup of the radioactivity of the soil-plant system. We do not consider it necessary to repeat what has already been said in the earlier parts of this book; however, some general statements should be added.

The radioactive contamination of soils is determined chiefly by the quantity of radioactive isotopes coming to the surface by precipitation or dry-out. They arrive in the deeper layers of the soil depending on the mechanical and chemical characteristics of the soil, and on the amount of water available, as well as on other parameters. The

soil fixes the different radioactive substances and absorbs them, thus making it difficult, sometimes even practically impossible, for the plants to take up the active isotopes. For instance, cesium is taken up by plants from water solution quite easily, but not so from the soil, chiefly because of the absorbing effect of the different clay minerals. The relative concentrations of the radioactive nuclides, just because of the different sorptive capacity of the soil for the different radionuclides or because of the selective ion-uptake of the plants, may be significantly different from the contamination proportions within the plants under the influence of the ion-uptake from the soil. (Of course, as already mentioned several times, plants may also take up radioactive substances through the leaves.) The different contamination proportions are due to several factors. The soil, in addition to the ^{90}Sr , ^{89}Sr and ^{137}Cs isotopes which play a dominant role in the buildup of the radioactive contamination of the plants, is also polluted in a significant degree by other radionuclides coming from radioactive fall-out, e.g., ^{106}Ru , ^{144}Cs and ^{239}Pu . However, only very small percentages of these substances come from the soil through the root into the plant organs, and most of the activity taken up from the soil remains in the root or in the underground organs of the plant. According to Yudintseva and Gulyakin (1968), for example only 0.1–0.5% of ^{106}Ru taken up from the soil reaches the above-ground organs in oats and 0.1–0.3% in beans. Only 0.7–1.5% of the ^{144}Cs taken up arrives in the shoots of oats and 1.4–4.5% in peas. Thus in effect they contaminate only the root.

As in the case of ^{106}Ru and ^{144}Cs , the uptake ratio of the transuranium elements by plants from the soil is also rather low (Nishita et al. 1981; Vyas, Mistry 1981). In pea, wheat and tomato the activity level of Pu expressed in dry matter was smaller by 4–6, for Am and Cm by 2–6 and in the case of Np by 1–4 orders of magnitude than the activity values of the soil (Romney et al. 1981).

The ^{137}Cs uptake from a given type of soil is determined to a great extent by the mechanical structure of the soil, the exchangeable (available) potassium content and the absorption capacity (Yudintseva et al. 1981). Although the ^{137}Cs uptake from the soil is negligible if there is uptake through the leaves, its accumulation in the case of grassy-podsol sand soil and crasnozem (red soil) may be higher by one order of magnitude than in a grassy-podsol clay soil and chernozem. According to a communication by Moiseyev et al. (1982), only about 0.05–0.85% of the Cs content of the different types of soil (chernozem, podsol, sodic soil, etc.) occurs in water-soluble form, i.e., is suitable for uptake by the plants. This is the reason why plants are able to take up cesium and ^{137}Cs from the soil only in very limited amounts. The decisive part of ^{137}Cs contamination of plants can be explained by the ion uptake through the leaves; however, the soil composition may also have an influence to a certain extent on the radioactive isotope uptake through the leaves. For example, in soils with plenty of potassium plants take up much smaller amounts of ^{137}Cs isotopes than in soil with lower potassium content (Yudintseva, Gulyakin 1968).

^{90}Sr is taken up from the soil by plants in a much greater proportion than ^{137}Cs . Thus, especially in the case of light soils, it may happen that harvesting plays a greater role in the decrease of the radiostromium contamination level of the soil than the radioactive decay or the migration of ^{90}Sr towards the deeper layers of the soil. Actually, the degree of strontium uptake from the soil, as we have already mentioned,

is determined by the exchangeable Ca capacity of the soil. The ^{90}Sr contamination of the plant tissues is related first of all to the chemical composition of the plant (mostly to its Ca content), and this is the reason why, for example, the ^{90}Sr contamination of papavers expressed in dry matter is significantly higher than the radiostrontium contamination of cereals.

It is interesting to observe that the radioactive elements (mother and daughter elements) in equilibrium in the soil, like ^{90}Sr and ^{90}Y , ^{210}Pb and ^{210}Po , ^{95}Zr and ^{95}Nb , do not accumulate in the plant in the same equilibrium proportions because of the different absorption characteristics of the soils, as well as because of the selective ion uptake of the plants. Plants take up ^{90}Y , a daughter element of ^{90}Sr only in a much lower amount than the ^{90}Sr isotope; thus the radioactive equilibrium between ^{90}Y and ^{90}Sr is reached only within the plant, after the ^{90}Sr uptake. (As the half-life of ^{90}Y is only 64 hours, in the plant tissue there is a virtual equilibrium 1–2 weeks after the ^{90}Sr uptake, i.e., the $^{90}\text{Sr} + ^{90}\text{Y}$ activity will be nearly double the ^{90}Sr activity.)

The radioactive contamination of the soil, or rather the soil–plant system, can be influenced to a great extent by the application of different agrotechnical and agrochemical methods. According to the measurements of Malikov et al. (1982), increasing the carbonate content of chernozem soils up to 2–3%, decreased the amount of ^{90}Sr taken up by the different plants (cereals, vegetables and fodder plants) to 35–90%, which in turn indicates the better strontium-absorption capacity of the soils with high carbonate content. At the same time ^{137}Cs uptake increased 1.3–4.0 times. This great ion mobility is due to the fact that the increase in the amount of organic substances easily solvable in water promotes the desorption of the ^{137}Cs nuclide fixed within the soil. Carbonate ions form easily dissolvable complex compounds with the organic substances; thus they promote the desorption of Cs and its uptake by the plants.

Table 52 Percentile rate of reutilization of onion and potato

Investigated plant	Mean reutilization (%)			
	^{90}Sr	^{65}Zn	^{54}Mn	^{60}Co
Onion	70	25	15	1
Potato	30	33	24	2

The ion uptake through the leaves, on the above-ground organs of plant, has a general feature practically independent of the polluting radio-isotope. The majority of the radioactive substances accumulate in the stem and in the leaves, that is in the vegetative organs, and much smaller amounts of radioactive substances contaminate the fruit grains, i.e., the reproductive organs. Thus, as we have mentioned earlier, the radioactivity of wheat straw is several times higher than that of the grain. This fact is due partly to the differences in chemical composition between the vegetative and reproductive parts of the plants (to the higher potassium and calcium content of the

vegetative parts), and partly to the decrease of the proportions Sr:Ca and Cs:K during the ion-transport from the vegetative parts to the reproductive parts, because of the lower mobility of these radioactive isotopes. The relative mobility of each radioactive isotope in relation to the inactive carriers, for ^{90}Sr to Ca, for ^{137}Cs to K, is not independent of the degree of reutilization. It may be important to know, for example, in assessing the applicability of seed-corn contaminated with radioactive materials. On the basis of investigations by Ryabova and Ryabov (1982), the degree of reutilization valid for onion and potato may be seen in Table 52.

2.4.5 Possibilities of decreasing radioactive contamination in agricultural products and vegetables.

The effect of different processes of food preparation on contamination

Decontamination, i.e., the decrease of the radioactive pollution of agricultural products and vegetable foodstuffs can be carried out using two basic methods. On the one hand, preventive methods can be applied; on the other hand the agricultural products, fodders and foodstuffs can be decontaminated.

The preventive methods have already been discussed: all the agrotechnical and agrochemical methods can be classified here which can contribute to the decrease in the amount of the radioactive substances passing from the soil into the plants. Actually, as part of the radioactive contamination always comes from surface dust

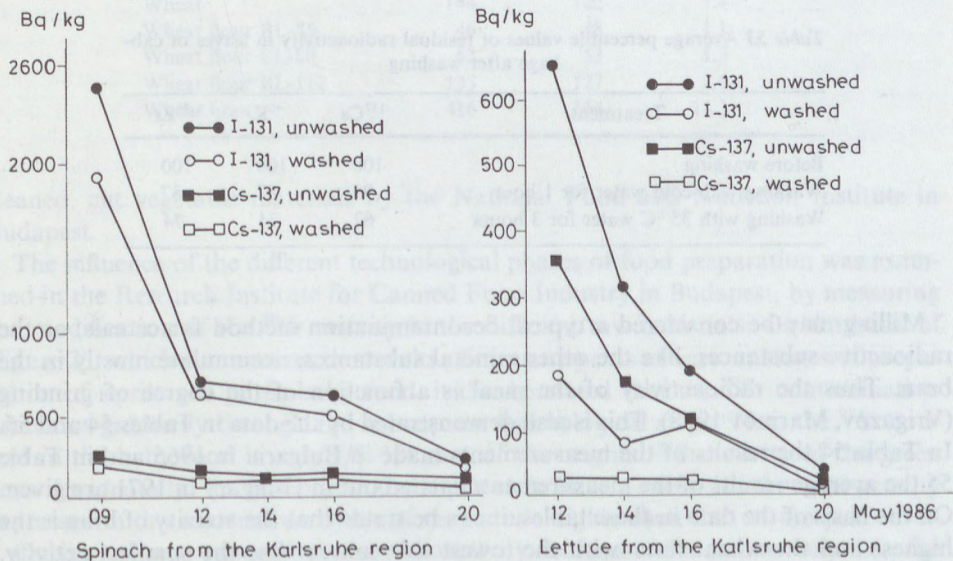


Fig. 22. ^{131}I and ^{137}Cs contamination of spinach and lettuce in FRG (Karlsruhe), May 1986

contamination, covering of the fruits or the vegetable parts, storage in covered places, and prevention of surface pollution can be placed among the preventive methods.

When most of the pollution of the plants originates from dust contamination, simple washing with clean water can also be a reasonably efficient method of decontamination (Pelshenke et al. 1967). For vegetables the efficiency of decontamination can be enhanced by mixing organic acids (e.g. citric acid, tartaric acid) with the washing water (Paulus 1968). Incidentally, cesium is easier to remove than strontium. According to a communication by Paulus (1969), the maximum efficiency of decontamination for ^{137}Cs and ^{90}Sr is 90–95% and 60–75%, respectively. Even cereals can be decontaminated by washing, when the washing water contains phosphoric acid or polyphosphates (Ocker 1969).

Figure 22 shows the ^{131}I and ^{137}Cs activities of spinach and lettuce samples examined in May 1986 in West Germany, i.e. immediately after the Chernobyl disaster (Diehl 1986). It can be seen that the activity of the washed-out (decontaminated) samples was significantly lower.

The elements taken up by the root, i.e., not from surface contamination, may be washed out from the leaves of the plants by water. They can be classified into three groups according to proportion of the washed-out amount as follows:

1. Easily washed-out (over 10%), e.g. Na, Mn, Cs;
2. Moderately easily washed-out (1–10%), e.g. Ca, Mg, Sr, S;
3. Not easily washed-out (under 1%), e.g. Fe, Zn, P, Cl.

The proportion that can be washed out is different in the various plants. Usually it is higher in older leaves, and lower in younger ones. Table 53 lists the percentages of the remaining amounts of ^{137}Cs , ^{40}K and ^{95}Zr within the leaves of savoy, after the water treatment (Rohleder 1972).

Table 53 Average percentile values of residual radioactivity in leaves of cabbage after washing

Treatment	^{137}Cs	^{40}K	^{95}Zr
Before washing	100	100	100
Washing with cold water for 1 hour	91	87	87
Washing with 35 °C water for 3 hours	62	74	74

Milling may be considered a typical decontamination method for cereals, as the radioactive substances, like the other mineral substances, accumulate mostly in the bran. Thus the radioactivity of the meal is a function of the degree of grinding (Vrigazov, Marinov 1968). This is also demonstrated by the data in Tables 54 and 55. In Table 54 the results of the measurements made in Bulgaria in 1965 and in Table 55 the average results of the measurements carried out in Hungary in 1971 are given. On the basis of the data in these tables, it can be stated that the activity of bran is the highest and the wheat flour with the lowest ash content has the smallest activity. (Incidentally, the maximum ash content of the wheat flour marked BL-55 is 0.55%,

the maximum of mineral substance content in the wheat flour marked BL-80 and BL-112 is 0.80% and 1.12%, respectively. BL stands for *búzaliszt*, wheat flour in Hungarian.)

During processing of the agricultural products, through the different technologies of food production (washing, cutting up, cooking, etc.) the level of radioactive contamination decreases significantly (Ralls, Maagdenberg 1971). During precooking and cooking there is a high loss of mineral (and radioactive) substances, a significant part of the mineral content of plants is released into the cooking water. Table 56 lists the cooking loss of some microelements. The measurements were carried out on

Table 54 ^{90}Sr activity of wheat, wheat flour and wheat bran in Bulgaria in 1965

Investigated sample	Ash content, %	^{90}Sr -activity Bq/kg
Wheat	1.781 ± 0.255	1.59 ± 0.23
Wheat flour	0.461 ± 0.062	0.28 ± 0.09
Wheat flour	1.124 ± 0.092	0.64 ± 0.11
Wheat bran	5.880 ± 0.647	5.77 ± 0.67

Table 55 Average results of Hungarian radiometric measurements on cereals in 1971

Sample	Activity, mBq/g dry material		
	Total- β	^{40}K	Metal ion-fraction
Wheat	144	122	7.4
Wheat flour BL-55	46	38	1.1
Wheat flour BL-80	63	53	1.5
Wheat flour BL-112	133	127	2.6
Wheat bran	416	364	13.3

cleaned, cut vegetable materials by the National Food and Nutrition Institute in Budapest.

The influence of the different technological phases of food preparation was examined in the Research Institute for Canned Food Industry in Budapest, by measuring the modification of the ^{90}Sr activity in the different raw materials. On the basis of Figure 23, which summarizes the results of measurements at various stages of preparation of sterile pea purée baby food, it can be stated that the ^{90}Sr contamination decreases gradually through the subsequent technological steps. Thus the ^{90}Sr content in the baby food is only 1/3–1/4 that of the original ^{90}Sr activity of the green peas. Incidentally, the pea pod and the pea stem have a metal ion-fraction activity expressed in dry material an order of magnitude higher than that of the pea.

During tobacco processing, the radioactivity decreases in a similar way, i.e., the radioactivity of the tobacco leaf is higher than that of the products. Supposedly it is

due to the fact that a certain part of the activity of the metal ion-fraction is due to surface dust contamination and through cleaning the level of radioactive contamination decreases.

The results of measurements carried out in order to determine the radioactivities of the tobacco and the different tobacco products are summarized in Table 57. It can be seen that the contamination of the tobacco leaves is the highest, that of the filter cigarettes is the lowest. Comparing the metal ion-fraction activity expressed in dry matter with the activity of the indicator or fodder plants, it appears that the contamination of tobacco is significantly higher. However, calculating the metal ion fraction activity expressed in terms of the Ca content, no significant difference can be seen. The reason is that the Ca content of tobacco is significantly higher than in other plants, the Ca content of tobacco ash is about 20% and that of the other plants is usually under 10%; thus tobacco accumulates radiostrontium to a greater extent than the other plants.

Table 56 Microelement loss of plants during cooking in percentage of the original microelement content

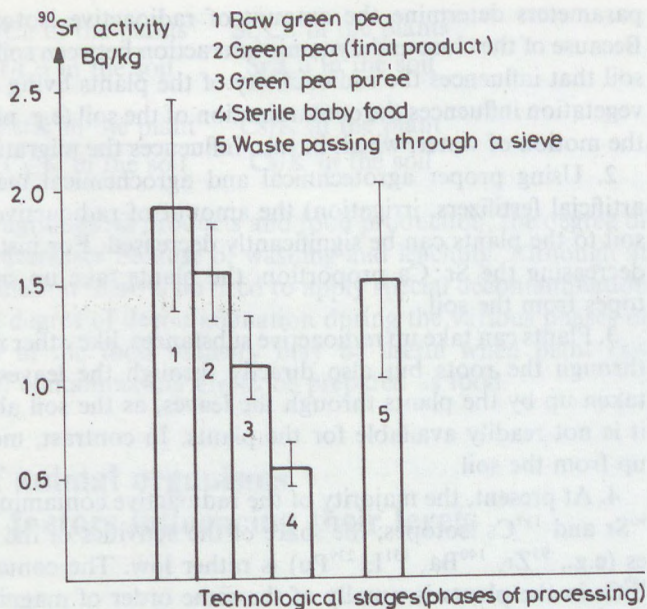
Vegetables		Loss in %		
		Cu	Zn	Pb
Potato	average	29.5	23.8	78.7
	range	14.4—65.4	12.3—37.2	47.7— 97.5
Spinach	average	10.4	23.8	23.0
	range	7.9—31.0	13.7—40.4	14.4— 53.8
Cauliflower	average	17.9	49.5	53.0
	range	10.3—46.6	19.0—74.2	22.6— 63.0
Savoy	average	13.2	44.4	74.3
	range	7.5—25.9	34.1—66.4	33.0—100.0

Table 57 Radioactivity of tobacco leaves and cigarettes

Sample	Activity, Bq/g dry material			
	Total- β	^{40}K	Metal ion-fraction	Metal ion-fraction activity Bq/g, Ca
Tobacco	1.30 ± 0.05	0.99 ± 0.05	0.28 ± 0.02	6.44 ± 0.48
Filter cigarettes	1.20 ± 0.08	0.90 ± 0.10	0.19 ± 0.01	4.44 ± 0.33
Cigarettes without filter	1.30 ± 0.10	0.90 ± 0.06	0.24 ± 0.03	5.77 ± 1.00

When assessing the radioactive contamination of the different agricultural products, fodders, raw materials for the food industry and foodstuffs, of course the nature of the contaminating isotopes must be taken into account. When chiefly isotopes of short half-lives are responsible for the radioactive contamination, decontamination means only a storage-conservation-processing problem, as the contamination will decrease after some weeks or months to a small fraction of the original one. As in such

Fig. 23. Changes of ^{90}Sr activity in the different technological phases of green pea purée baby food processing



cases the majority of the contamination is usually caused by surface contamination, simple washing with water can be an efficient method of decontamination.

In spite of the fact that in many countries of Europe in May and June 1986 the level of radioactive contamination was rather high, fortunately the present contamination levels do not necessitate the application of any special decontamination method. However, knowledge of the decontaminating effect (contamination-decreasing influence) of certain phases of food-production and processing of agricultural products may be important if there is a higher level of contamination. Knowing the efficiency of each phase, it is possible to produce food even from agricultural products of higher contamination, which by consumption may incur only hygienically admissible, low level risk.

2.4.6 Summary

The process of the buildup of natural and artificial radioactivity in plants, and its relation to the vegetation period and the plant species have been discussed in this subchapter. The factors influencing radioactive contamination of the soil-plant biological chain have been examined. Finally the methods of decontamination of the agricultural products and foodstuffs of vegetable origin, as well as the contamination-reducing influence of some phases of food production, were presented. The most important statements are as follows:

1. As the majority of the mineral substances is taken up by the plants through their root from the soil, first of all the contamination of soil, its physical and chemical

parameters determine the amount of radioactive isotopes accumulated in plants. Because of the close and dynamic interaction between soil and plant, it is not only the soil that influences the radioactivity of the plants living on it (or in it), but also the vegetation influences the contamination of the soil (e.g. plant transpiration influences the motion of water, which in turn influences the migration of radioactive isotopes).

2. Using proper agrotechnical and agrochemical methods (e.g., application of artificial fertilizers, irrigation) the amount of radioactive isotopes passing from the soil to the plants can be significantly decreased. For instance, with liming, i.e., with decreasing the Sr:Ca proportion, the plants take up smaller amounts of ^{90}Sr isotopes from the soil.

3. Plants can take up radioactive substances, like other mineral substances, not only through the roots but also directly through the leaves. The majority of ^{137}Cs is taken up by the plants through the leaves, as the soil absorbs ^{137}Cs very well; thus it is not readily available for the plants. In contrast, most of ^{90}Sr is usually taken up from the soil.

4. At present, the majority of the radioactive contamination of plants is caused by ^{90}Sr and ^{137}Cs isotopes; the share of the activities of the other radioactive substances (e.g., ^{95}Zr , ^{140}Ba , ^{131}I , ^{239}Pu) is rather low. The contamination level of ^{90}Sr and ^{137}Cs in the plants is usually of the same order of magnitude, or else the ^{137}Cs concentration can be several times higher than the ^{90}Sr activity. At present the level of contamination is low, the measurable activities of radioactive isotopes in plant tissues indicating artificial contamination is generally by one order of magnitude lower than the activity of the natural potassium (^{40}K) content.

The maximum values of contamination of the plants due to nuclear weapon tests could be measured in European countries between 1961 and 1964. In 1986, the Chernobyl disaster caused serious contamination (mostly ^{131}I , ^{137}Cs and ^{134}Cs isotopes), which made the application of special decontamination methods necessary. In some countries large quantities of food (e.g. vegetables) had to be destroyed.

5. The level of contamination is influenced by the species and the vegetation period of the plants, as well as by the stage of plant development at the time of the peak radioactive pollution. For example, young leaves take up radioactive substances more quickly than the older ones do.

6. The contamination levels of the different parts of plants are not the same. This is due, on the one hand, to their different chemical composition, and on the other hand, to ion transport from the vegetative to the reproductive organs, as well as from the root to the above-ground parts of the plants and vice versa. The mobility of ^{90}Sr and ^{137}Cs , of paramount importance in the contamination, is less than that of Ca and K, considered to be inactive carriers. Thus during ion transport the proportions Sr:Ca and Ca:K decrease. The smallest contamination level is measured in the reproductive organs.

7. During ion transport from the soil to the plant, the concentrations of the radioactive isotopes change. The degree of relative concentration change is indicated by the discrimination factors. When the uptake of the radioactive isotopes is from the soil, the discrimination factors in relation to Sr and Cs can be calculated as follows:

$$D_{\text{Sr}} = \frac{{}^{90}\text{Sr/Ca in the plants}}{{}^{90}\text{Sr/Ca in the soil}} = \frac{\text{Sr/Ca in the plants}}{\text{Sr/Ca in the soil}}$$

$$D_{\text{Cs}} = \frac{{}^{137}\text{Cs/K in the plant}}{{}^{137}\text{Cs/K in the soil}} = \frac{\text{Cs/K in the plant}}{\text{Cs/K in the soil}}$$

8. During processing the agricultural products and food production, the degree of radioactive contamination decreases because of washing and leaching. Although at the present level of contamination there is no need to apply special decontamination methods, knowledge of the degree of decontamination during the various phases of processing and operations in the food industry may be useful when plant raw materials of higher levels of contamination are to be prepared as food.

2.5 Radioactivity of animal organisms and tissues, and factors influencing their levels of radioactivity

Animals exist in close relationship with their environment. Because of their highly complex and dynamic interactions with plants, soil, atmosphere and hydrosphere, the radioactive isotopes are continuously exchanged among the components of the biosphere and their concentration changes continuously under the influence of the various biotic and abiotic factors. The radioactive contamination of animal organisms and the migration of the different radioactive isotopes through the food chain into the animal tissues is determined mainly by the contamination level of the vegetation. This is true even for carnivorous animals, in spite of the fact that, as we will see later, there is a significant difference between herbivorous and predatory animals regarding contamination by certain radioactive isotopes. The amount of radioactive isotopes accumulated in the body of carnivorous animals is also determined after all by the level of contamination of the vegetation, the herbivorous animals being at the penultimate stage in the food chain, and the radioactivity level of their somatic tissues varies as a function of the radioactivity of the vegetation. The amount of radioactivity received by animals from the air or drinking water, is usually negligible or significantly lower compared to the contamination by plant materials consumed as fodder or food.

Animals form a close unit with their environment, so it is not just the environment that has an influence on the radioactivity of the animals and their tissues—the animals also affect their environment. That means that the animals have an influence on the contamination level of the habitat where they live. This influence can be direct (e.g. the substances—urine, excrement—leaving the animal organism start again their biological cycle) or indirect (e.g. the soil-loosening activity of earthworms influences the water balance of the soil which in turn can modify the migration of the radioactive isotopes in the soil). Finally, radioactive contamination of the food of animal origin consumed by humans determines, together with the agricultural products and the

foodstuffs of vegetable origin, the amount of radioactive substances passing into humans and the contamination of the different tissues of the human body.

During the biological chain process, with the matter transport from one component of the food chain to the other, the relative concentrations of the chemical elements change. The degree of change is determined by several factors. Considering the radioactivity in the plant-animal or rather the fodder-animal system, the discrimination ability of animals plays a decisive role, that is, the rate of assimilation of a given radioactive isotope from the fodder or food into the tissues of the animal in question. As we will see later, the discrimination factor characteristic for the animals is very different for the various chemical elements. For example, the proportion Sr:Ca—because of the very good discriminating ability of the animal organisms for strontium—is lower by one order of magnitude in the body than in the fodder. However, there is no similar effect for Cs. Apart from its physiological importance, knowledge of the discrimination factors is interesting and necessary because in this way the radioactivity of animal somatic tissues can be assessed on the basis of the radioactive contamination of the vegetation, or rather the fodder.

Just as in the other elements of the biosphere, the radioactive isotopes incorporated into animal bodies can be classified into two groups. The first group contains the radioactive isotopes of natural origin, i.e., those which are present in the biosphere independently of any human activity. The second group is formed by the radioactive matters of artificial origin; they contaminate the animals through the air-water-soil-plant chain, from nuclear explosions, as well as from the radioactive emissions of different nuclear facilities. Of course, a certain part of the contamination, as we have mentioned earlier, may also originate from drinking water or inhaled air.

Of course animals receive radiation not only from the radioactive isotopes incorporated into their own bodies, but also from the radiation load originating from their immediate environment (soil, plant, air). The environmental radiation load is usually of the same order of magnitude as the internal radiation load due to the incorporated radioactive substances, unless the natural radioactivity of the soil is substantially higher than the average due to special geological circumstances.

In this subchapter the buildup of natural and artificial radioactivity in animals will be discussed, as well as the factors influencing the contamination and the level of radiation pollution in the fodder-animal system. The possibilities will also be discussed by which the contamination of the foods of animal origin may be decreased.

As regards the different radionuclides, the importance of discrimination between plants and animals and the connection between the biological half-life and the change of contamination level will be stressed. Incidentally, only knowing these factors is it possible to assess the radioactive contamination of the different animal tissues as a function of the radioactive pollution of the plants, or rather that of the fodder. As the radioactive substances may pass from animal tissues into human tissues carried by foods of animal origin, the application of decontamination methods may be of paramount importance in an emergency. During food processing, the radioactive contamination of the ingredients may be decreased in the final product.

2.5.1 Natural radioactivity of animals

Among the radioactive substances of natural origin, not considering extreme contamination conditions, the ^{40}K isotope occurs in the largest quantity in animal organisms. This is due to the fact that potassium is an essential macroelement for animals so it occurs in the body of every animal in a relatively high concentration. In animals' bodies, potassium of the mineral elements occurs in the highest amount next to calcium. It can be found in the organism mostly in the muscles and in the inner organs, in the bones the K content is significantly lower.

In addition to potassium, animals also contain rubidium (and its radioactive isotope, ^{87}Rb), which is chemically very similar to potassium, but it is not an essential element. Rubidium is a typical microelement, it usually occurs in the mg/kg concentration range within the animal tissues (Schormüller 1965). For example, the rubidium content of the milk powder sample (IAEA A-8) used as standard reference material in activation analysis is 36 ± 3 mg/kg (Schelenz 1977). The radiation load in the animal tissues due to the radiation of the incorporated ^{87}Rb is usually smaller by 1–2 orders of magnitude than the dosage due to the radiation of the ^{40}K isotope occurring in the potassium (UNSCEAR 1977).

Considering the internal radiation load of the animal somatic tissues, mostly the bones, the incorporated and accumulated amounts of ^{226}Ra , ^{210}Pb and ^{210}Po are important factors. Uranium also accumulates in the bones and teeth. According to a communication by Belova and Lunkina (1967), the uranium content of animal bones is between $6 \cdot 10^{-6}$ and $88 \cdot 10^{-6}$ g/kg, and the value of the ^{226}Ra activity varies between 0.77 and 1.87 Bq/kg. Homma and Murakami (1977) have measured the activity of ^{226}Ra in eggs and pig bones. The average results were 0.48 and 11.5 mBq/g, respectively.

The ^{210}Pb and ^{210}Po activities of cattle bones are given in Table 58 (Yermolayeva-Makovskaya, Litver 1978). To make the data comparable, the table also lists the ^{210}Pb and ^{210}Po activities of certain fodders. It can be stated that there is no significant difference between the bones from different parts of the animal body, but, the radioactivity of the bones increases as a function of time. This can be

Table 58 ^{210}Pb and ^{210}Po activity in cattle bones and in certain fodders

Age	Activity, Bq/kg					
	^{210}Pb		^{210}Po		$^{210}\text{Po} : ^{210}\text{Pb}$	
	upper jaw-bone	dorsal vertebra	upper jaw-bone	dorsal vertebra	upper jaw-bone	dorsal vertebra
< 1 year	2.7 ± 0.3	2.3 ± 0.1	2.8 ± 0.4	2.7 ± 1.1	1.0	1.2
3–7 years	17 ± 2	11 ± 2	10 ± 3	8 ± 2	0.6	0.7
> 8 years	18 ± 1	13 ± 2	10 ± 1	8 ± 2	0.6	0.6
Hay	22 ± 2		11 ± 3		0.5	
Trefoil	32 ± 1		14 ± 1		0.4	

explained partly by the dietary differences between the young and the older animals (the ^{210}Pb activity of milk is lower than that of the fodder), and partly by the ash content of the bones in young animals being lower than in the grown-up, older ones. Incidentally, lead is incorporated into the crystal lattice of the apatite building up the bone tissues, and there it partly replaces calcium.

Bunzl et al. (1979, 1980) have examined the amount of ^{210}Pb and ^{210}Po accumulated in the liver and kidney of cattle raised in agricultural regions, the ^{210}Pb concentrations were 0.72 ± 0.27 Bq and 1.67 ± 0.48 Bq in the liver and kidney, respectively, the ^{210}Po concentrations were 1.60 ± 0.95 Bq and 6.58 ± 1.72 Bq calculated for 1 kg original matter. It is fairly clear that the ^{210}Po activity is higher than that of ^{210}Pb , and expressed in mass units, the kidney indicates significantly higher ^{210}Pb and ^{210}Po activities than the liver. However, if the activity was expressed in the amount of inactive lead, the concentration of Pb was 0.12 ± 0.04 mg/kg in the liver and 0.35 ± 0.13 mg/kg in the kidney, so it can be seen that the ^{210}Pb activity in the two organs is at approximately the same level. The measurements indicate that lead (which has toxic effects similarly to some other heavy metal trace elements) accumulates in the kidney in higher amounts than in the liver (Kreuzer et al. 1978).

The ^{210}Pb and ^{210}Po activities of seawater and marine animals depends mostly on the depth of the water, as well as on the distance from the continents. The deeper the water layer in question, the more independent is the chemical composition and the ^{210}Pb activity of the seawater of the amount of radioactive substances entering the surface layers by precipitation. The activity decreases as a function of the distance from the coast (Yermolayeva-Makovskaya, Litver 1978). Some informative data about the ^{210}Pb content of sea fishes and mammals can be found in Table 59.

Table 59 ^{210}Pb activity of body tissues in marine animals, in Bq/kg units

Animal species	Locality	Tissue	^{210}Pb activity Bq/kg	References
Cod	Atlantic Ocean	muscle	0.33 ± 0.16	Yermolayeva-Makovskaya 1968
Cod	North Sea	muscle	0.94 ± 0.25	Yermolayeva-Makovskaya 1968
Grey whale	Bering Sea	bone	2.51 ± 0.47	Sokolova, Parchevskiy 1972
Flying fish	Bering Sea	bone	3.51 ± 0.11	Sokolova, Parchevskiy 1972
Herring	North Sea	muscle	0.06 ± 0.01	Teplik, Ramzayev 1973
Hake	North Sea	muscle	0.07 ± 0.01	Teplik, Ramzayev 1973

In addition to the radioactive isotopes discussed earlier, the isotope ^{14}C must be mentioned. Its effect in the internal radiation load of the animal organisms is not negligible, because as the concomitant of the stable ^{12}C isotope, radiocarbon is present in every plant tissue used as fodder, and consequently in every animal somatic tissue. The biological influence of ^{14}C is related to the radiation and transmutation effects (Vasilenko et al. 1982). The transmutation effect is especially important as ^{14}C is also present in the DNA and RNA molecules.

2.5.2 Radioactive contamination of animal somatic tissues

Among terrestrial animals, the radioactivity of somatic tissues is determined first of all by the radioactivity of the food consumed. For aquatic animals, the radioactivity of the water environment is also important.

Under the present contamination levels, the majority of the radioactive contamination of the animal tissues is due to the activity of the ^{90}Sr and ^{137}Cs isotopes. (However, in consequence of the Chernobyl disaster, in 1986 contamination by ^{131}I and ^{134}Cs also increased significantly, and by ^{90}Sr to a lesser extent.) At present, the concentration or rather the share of activity of the other radioactive isotopes (e.g., ^{89}Sr , ^{140}Ba , ^{134}Cs , ^{129}I , ^{131}I , ^{239}Pu) is usually significantly smaller. As animal somatic tissues contain even the inactive isotopes of the radioactive strontium and cesium, and since the uptake and accumulation of radioactive isotopes is also influenced by the concentration of these stable isotopes, it is worth paying some attention to strontium and cesium as typical microelements in the animal somatic tissues.

Strontium probably plays a well-defined biological role in the animal (and human) organism. For example, in rats living on strontium-deficient diet, bone weakness appears, indicating that strontium contributes to the mechanical stability and hardness of the bones and teeth. A relatively high amount of strontium is taken up by animals from plants because, as we explained in the earlier sub-chapter, the concentration of strontium in vegetation is usually lower by two orders of magnitude than that of Ca. However, in plants growing on soil of high strontium content, the Sr:Ca proportion can even reach 10%. Because of the discriminating ability of animal organisms for Sr, the Sr:Ca proportion in animal somatic tissues is lower than in vegetation, or in the fodder consumed by them. For instance, according to our X-ray fluorescence measurements (Szabó et al. 1984), the Sr content of sheep bones in the different parts of the body had a value of 0.76 ± 0.05 mg Sr/g ash, and at the same time the Ca content of the ash was $37.8 \pm 0.9\%$.

Cesium occurs in plant and animal somatic tissues in a concentration much lower than that of strontium. This is due, on the one hand, to the Cs concentration in the soils being usually much lower than the Sr concentration, and on the other hand, to the fact that soil absorbs cesium very well, so the plants can take up only a very small amount of cesium through the root. This is the reason why the Cs concentration of animal organisms and tissues is rather low, with a value usually under 0.1 mg/kg. According to our present knowledge of cesium, it is not an essential chemical element for animal (and human) organisms.

There is a rich literature on the ^{90}Sr and ^{137}Cs content of different animal somatic tissues and foods of animal origin. This is so because in most countries of the world, with the awareness of the contaminating effect and the danger of radioactive isotopes, during the past three decades there have been systematic investigations of the radiation contamination, the results of which are regularly published in various journals of agronomy, food industry, chemistry, radiation biology, ecology, nuclear technology, etc.

Table 60 summarizes the results of milk measurements carried out by the Radiological Control Network of the Hungarian Ministry of Agriculture and Food in 1980. In

order to show how much of the radioactive isotopes in the milk originate from the fodder plants, in addition to the data of milk samples, the results of the fodder sample measurements are listed in the table.

The data in Table 60 demonstrate that the radioactive contamination of milk, and animal somatic tissues in general, was low in 1980; most of the gross β -activity (here on the average 83.8%) was due to the natural ^{40}K isotope. The share of the av-

Table 60 Results of milk and fodder radiometric investigations carried out by the RCN in 1980

Characteristic	Milk	Fodder
Number of samples	171	171
Total β -activity		
mBq/100 g milk or rather		
mBq/g dry material	4765 ± 438	748 ± 250
^{40}K activity		
mBq/100 g milk or rather		
mBq/g dry material	3991 ± 444	593 ± 230
Metal ion-fraction activity		
mBq/100 g milk or rather		
mBq/g dry material	62 ± 127	40 ± 18
Metal ion-fraction activity		
mBq/g Ca	480 ± 490	3569 ± 1732
^{90}Sr activity		
mBq/100 g milk or rather		
mBq/g dry material	20 ± 16	20 ± 9
^{90}Sr activity		
mBq/g Ca	173 ± 135	1599 ± 374
^{137}Cs activity		
mBq/100 g milk or rather		
mBq/g dry material	93 ± 46	11 ± 9
^{137}Cs activity		
mBq/g K	825 ± 442	580 ± 358

Table 61 The radioactive contamination of milk in the region of Berlin between 1962 and 1970

Year	^{90}Sr activity	
	mBq/l	mBq/g Ca
1962	255 ± 59	218 ± 38
1963	603 ± 154	503 ± 128
1964	551 ± 58	459 ± 50
1965	488 ± 40	407 ± 31
1966	370 ± 42	303 ± 34
1967	278 ± 28	241 ± 20
1968	244 ± 21	215 ± 14
1969	185 ± 15	155 ± 17
1970	181 ± 18	152 ± 15

Table 62 ^{90}Sr contamination of milk and meat in the Soviet Union between 1963 and 1973

Year	^{90}Sr activity, Bq/kg or rather Bq/l						
	milk	beef	lamb	pork	reindeer meat	sea fish	freshwater fish
1963	1.11 $\pm$ 0.03	0.89 $\pm$ 0.07	1.11 $\pm$ 0.11	0.33 $\pm$ 0.11	7.4 $\pm$ 2.1	—	2.3 $\pm$ 0.3
1964	1.15 $\pm$ 0.03	0.85 $\pm$ 0.15	1.26 $\pm$ 0.33	0.37 $\pm$ 0.09	3.0 $\pm$ 0.6	2.15 $\pm$ 0.15	4.4 $\pm$ 0.4
1965	0.70 $\pm$ 0.02	0.48 $\pm$ 0.03	0.85 $\pm$ 0.09	0.30 $\pm$ 0.02	1.6 $\pm$ 0.4	0.96 $\pm$ 0.02	3.4 $\pm$ 0.4
1966	0.52 $\pm$ 0.01	0.32 $\pm$ 0.03	0.60 $\pm$ 0.10	0.25 $\pm$ 0.02	2.8 $\pm$ 0.9	0.81 $\pm$ 0.09	5.0 $\pm$ 0.7
1967	0.37 $\pm$ 0.01	0.31 $\pm$ 0.02	0.33 $\pm$ 0.04	0.26 $\pm$ 0.02	2.7 $\pm$ 0.9	0.74 $\pm$ 0.08	4.3 $\pm$ 0.4
1968	0.34 $\pm$ 0.01	0.26 $\pm$ 0.01	0.28 $\pm$ 0.02	0.21 $\pm$ 0.02	3.5 $\pm$ 1.5	0.89 $\pm$ 0.07	5.1 $\pm$ 0.3
1969	0.30 $\pm$ 0.01	0.21 $\pm$ 0.01	0.22 $\pm$ 0.02	0.20 $\pm$ 0.01	3.1 $\pm$ 0.3	0.58 $\pm$ 0.04	2.3 $\pm$ 0.2
1970	0.32 $\pm$ 0.01	0.23 $\pm$ 0.02	0.27 $\pm$ 0.02	0.27 $\pm$ 0.04	3.7 $\pm$ 0.7	1.00 $\pm$ 0.15	2.2 $\pm$ 0.3
1971	0.33 $\pm$ 0.01	0.23 $\pm$ 0.02	0.34 $\pm$ 0.03	0.21 $\pm$ 0.02	2.7 $\pm$ 0.3	0.63 $\pm$ 0.07	1.9 $\pm$ 0.2
1972	0.30 $\pm$ 0.02	0.20 $\pm$ 0.01	0.25 $\pm$ 0.03	0.16 $\pm$ 0.02	1.8 $\pm$ 0.6	0.84 $\pm$ 0.25	2.7 $\pm$ 0.6
1973	0.25 $\pm$ 0.01	0.19 $\pm$ 0.02	0.25 $\pm$ 0.03	0.18 $\pm$ 0.03	0.9 $\pm$ 0.1	0.87 $\pm$ 0.36	2.7 $\pm$ 0.4

Table 63 ^{137}Cs contamination of milk and meat in the Soviet Union between 1963 and 1973

Year	^{137}Cs activity, Bq/kg or rather Bq/l						
	milk	beef	lamb	pork	reindeer meat	sea fish	freshwater fish
1963	7.77 $\pm$ 0.11	10.63 $\pm$ 2.52	10.73 $\pm$ 2.22	—	230 $\pm$ 40	—	3.33 $\pm$ 0.85
1964	5.92 $\pm$ 0.19	12.95 $\pm$ 0.21	36.63 $\pm$ 9.55	6.66 $\pm$ 1.51	1630 $\pm$ 290	—	—
1965	2.89 $\pm$ 0.09	9.25 $\pm$ 0.48	12.80 $\pm$ 1.26	7.10 $\pm$ 0.44	1180 $\pm$ 220	17.39 $\pm$ 2.74	33.30 $\pm$ 3.70
1966	2.07 $\pm$ 0.04	7.77 $\pm$ 0.63	6.77 $\pm$ 1.11	6.29 $\pm$ 0.48	929 $\pm$ 141	11.10 $\pm$ 1.18	25.16 $\pm$ 3.33
1967	1.41 $\pm$ 0.03	5.03 $\pm$ 0.22	3.70 $\pm$ 0.41	4.33 $\pm$ 0.26	799 $\pm$ 297	4.92 $\pm$ 0.52	22.35 $\pm$ 1.92
1968	1.11 $\pm$ 0.07	2.89 $\pm$ 0.26	1.81 $\pm$ 0.26	2.18 $\pm$ 0.19	418 $\pm$ 44	1.70 $\pm$ 0.19	14.06 $\pm$ 3.33
1969	0.74 $\pm$ 0.03	1.63 $\pm$ 0.06	1.74 $\pm$ 0.15	1.59 $\pm$ 0.19	140 $\pm$ 18	1.15 $\pm$ 0.07	5.29 $\pm$ 0.85
1970	0.85 $\pm$ 0.06	2.07 $\pm$ 0.19	2.22 $\pm$ 0.15	1.81 $\pm$ 0.19	318 $\pm$ 31	2.52 $\pm$ 0.15	5.11 $\pm$ 0.44
1971	1.00 $\pm$ 0.11	2.07 $\pm$ 0.08	1.89 $\pm$ 0.19	1.48 $\pm$ 0.07	189 $\pm$ 16	1.30 $\pm$ 0.11	5.14 $\pm$ 0.48
1972	0.78 $\pm$ 0.22	1.59 $\pm$ 0.30	1.52 $\pm$ 0.15	1.22 $\pm$ 0.07	155 $\pm$ 25	1.04 $\pm$ 0.19	2.66 $\pm$ 0.37
1973	0.67 $\pm$ 0.15	1.04 $\pm$ 0.15	0.81 $\pm$ 0.11	0.96 $\pm$ 0.22	176 $\pm$ 25	1.30 $\pm$ 0.48	2.81 $\pm$ 0.33

erage activity of the metal ion-fraction was as low as 1.3%, the ^{90}Sr activity was 0.4%, the ^{137}Cs activity was 2.0% of the gross activity. These proportions are of course not characteristic for the bones, because the majority of their radioactive contamination is due to the ^{90}Sr content. The reason is that the mineral substance of the bone tissues is mostly hydroxylapatite with a composition of $\text{Ca}_3(\text{PO}_4)_2 \cdot \text{Ca}(\text{OH})_2$, and strontium is accumulated together with calcium.

Let us now see some data compared in Tables 61, 62 and 63. Table 61 summarizes the ^{90}Sr activity in milk samples taken from the environs of Berlin, on the basis of the data of Moldenhawer and Schulze-Wollgast (1971) and Schulze-Wollgast (1972). It is apparent that the maxima in contamination occurred in 1963–1964.

In Tables 62 and 63 the ^{90}Sr and ^{137}Cs contamination of meat and milk samples taken in the Soviet Union can be seen (Marey 1980). It is remarkable that the

radioactive contamination of the meat of reindeer was several times higher than that of the meat of other animal species. Considering the ^{137}Cs content, the difference is two orders of magnitude. The reason is that lichens, mosses and other Arctic plants are the main food of reindeers, and these plants accumulate the ^{137}Cs isotopes to a greater extent. The data in the tables demonstrate that the radioactive pollution after the contamination maxima gradually decreased between 1965 and 1968, and since 1969 a relatively low level of contamination has been observed. The process of decrease or rather stabilization went on until 1986. In consequence of the Chernobyl disaster, the contamination level increased significantly over almost all Europe.

It is not always necessary to kill an animal in order to study its somatic tissues. Sometimes, using measuring techniques with proper sensitivity, *in vivo* examinations can be carried out. For example, Rabon and Johnson (1973) have developed a quick measuring method (^{137}Cs determination) for the field measurement of somatic tissues of deers living in the environs of a nuclear power plant in South Carolina (USA).

In Table 64 the results of our radiometric measurements of milk samples taken in Győr-Sopron county (Hungary) in 1968–1980 are summarized. It can be stated that the activity of the metal ion-fraction indicating the radiostrontium pollution of milk is significantly lower than the value measured in the vegetation. The decisive part of the gross β -activity (on the average 90.3%) is due to the activity of the ^{40}K isotope, the activity of the metal ion-fraction is by two orders of magnitude smaller (on the average 1.4%). Thus the standard deviation of gross β -activity (16.8%) is determined by the standard deviation of ^{40}K (9.3%). Although the standard deviation

Table 64 Average radioactivity of milk samples collected in Győr-Sopron county (Hungary) between 1968 and 1980

Year	Activity, mBq/100 g milk			Metal ion-fraction activity mBq/g, Ca
	Total- β	^{40}K	Metal ion-fraction	
1968	5125	4647	67	555
1969	5088	5910	41	348
1970	5210	4991	133	1206
1971	5254	5099	70	596
1972	4829	4540	122	1080
1973	4669	4192	52	470
1974	4603	4425	74	781
1975	4514	4089	111	1003
1976	4832	4103	52	522
1977	5047	4237	30	296
1978	4577	3811	37	448
1979	5183	4506	33	254
1980	4867	4089	102	892
Average	4909 ± 824	4434 ± 413	67 ± 36	648 ± 296
CV (%)	16.8	9.3	53.7	45.7

of the activity of the metal ion-fraction is rather high (a value over 50%), this is not a dominant factor in the yearly changes of gross activity.

We have already mentioned that in, addition to the ^{90}Sr and ^{137}Cs isotopes contributing the most to radioactive contamination, other radionuclides occur in animal somatic tissues and in foods of animal origin. The appearance of fission products of relatively short half-life can be expected during the weeks and months immediately after a nuclear explosion or nuclear facility disaster. For example, according to measurements by Wiechen (1970), in 1961–1962 the activity of ^{89}Sr and ^{140}Ba in milk was several times higher than the activity of ^{90}Sr .

Occasionally the radioactive isotopes of iodine can also cause significant contamination. Although ^{131}I is dangerous only within a few weeks of the explosion or disaster because of its relatively short half-life, the radioiodine reaching the plants and entering the surface waters through fall-out, may soon be grazed by cattle and excreted with the milk (Wiechen 1970). The 4–20%, on average the 8%, of the ^{131}I isotope taken up daily by cows from fodder and drinking water is excreted with the milk. The majority of ^{131}I reaches the thyroid gland, where the accumulated iodine, in spite of its short half-life, can cause serious danger, as radioactive iodine has a relatively high radiation energy and the isotope remains in the thyroid gland for practically all its radioactive life. In the environs of reprocessing plants ^{129}I activ-

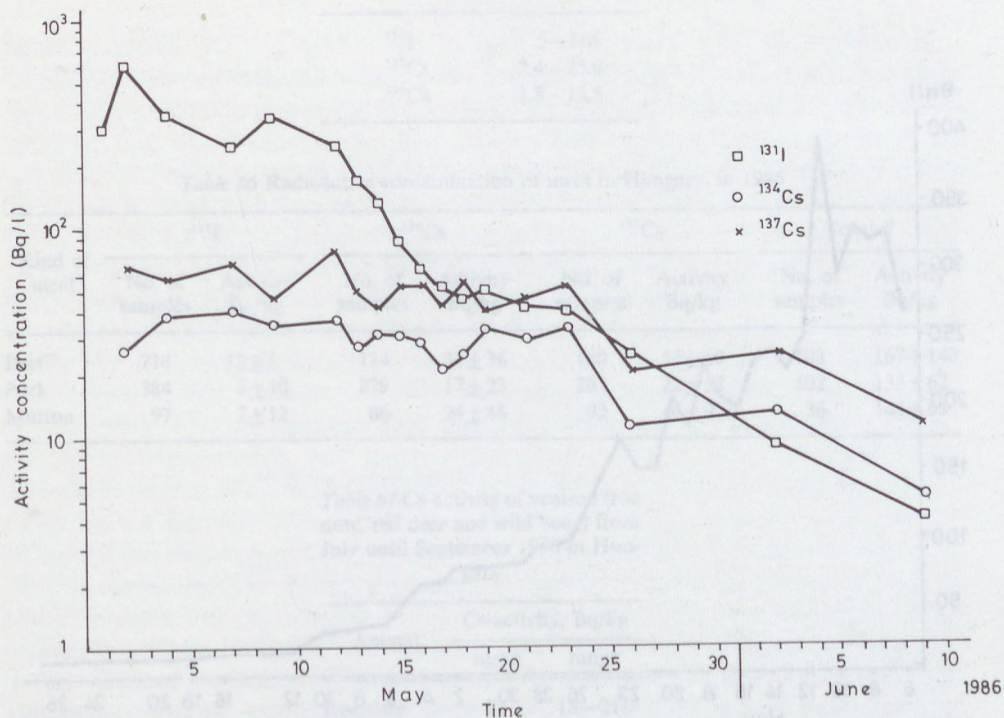


Fig. 24. Time dependence of activity concentration in milk from a cow grazing on pastures in Nógrád county

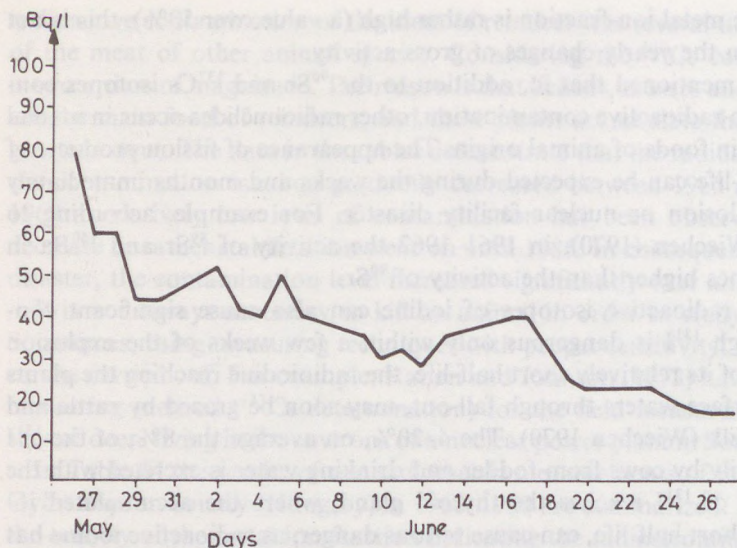


Fig. 25. The average daily $^{134}\text{Cs} + ^{137}\text{Cs}$ activity of cow milk in the Hungarian milk industry (May and June 1986)

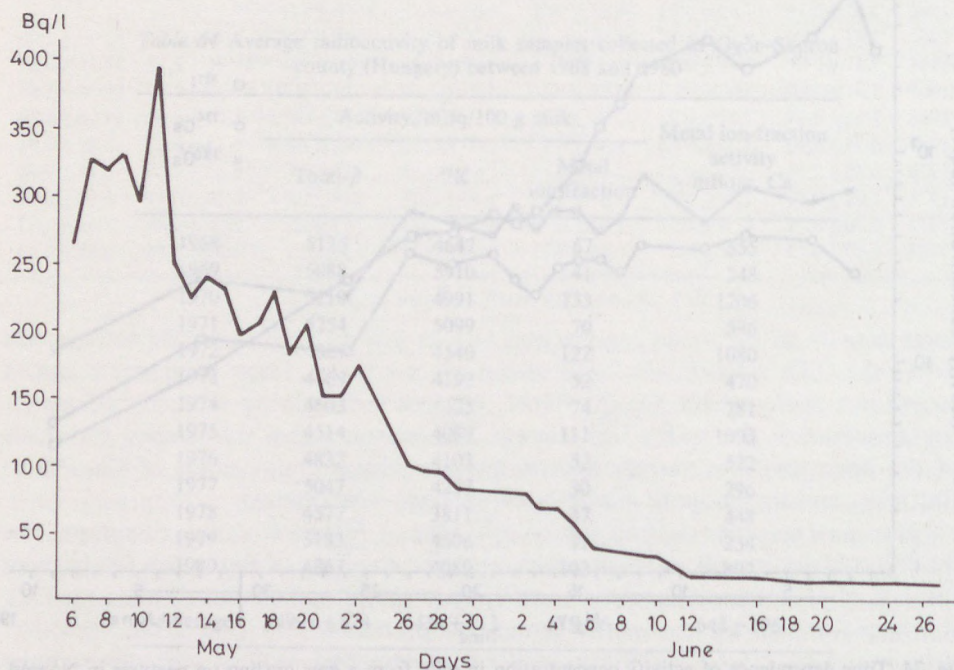


Fig. 26. The average daily ^{131}I activity of cow milk in the Hungarian milk industry (May and June 1986)

ity is significantly higher than the background radiation, because the majority of the emitted ^{129}I passes into the water and into the atmosphere, and from there into living organisms. For example, in the 5 km radius surrounding the reprocessing centre of West Bell, the ^{129}I activity of the thyroid glands of cattle was about $4 \cdot 10^3$ Bq/g iodine, and this value is higher by 3–4 orders of magnitude than the natural background radiation due to the activity of ^{129}I produced from xenon in the atmosphere under the influence of cosmic radiation. Its value is 0.8–2.6 Bq/g iodine (Zikova, Gneusheva 1976).

Kracke and Bunzl (1980) investigated the $^{239,240}\text{Pu}$ content of animal somatic tissues. According to the measurements carried out in the agricultural regions of West Germany, the $^{239,240}\text{Pu}$ content of cattle liver was 0.89 mBq/kg original tissue, the 95% confidence interval being 0.59 and 1.00 mBq. The ^{137}Cs activity was also measured, its value was 0.44 Bq, the 95% confidence interval being 0.07 and 0.59 Bq.

The water content of animal somatic tissues is usually rather high, and since the water also contains tritium, ^3H contamination of the somatic tissues should be

Table 65 Radioactivity of milk in Belgium, 5–23 May 1986

Radioactivity isotope	Activity, Bq/l
^{131}I	5–166
^{137}Cs	2.4–25.0
^{134}Cs	1.8–13.5

Table 66 Radioactive contamination of meat in Hungary, in 1986

Kind of meat	^{131}I		^{134}Cs		^{137}Cs		Total- β	
	No. of samples	Activity Bq/kg	No. of samples	Activity Bq/kg	No. of samples	Activity Bq/kg	No. of samples	Activity Bq/kg
Beef	214	13 ± 1	174	31 ± 36	180	53 ± 60	503	167 ± 140
Pork	384	9 ± 10	279	17 ± 23	292	22 ± 38	102	135 ± 62
Mutton	97	7 ± 12	86	24 ± 46	92	46 ± 92	36	149 ± 83

Table 67 Cs-activity of venison (roe deer, red deer and wild boar) from July until September 1986 in Hungary

Animal	Cs-activity, Bq/kg	
	mean	range
Roe deer	85	18–211
Red deer	60	14–177
Wild boar	35	23–49

considered as well. For instance, in 1974 in the USA the ^3H activity of milk was $9.99 \pm 7.77 \text{ Bq/l}$ (Milk and Food 1974). However, the energy of the radiation emitted by ^3H is extremely low, so the radiation load due to the tritium content is usually negligible (Bennett 1973).

Data about the changes of radioactive contamination of milk in Hungary during the weeks after the Chernobyl disaster have been compiled in Figures 24, 25 and 26. Figure 24 shows the contamination of milk from a cow grazing on pastures in Nógrád county (Andrási et al. 1986). Figures 25 and 26 show the daily average radiocesium ($^{137}\text{Cs} + ^{134}\text{Cs}$) and radioiodine (^{131}I) activity in milk distributed by the state milk industry. The most important emergency measure taken was the limitation of grazing by the milk-producing animals, and feeding them fodder harvested earlier. Comparable data from Belgium are listed in Table 65 (Deworm 1986).

The Radiological Control Network in Hungary also carried out the analysis of a great number of meat samples. The data for 1986 can be seen in Table 66. In Table 67 the Cs activities of venison are listed.

2.5.3 The relationship between radioactive contamination and the animal species, and the discriminative capability of animal organisms

In addition to the contamination level of the fodder and the drinking water consumed, the radioactivity of animal somatic tissues and therefore of foodstuffs of animal origin is determined by the species and the age of the animal. During physiological processes, when the animal builds up its own tissues from fodder and drinking water, the concentrations of the different radioactive isotopes change, i.e., the radioactive contamination of the fodder and the animal somatic tissues is not the same. There is a great difference between the animals, discriminating ability for Sr and Cs, as the different animal organisms, in a slightly different degree according to their species, discriminate very well in the case of Sr, but they may even increase the Cs content. The value of the discrimination factor of cow milk/fodder is usually around $D=0.13$ for ^{90}Sr ; the value of the discrimination factor rat muscle/fodder for ^{137}Cs is between 1.0 and 5.3 (Wasserman, Comar 1961).

The calculation of the discrimination factor is as follows:

$$D_{\text{Sr}} = \frac{{}^{90}\text{Sr}/\text{Ca in the animal tissue or milk}}{{}^{90}\text{Sr}/\text{Ca in the fodder}}$$

$$D_{\text{Cs}} = \frac{{}^{137}\text{Cs}/\text{K in the animal tissue or milk}}{{}^{137}\text{Cs}/\text{K in the fodder}}$$

When the discriminating ability of animal organisms, on the basis of the radioactivity of fodder, is known the contamination level in the different animal somatic tissues can be assessed. On the basis of the radiostromium discriminating capability of cattle, we have determined the value of the discrimination factor (Szabó, Mezei 1978):

$$D_{\text{Sr}} = 0.103 + 0.019$$

This value indicates that the ^{90}Sr contamination referred to the Ca content in the milk is lower by one order of magnitude than in the fodder. Csupka (1970) reported similar results. According to Csupka there is a relationship between the ^{90}Sr contamination of milk and fodder as follows:

$$A = k_1 F_1 + k_2 F_2$$

where:

- A = the ^{90}Sr activity of milk expressed for 1 g of Ca;
- F_1 = the ^{90}Sr activity of fodder plant expressed in g of Ca;
- F_2 = average ^{90}Sr fall-out activity for $\text{m}^2 \cdot \text{year}$;
- k_1, k_2 = constants, the value of k_1 being the discrimination factor, 0.11.

Since there is a second constant in the formula, k_1 is only approximately the same as the discrimination factor.

Bezzegh-Galantai (1972) determined a relationship between the ^{90}Sr contamination of the grass consumed by dairy cows and that of their milk. According to his report:

$$\frac{^{90}\text{Sr activity Bq/l milk}}{^{90}\text{Sr activity Bq/kg dry matter of grass}} = 0.01-0.04$$

According to our measurements the relationship can be written as follows:

$$y = (1.079 \pm 0.174) x$$

where:

- y = ^{90}Sr contamination of milk in $\text{mBq}/100 \text{ g milk unit}$;
- x = ^{90}Sr contamination of fodder consumed by cattle in $\text{mBq/g dry matter unit}$.

It can be clearly seen that the ^{90}Sr contamination of 100 g milk is equivalent to about 1 g dry matter of the fodder. (In the units given by Bezzegh-Galantai the value of the quotient would be 0.011, that is within the given interval.)

It is necessary to stress that the function $y = (1.079 + 0.174)x$ gives only the relationship between the radioactive contamination of the milk and the fodder. It has been calculated only for cow milk and fodder used for feeding cattle, first of all hay and silage. The data do not refer, for example, to goat and sheep milk, as their chemical composition is different from cow milk. Incidentally, the radioactive contamination of sheep milk is rather different from cow milk, which is due partly to the different grazing-feeding methods, and partly to the discrimination factor of sheep being different from that of cattle. In a similar way, the relationship is not valid for fodders which have a significantly different chemical composition, especially a Ca content different from the average Ca values of fodders used for feeding cattle. For example, the ^{90}Sr contamination of the cereal (grain) fodder, a staple feed for pigs (maize, barley) expressed in dry matter is very low compared to the other fodders containing vegetative organs of the plants, just because of their low Ca content. In order to provide a general picture, Table 68 lists the Ca contents of some types of fodder. Incidentally, according to our measurements, the average Ca content of fodders used for feeding cattle expressed in dry matter was 1.26%; the average Ca content of cow milk was 0.114%.

The level of the radiostrontium contamination of animal bones is low, just like that of milk. The results of studies of bone radioactivities carried out in 1980 by the Radiological Control Network of the Hungarian Ministry of Agriculture and Food are listed in Table 69. As the K content of the bones is low (thus the occurrence of ^{137}Cs can be expected in very low concentration), only the activities of the metal ion-fraction and ^{90}Sr are listed. There are no substantial differences as regards radioactive contamination among the bones from the different parts of body, i.e., the

Table 68 Average Ca content of fodders
expressed in dry material content

Fodder	Ca, %
Alfalfa	2.0
Sunflower leaves	1.8
Green maize leaves	0.5
Hay	0.9
Beet slices from sugar factory	0.9
Maize	0.1

Table 69 Results of radiometrical bone measurements carried out by the RCN
in 1980

Sample	No. of samples	Metal ion-fraction activity		^{90}Sr activity	
		mBq/g dry material	mBq/g Ca	mBq/g dry material	mBq/g Ca
Pig	55	8 ± 6	48 ± 11	4 ± 3	24 ± 19
Cattle	60	45 ± 25	246 ± 143	31 ± 24	157 ± 123
Sheep	96	—	—	34 ± 13	225 ± 89
Fish	73	81 ± 51	701 ± 572	—	—

degree of the radioactive isotope accumulation can be considered the same. This is also demonstrated by the data in Table 70, which lists the average results of radiometrical measurements on femur (thigh bone), metacarpus and costa (rib) carried out in Hungary in 1972.

We can also say on the basis of the data in Table 69 that the radioactive contamination of different animals is not the same. The differences are due partly to the different feeding methods and partly to the different discriminating abilities of the various species of animals. Analysing different animal bones, we could state, that the level of radioactive contamination in pig and in poultry are the lowest, the values in the bones of cattle being higher. The sheep bones are more contaminated, and fish contain the most contamination. Keil (1968) reported similar results; according to his measurements the amount of ^{90}Sr accumulated in the somatic tissues of fish is many times higher than the contamination measured in the tissues of poultry. We should mention that in addition to the domestic animals, we have also carried out several studies on

the amount of radioactive substances accumulated in the bones of different game species (deer, pheasant, etc.). Although, because of the relatively small number of samples, a rank order could not be ascertained, or rather their insertion into the earlier obtained sequence contamination levels going from poultry-pig-cattle-sheep-fish is not justified, it is safe to say that the radioactive contamination levels of the somatic tissues of game is usually not higher than those measured in domestic animals, or at least the degree of radioactive contamination is not many times higher. However, in

Table 70 Average metal ion-fraction activity in growing (young) cattle

Sample	Number of samples	Metal ion-fraction activity	
		Bq/g bone	Bq/g Ca
Femur	70	0.11	0.83
Metacarpus	118	0.13	0.94
Costa	67	0.13	0.85

both game and domestic animals there are significant differences among certain organs and tissues in the level of radioactive contamination because of their different chemical composition. For example, the thyroid gland, the lymphatic nodes, the liver and the kidney are more polluted than the other organs and muscle tissues (Karavayev 1969). Tataruch et al. (1988) are among those who have dealt with the question of radioactive contamination of game species.

The trend observed over many years that the radioactive contamination of sheep bones is higher than that of cattle bones, i.e., that ^{90}Sr accumulates more in sheep than in the somatic tissues of cattle, is also demonstrated by the radiometric measurements carried out for comparison of cow milk and sheep milk. According to the measurements carried out in 1980, in sheep milk the activity of the metal ion-fraction is 2.05 times, the ^{90}Sr activity 1.81 times and the ^{137}Cs activity 1.07 times higher in terms of mass unit than in cow milk. When the basis of comparison was the Ca or rather the K content, the contamination of sheep milk was 1.20 times that of cow milk for ^{90}Sr activity, and 1.35 for ^{137}Cs activity.

Among the studied animals, the radioactive pollution of fish is usually much higher than of terrestrial animals. This fact indicates that fish accumulate the radioactive substances in a much greater degree. The origin of the fish is very important, as the radioactive contamination of fish from lakes is usually higher than of fish from rivers, because of the usually higher level of contamination of lakes in general. We have observed this fact during the radiometric investigation of carp samples collected from the Lake Fertő and the River Danube. Similar results were also observed when measuring the radioactive contamination of fish from Lake Balaton and River Dráva. When studying the amount of ^{90}Sr accumulated in the tissues of fish, the inactive strontium content of the water is also an important parameter. The lower the strontium content of the water, the higher the amount of ^{90}Sr in the bodies of fish (Leynerte et al. 1977).

The radioactive contamination of fish of different species from the same locality can be very different. The difference can be explained by their different food or by their different discrimination abilities. For example the data in Table 71 demonstrate that the radioactive contamination of bream is significantly higher than that of carp.

Because the animal organisms, among them also the fish, discriminate very well against strontium, the ^{90}Sr contamination of the tissues of herbivorous fish (e.g. amur) is significantly higher than that of carnivorous fish, eating smaller fish. This can

Table 71 Average metal ion-fraction activity in samples of bream and carp taken from Lake Balaton related to g ash

Year	Sample	Metal ion-fraction activity, Bq/g ash	
		bream	carp
1970	bone	0.83	0.26
	meat	0.15	0.01
1971	bone	0.56	0.27
	meat	0.09	0.03
1972	bone	0.64	0.58
	meat	0.13	0.04
1973	bone	0.74	0.49
	meat	0.12	0.05

be explained by the predatory fish eating food of lower contamination level (because of their good discrimination) than the fish eating plants. Of course this is true only for ^{90}Sr . In the case of ^{137}Cs pollution, the situation is quite different, as we will see later.

We have already mentioned that between 1971 and 1974 we carried out an extensive survey to determine the radioactive pollution of foodstuffs which were not in the programme of the Radiological Control Network (Szabó 1977). Such measurements were also carried out for different foodstuffs of animal origin. The mean values of the measurements are listed in Table 72. We can say that the radiostrontium pollution of the animal tissues (meat, milk, egg, offal) is significantly lower than the values measured in vegetables. It must be added, that the good discrimination ability of the animal organisms was also demonstrated by the milk, fodder and bone analyses based on the inactive strontium content. According to our measurements, the Sr:Ca proportion measured in milk and bones was only 8–15% of that in the fodder consumed by the animals.

As we have seen, the discrimination factors indicating the discriminating ability of the organism show how the relative amount of the isotope or chemical element in question changes in the next stage of the biological chain during metabolism. In the Ca-Sr system the discrimination ability for Sr is very good. Physiologically it means that from the nutrient consumed by the animals much less Sr is assimilated than Ca, i.e., the majority of ^{90}Sr will not be absorbed, but is passed through the gastro-

intestinal system. Thus a larger part of Sr is defecated than urinated. Let us now see what happens with the other isotope which plays an important role in the buildup of lasting contamination, ^{137}Cs or rather the inactive Cs.

As we mentioned earlier, the discrimination factor for Cs, or rather ^{137}Cs , has a value between 1.0 and 5.3, depending on the chemical composition of the food. The Cs:K, or rather ^{137}Cs :K, proportion in the animal tissues is either the same or higher than the value characteristic of the fodder or food. It can be explained by

Table 72 Average radioactivity of foodstuffs of animal origin in Győr-Sopron county (Hungary)

Sample	Activity, Bq/g dry material		
	Total- β	^{40}K	Metal ion-fraction
Pork liver pâté	111	96	7
Hunter sausage	174	163	7
Brawn	81	67	11
Raw ham	170	166	2
Boiled ham	385	370	4
Bologna sausage	204	196	4
Chicken breast	374	352	7
Chicken thigh	400	381	4
Egg white	418	333	48
Egg yolk	89	44	33

virtually all of the Cs in the fodder being incorporated into the animal's body, i.e., the efficiency of the Cs absorption is practically 100% (Nevstruyeva et al. 1972). Incidentally, the value of a discrimination factor different from unity demonstrates the presence of active ion-transport through the cell membranes.

On the basis of radiometric studies of milk and fodder samples taken simultaneously at regular intervals, the ^{137}Cs :K, or rather the Cs:K, proportions were determined (Szabó 1983). We have found that

$$D'_{\text{Cs}} = \frac{{}^{137}\text{Cs}/\text{K in the milk}}{{}^{137}\text{Cs}/\text{K in the fodder}} = 0.90 + 0.40,$$

and

$$D''_{\text{Cs}} = \frac{\text{Cs}/\text{K in the milk}}{\text{Cs}/\text{K in the fodder}} = 0.83 + 0.18.$$

We can say that the ^{137}Cs :K, or rather the Cs:K, proportion in the fodder consumed by the cattle and in the milk is virtually the same, so the discrimination factor does not differ significantly from 1. Thus practically we cannot speak of discriminating ability for Cs. The data in Table 60 demonstrate the same: the ^{137}Cs contamination of milk and fodder in 1980 in Hungary were 825 ± 442 and 580 ± 358 mBq/g K, respectively. The value of D_{Cs} is 1.42 ± 1.16 , and based on the standard

deviations, we cannot say that there is any significant difference between the ^{137}Cs contaminations of milk and fodder.

Most of the ^{137}Cs or rather the Cs taken up by the animal body is absorbed quickly, and is accumulated mostly in the muscles together with the potassium and sodium that have a transport system in common. In cows, usually about 1–2% of the Cs taken up daily is excreted into the milk; however, in other animal species very different values can be measured. For instance, in goat the amount of Cs excreted into the milk can be as high as 9% of the total amount taken up daily.

We have mentioned when discussing radiostrontium contamination that the radioactive pollution of the somatic tissues of different animals is not the same, which can be explained partly by the different feeding habits, and partly by the unequal discrimination factors of the various animal species. This statement is also valid for radiocesium contamination. For example, on the basis of the data in Table 63, it can be asserted that ^{137}Cs contamination of venison is higher by two orders of magnitude than that of the other kinds of meat.

The data in the literature (e.g., Marinov et al. (1969), Schulze-Wollgast (1972), Marey (1980)) demonstrate that the $^{137}\text{Cs}:\text{K}$ or rather the Cs:K ratios in muscles are usually higher than the same measured in the fodder consumed by the animals. Thus the relative cesium or rather the radiocesium contamination of animal muscle tissues is either higher than the contamination of fodders or at the same level—but never lower. The ability to accumulate also depends on the K content of the fodder. The animals' organism does not discriminate for Cs against K but just the opposite, i.e., it is not the concentration of cesium that decreases compared to that of potassium, but that of potassium compared to cesium. As the Cs entering the gastro-intestinal system with the fodder is completely absorbed, the higher the K concentration of the fodder, the lower the absorption rate of K, and subsequently the proportion Cs:K is higher.

The fish accumulate ^{137}Cs very well. Since the $^{137}\text{Cs}/\text{K}$ discrimination factor in fish is greater than 1, the ^{137}Cs contamination of carnivorous species is higher than that of the herbivorous species.

In addition to ^{137}Cs , the organisms of fish accumulate other radioactive isotopes, e.g. a significant amount of radioiodine. For example, $(2-20) \cdot 10^3 \text{ Bq/g } ^{129}\text{I}$ activity was measured in the somatic tissues of fish living in rivers draining the active sewage water of a reprocessing plant, which would result in a not-negligible radiation load for a human consumer (Zikova, Gneusheva 1976).

Interesting data are given by Djuric et al. (1987) about the variations of the activity and concentration factors of ^{137}Cs in a "fish-water" system. The concentration factors were calculated as the ratio of the average ^{137}Cs activity in a fish species to the average ^{137}Cs activity of the water for one year. The results of the calculations are presented in Table 73. The average concentration factor, irrespective of the species of fish, locality of the lake or the year of sampling, was found to be 76 ± 20 . The investigations included nine lakes of different origin in Yugoslavia. It can also be concluded that in the aquatic system of this type, bream (*Abramis brama*) can be an indicator organism for the accumulation of the ^{137}Cs radionuclide. It is necessary to stress that the accumulation of radionuclides in aquatic organisms is strongly

influenced by salinity: freshwater organisms accumulate significantly more radionuclides than do the saltwater species (Lowmann 1971).

It is necessary to add that, in order to better approximate the metabolic processes, instead of using simple discrimination factors, approaches consisting of different linear and/or exponential terms have been used recently (UNSCEAR 1977, 1982, 1988). For instance, assume the following relation between the concentration of a

Table 73 Concentration factors ($F_i = A^{137}\text{Cs}_{\text{fish}}/A^{137}\text{Cs}_{\text{water}}$) for ^{137}Cs in a "fish-water" system

Year	Bream (<i>Abramis brama</i>)	Barbel (<i>Barbus fluviatilis</i>)	Roach (<i>Scardinius erythrophthalmus</i>)
1982	108	49	85
1983	108	49	85
1984	77	35	60
1985	90	41	70
Average	102	46	80

certain radionuclide in milk and the concentration of the same radionuclide in the fodder consumed by the milk-producing animal

$$c_i^M = F_m c_i^v Q_F e^{-\lambda_i t_f}$$

where:

- c_i^m = concentration of radionuclide i in the milk, Bq/l;
- c_i^v = concentration of radionuclide i in the fodder, Bq/kg dry matter;
- F_m = percentage of uptake of a chemical element, which in equilibrium occurs in unit volume of the milk, day/l;
- Q_F = amount of fodder consumed daily by the animal, kg dry matter/day;
- λ_i = decay constant of radionuclide i , day⁻¹;
- t_f = time interval (usually 4 days) needed for the radioactive isotope in the fodder to reach the consumer of milk, day.

In a similar way, one can be also calculated for example, the expected radionuclide concentration of meat. We have to mention here that in cattle, sheep and pig, the ^{90}Sr discrimination factor for muscle tissues has a value between 0.18 and 0.24; however, in the inner organs (e.g. liver, kidney) the transfer coefficient may be higher. The animals also discriminate very well for the iodine; for example, in the muscle tissue of cattle 0.15% of the daily ^{131}I uptake was found.

Measurements of the transfer coefficients for ^{137}Cs , ^{60}Co , ^{54}Mn , ^{22}Na , ^{131}I and $^{95\text{m}}\text{Tc}$ from feed to milk and beef were carried out by Voight et al. (1988). The transfer of the radionuclides was studied experimentally for 140 days, after which the cow was slaughtered.

The calculation of the transfer coefficient for milk was performed by fitting the integral

$$A(t) = \int_0^t A_o(t') \cdot R(t-t') \cdot dt'$$

Table 74 Mean transfer coefficients from feed to meat, organs, milk, and milk products of a cow (in d/kg, resp. d/t) (Voight et al. 1988)

Radionuclide:		$^{22}\text{Na} \cdot 10^{-2}$	$^{60}\text{Co} \cdot 10^{-4}$	$^{137}\text{Cs} \cdot 10^{-3}$
Organ:	rennet-bag	1.7	11	2.6
	reticulum	2.2	35	3.6
	omasum	1.8	71	3.1
	abomasum	1.5	16	3.4
	liver	1.4	71	4.3
	udder	2.8	9.5	1.9
	lungs	2.8	7.2	2.5
	brain	2.0	$\leq 2^a$	1.2
	bladder	2.4	$\leq 2^a$	1.9
	duodenum	1.6	$\leq 2^a$	1.6
	caecum	1.8	$\leq 2^a$	3.1
	large intestine	1.4	14	2.3
	small intestine	1.1	13	1.7
	spleen	1.3	$\leq 2^a$	3.1
	kidneys	3.0	24	5.3
	thymus	1.4	$\leq 2^a$	1.6
	oesophagus	2.0	$\leq 2^a$	2.4
	pancreas	1.2	$\leq 2^a$	$\leq 0.01^a$
	skin	1.9	$\leq 2^a$	4.8
Meat:	lumbar muscle	0.9	0.8	5.6
	thigh	1.0	1.1	7.8
	shoulder	1.1	1.2	6.2
	neck muscle	1.1	1.6	5.4
	fillet	0.8	$\leq 1.6^a$	6.1
	leg muscle	1.3	1.0	5.5
	pelvis muscle	0.8	1.6	7.4
	breast	1.2	$\leq 1.6^a$	4.0
	tibia muscle	1.1	1.6	6.7
Bone:	shoulder blade	5.6	2.7	2.6
	thigh	5.2	0.9	0.35
	milk	1.6	$\leq 2^a$	2.2
	cheese	1.3	$\leq 2^a$	1.7
	whey	1.7	$\leq 2^a$	2.1

to the measured data. $A(t')$ is the activity administered at time t' , $R(t')$ is the retention function which was assumed to be

$$R(t') = a_1 e^{-\lambda_1 t} + a_2 e^{-\lambda_2 t}.$$

This assumption is based on the retention function given by Coughtrey et al. (1983), without taking into account the small long-term component (retention in the skeleton), which cannot be observed during the measurement interval.

Taking into consideration the mean daily intake of radiocesium, and fitting the measured data to a linear two-compartment model, the following time-dependent concentration in milk after a single intake was derived (t in days):

$$M(t) = 0.00049 (0.98e^{-0.22t} + 0.012e^{-0.63t}) \text{ (Bq/l per Bq ingested)}$$

The resulting equilibrium transfer factors for meat and various organs are shown in Table 74. Two different groups can be distinguished: skeletal muscle with a mean value of about 0.006 d/kg and soft tissues with a mean value of about 0.003 d/kg.

2.5.4 The connection between radioactive contamination and the age of the animal, as well as the biological half-life

The amount of radioactive isotopes accumulating in the animal body and contaminating the tissues depends, as well as on the feeding conditions and the species, on the age of the animal. The biological half-life, indicating the rate of exchange of the substances ingested by the animal, is much shorter in young animals because of their faster metabolic processes. That is, the young growing animal reacts much faster to the changes in the contamination level of the ambient environment (here through the fodder). The radioactivity of the somatic tissues follows with a shorter time-lag the change of the radioactivity in the environment. However, the internal radiation load affecting the organism, i.e., the actual change of the activity of the radioactive isotopes incorporated into the organism, depends not on the biological half-life, but on the so-called effective half-life ($T_{1/2 \text{ eff}}$), which can be calculated as a function of the biological ($T_{1/2 \text{ biol}}$) and the physical ($T_{1/2 \text{ phys}}$) half-lives as follows:

$$(T_{1/2 \text{ eff}}) = \frac{T_{1/2 \text{ phys}} \cdot T_{1/2 \text{ biol}}}{T_{1/2 \text{ phys}} + T_{1/2 \text{ biol}}}$$

It can be seen that if $(T_{1/2 \text{ phys}}) \gg (T_{1/2 \text{ biol}})$, then $(T_{1/2 \text{ eff}}) \approx (T_{1/2 \text{ biol}})$. For ^{90}Sr and ^{137}Cs the situation is virtually the same, as the physical half-life (28 or rather 30 years) is very long, thus the effective half-life is determined by the biological half-life. Of course, in the case of isotopes with shorter half-lives (e.g. ^{89}Sr , ^{131}I) the decrease of radioactivity due to physical decay is not negligible. If the biological half-life is relatively long, it can even be the dominant factor.

Considering that in grown-up animals the total amount of the examined mineral elements (e.g. K, Ca, Sr, Cs) practically does not change, i.e., the amount taken up daily from fodder and drinking water and the amount excreted with the faeces and urine are the same, the biological half-life can be calculated (Szabó 1981, 1982).

Let a be the total amount of the chemical element in question in the animal, and b the mean amount of daily requirement. In the case of the essential elements K and Ca, b means the actual daily requirement, and in the case of Sr and Cs it means the actual amount of daily absorption (not incorporation), simply because we cannot speak of requirements for non-essential elements. Since

$$\frac{da}{dt} = b - \lambda_b a \quad \text{and} \quad \frac{da}{dt} = 0,$$

where:

t = time (day);

λ_b = constant of biological decay (day^{-1});

the biological half-life is

$$T_{1/2 \text{ biol}} = \frac{a}{b} \ln 2,$$

as it is well known that

$$\lambda = \frac{\ln 2}{T_{1/2}}$$

It can be seen that the biological half-life indicating the rate of exchange of the chemical element in question within the organism does not depend on the absolute amount of a and b , only on their proportion.

On the basis of the data in the literature, as well of the results of our studies of the body composition and daily requirements, Table 75 summarizes the mean Ca and K content as well as the daily mean Ca and K requirement of the 500 kg cattle, 100 kg pig, 60 kg sheep, 3 kg rabbit and 1 kg poultry. Table 76 gives the data of the mean K and Ca content of the meat of these animal species.

Table 75 Whole body Ca and K content, as well as daily Ca and K requirement of some animal species, regarded as average

Animal species	Content in body (g)		Daily requirement (g)	
	Ca	K	Ca	K
Cattle (500 kg)	7500	1500	38	35
Pig (100 kg)	800	350	16	12
Sheep (60 kg)	720	180	6	8
Rabbit (3 kg)	40	6	1.0	1.0
Poultry (1 kg)	12	3	0.6	0.3

Table 76 Average K and Ca content in meat of some animal species

Animal species	K	Ca
	mg/100	raw meat
Pig	440	13
Cattle	380	10
Poultry	400	5
Rabbit	210	19
Sheep	350	12

Using the data in Table 75, the biological half-lives calculated by the formula

$$T_{1/2 \text{ biol}} = \frac{a}{b} \ln 2,$$

based on the Ca metabolism for the Ca-Sr system, and on the K metabolism for the K-Cs system (rounded to days) are listed in Table 77. During the calculations we

assumed that both Sr and Ca, and the element pair Cs and K have the same biological half-lives. However, this is only an approximation, as the behaviour of Cs and K is not so closely related as that of Sr and Ca.

Of course the requirements and the body composition of an animal depend on the species, age, sex, body mass and type of fodder used; thus the biological half-life in different individuals may be rather different. For example, there is a big difference

Table 77 Typical biological half-lives of the Sr-Ca and the Cs-K systems

Animal species	Biological half-life (days)	
	Sr-Ca system	Cs-K system
Cattle	137	30
Pig	35	20
Sheep	83	16
Rabbit	30	4
Poultry	14	7

between the requirement for mineral substances in a lactating sow and a porker, or a breeder sire bull and a dairy cow. It is well known that e.g. sheep can be classified into different types on the basis of the different K content of their blood. Incidentally, the K content of the blood depends on that of the erythrocytes, which in turn is determined genetically. The sum of the K and Na content of the erythrocytes is fairly constant; their proportion can, however, be very different.

Comparing the biological half-lives calculated for the K-Cs and the Ca-Sr systems, we can state that the exchange rate of the latter is substantially lower. That means that K and Cs and thus even ^{137}Cs leave the body much faster than Ca, Sr and ^{90}Sr . As a consequence, radioactive contamination of the animal somatic tissues with ^{90}Sr follows with a substantially longer time-lag the changes of the radioactive contamination in the fodder than in the case of ^{137}Cs contamination.

It must be mentioned that, body mass plays a fundamental role in the length of the biological half-life, as it is clearly seen in Table 77. The reason is the difference in the values of the specific requirements. There is a difference of one order of magnitude between the biological half-lives for cattle with a body mass of 500 kg, for hare with a body mass of 3 kg and for poultry with a body mass of 1 kg. The metabolic rate of the small animals with smaller body mass and high specific body surface is much more rapid, their rates of assimilation and exchange being also higher.

The biological half-life indicating the speed of exchange of Ca (and Sr) content in cattle was calculated using the formula

$$T_{1/2 \text{ biol}} = \frac{7500}{38} \ln 2,$$

and found to be 136.8 days.

The exponential equation describing the decrease of the original Ca content is as follows:

$$y = y_0 e^{-0.00507t}$$

where:

y_0 = original Ca content at time t_0 (here 7500 g);

t = time (in days).

The following formula serves to calculate the rate of exchange of the K (and Cs) content:

$$T_{1/2 \text{ biol}} = \frac{1500}{35} \ln 2 = 29.7 \text{ days}.$$

The decrease in the original K content of the body of cattle is described by the following exponential formula:

$$y = y_0 e^{-0.0233t}$$

where:

y_0 = original K content at time t_0 (here 1500 g);

t = time (in days).

Note that the exponential equation is only an approximation of the changes in the mineral components within the organism because of the great complexity of metabolism.

The amount of K and Ca ingested by an animal is usually greater than required. The amount needed by the organism is absorbed in the gastro-intestinal system, and is excreted primarily with the urine. The excess is passed out with the faeces, and since it is not assimilated, it does not participate in the metabolic process, and it has virtually no influence on the biological half-life indicating the rate of exchange of the mineral substances in the animal's body. Thus knowing the amount of K, or rather Ca excreted daily with the urine (and of course the total amount of K and Ca in the organism), the biological half-life can be calculated. The same can also be said for Sr or Cs, i.e., the measurement of the amount of Cs, or rather Sr, excreted with the urine is important, because it reflects the amount of these elements taken up by the organism. In order to better understand the data in Table 77, we should mention that the biological half-life of cesium in different animals usually has a value between 2 and 30 days. According to Seletskaya et al. (1976), the mean value of the biological half-life of cesium in rats is 23 days.

The method of calculation demonstrated earlier is suitable not only for determining the biological half-lives of K and Ca, but also for the other substances (elements) building up an organism. However, since calcium has a common transport system with strontium, and potassium with cesium, the biological half-lives calculated for Ca and K are also applicable for Sr and Cs, respectively. Of course, the calculation can also be made knowing the inactive Sr and Cs content of the organism, as well as the daily Sr and Cs uptake. According to our measurements, the biological half-lives calculated on the basis of the Ca and K metabolism are in good accordance with the values determined from Sr and Cs metabolism.

In addition to the measurements of the biological half-lives of the ^{90}Sr and ^{137}Cs isotopes of artificial origin, there are publications in the literature mostly on the determination of the biological half-life of ^{210}Pb . Lead, chiefly ^{210}Pb , accumulates in the bones. According to the investigations of Jaworowski (1965), Lloyd et al. (1975)

and Yermolayeva-Makovskaya and Litver (1978), the biological half-lives in rats and dogs are long, usually several years.

The biological half-life of the Sr-Ca system in cattle, calculated with the given method, is about 137 days, that of the Cs-K system is about 30 days. Thus the radiostrontium concentration is less variable in the somatic tissues and because of its longer biological half-life it follows with a relatively longer time-lag possible changes in contamination level in the fodder. Thus, if an animal consumes fodder of a given contamination level for a relatively long time (several months) and then eats for a shorter time (some weeks) fodder with significantly higher contamination, the ^{90}Sr activity of the somatic tissues increases only slightly. Similarly, the radioactive contamination level characteristic earlier for an animal will decrease only very slowly if the activity of its food is lowered. At the same time, the change in the ^{137}Cs content is faster, as its biological half-life is shorter. Consequently the radiocesium contamination of the animal somatic tissues follows the changes in the contamination level of the fodder with a shorter time-lag than in the case of radiostrontium contamination.

Of course, the relatively fast changes characteristic of the Cs-K system, or the relatively slow changes characteristic for the Sr-Ca system are only valid in one animal species, as the biological half-lives are quite different in the various animal species. For example, in the rabbit the biological half-life of the radioactive contamination of the somatic tissues is significantly shorter than in cattle. The consumption of fodder of different levels of contamination can cause substantial changes in the Sr-Ca system within weeks and in the Cs-K system within days. However, it is a fact that the biological half-life of the Sr-Ca system is longer by one order of magnitude than that of the K-Cs system of any animal.

The contamination level measured in young, growing animals changes much faster than in mature older animals. It can be explained by their smaller body mass, shorter biological half-life, greater requirements for mineral substances, more intensive metabolism and higher assimilation rate. There is a substantial difference between the growing and older animal. In the young, growing animal there is no equilibrium between the mineral substances ingested and excreted; because of growth the rate of assimilation is higher than the amount of mineral substances leaving the organism. Thus in young animals the level of contamination changes faster than that expected from the biological half-life calculated using the formula

$$T_{1/2 \text{ biol}} = \frac{a}{b} \ln 2,$$

if the radioactivity of the ambient environment changes (mediated by the fodder or drinking water). The formula is suitable for calculating the biological half-life only if the rate of daily assimilation (anabolism) and dissimulation (catabolism) is nearly equal.

Radioactive contamination of the tissues of young and old animals can be significantly different. It is apparent from Table 78 that the radiostrontium contamination of the bones of calf (maximum 3 months of age) is lower than that of the growing heifers (between the age of 3 and 6 months). The difference can be explained only partly by the

fact that the older the animal, the higher the mineral (therefore also the ^{90}Sr) content of the bones, because the activity of the metal ion-fraction due to the Ca content is also lower in the calf bones. The fact that the Sr:Ca, or rather the ^{90}Sr :Ca, ratio in the calf bones is smaller than in the bones of the older animals is connected with the fact that the Sr content, or rather the ^{90}Sr contamination of the milk, because of the

Table 78 Average metal ion-fraction activity of bones (metacarpus) in calf and growing heifers in Hungary

Year	Metal ion-fraction activity			
	mBq/g bone		mBq/g Ca	
	calf	growing heifers	calf	growing heifers
1971	81	174	770	1240
1972	93	133	858	940
1973	59	130	560	930

good discriminating ability of the cow, being one order of magnitude lower than in the fodder. Thus the ^{90}Sr activity of the younger animals fed on milk of low contamination level, is smaller than that of the older animals consuming plants as fodder. Actually there is a doubled discrimination effect. In a similar way the Pb and ^{210}Pb contents of the somatic tissues of the young animals are lower.

In contrast to Sr, there is no such discrimination effect for Cs; the animal organisms do not alter the relative concentration of cesium in relation to potassium, or they enrich it. Therefore we cannot say that for radiocesium there is any difference between the radioactive contamination of the somatic tissues of younger and older animals. Moreover, if the value of D_{Cs} is greater than 1, the ^{137}Cs contamination of the somatic tissues of young animals fed on milk or recently weaned can be higher than in older animals feeding on plants. According to our measurements, as we have already seen, the ^{137}Cs contamination of the milk is about the same as that of the fodder consumed by the cows, and thus the activities of the somatic tissues of the young animal fed on milk is expected to be identical to the radioactive contamination of the older, grown-up individuals.

Generally it can be stated that the older the animal, the greater the amount of radioactive isotopes, accumulated in the somatic tissues, even at constant levels of environmental contamination. For example, according to the investigations of Kulikov and Kulikova (1977), the ^{137}Cs contamination of pike increases linearly with age.

2.5.5 Factors influencing radioactive contamination of the fodder-animal biological chain

Knowing the level of radioactive pollution of different plants, as well as that of surface waters and drinking water, there is no doubt that radioactive isotopes reach the animal organisms mainly from fodder. Thus the contamination level of the fodder

is the decisive factor which, depending on the discrimination factors characteristic of each animal species, determines the amount of radioactive substances accumulated in the somatic tissues of animals, and in this way in our foods of animal origin. The factors determining the contamination level of fodder or rather the plants are as follows:

- geologic, geographic, orographic and edaphic factors;
- climatic and meteorological parameters;
- the species and vegetation period of the cultivated plants;
- the applied methods of cultivation, melioration.

The proportion of the radioactivity arriving from fodder plants in the different animal somatic tissues is determined by the animal species, the circumstances of animal husbandry and feeding, and the age of the animal. The radioactivity of the animal somatic tissues is determined by the following factors:

- the ambient environment (fodder, drinking water, the aquatic environment in the case of aquatic animals);
- the discriminating ability of the organism, which in turn is determined by the species, variety, body mass, sex, etc.;
- the biological half-life of the radionuclides, which is determined chiefly by the age of the animals;
- the circumstances of husbandry.

As we have already mentioned, the ^{90}Sr and ^{137}Cs contamination of the vegetation and of the plants used to feed the animals is usually of the same order of magnitude. (Of course in 1986, because of the Chernobyl disaster, the situation was

Table 79 ^{137}Cs : ^{90}Sr proportion in milk, beef and pork

Year	^{137}Cs : ^{90}Sr proportion		
	Milk	Beef	Pork
1967	3.8	16.0	17.0
1968	3.2	11.3	10.2
1969	2.5	7.6	7.8
1970	2.7	8.9	6.8
1971	3.1	9.2	6.9
1972	2.6	8.0	7.7
1973	2.7	5.4	5.3
Average	3.0	9.5	8.8

different in many European countries: the level of ^{137}Cs contamination was many times higher than that of the ^{90}Sr contamination.) As a matter of fact, animal organisms discriminate well against ^{90}Sr , but they virtually do not discriminate against Cs (or if so, in the opposite direction). Thus in the animal somatic tissues, but not in the bones, the level of ^{137}Cs contamination is predominant compared to the

^{90}Sr activity. This is proved also by the data in Table 79, which were calculated from the data in Tables 62 and 63. They show the mean $^{137}\text{Cs} : ^{90}\text{Sr}$ ratio for the period 1967 to 1973 in milk, beef and pork, based on measurements carried out in Soviet Union (Marey 1980). It can be seen that in milk the level of ^{137}Cs contamination was about three times higher than that of the ^{90}Sr contamination, and in meat the difference was almost exactly one order of magnitude. The fact that the $^{137}\text{Cs} : ^{90}\text{Sr}$ ratio in meat is higher than in milk can be explained by the substantially higher K : Ca ratio in the former (see Table 76). So it is evident that in meat first of all the accumulation of ^{137}Cs , which is chemically similar to potassium, can be expected. For comparison, in the GDR from 1967, to 1970 the average $^{137}\text{Cs} : ^{90}\text{Sr}$ ratio in milk was 4.9, 3.2, 4.8 and 4.8, respectively (Schulze-Wollgast 1972). In Bulgaria, the ^{137}Cs contamination of pork and beef in 1967 was 4.6 ± 0.8 and 4.0 ± 1.2 Bq/kg of meat, respectively (Marinov et al. 1969).

The radioactive contamination of animal somatic tissues is also determined by husbandry. Thus, for example, the radionuclides of relatively short half-lives, which by fall-out reach the surface of the plants and cause practically only surface contamination, can enter the organisms of animals kept in the open-air on pastures and thus they may contaminate foodstuffs such as milk, egg and meat. Such isotopes are e.g. ^{131}I or ^{140}Ba . In animals kept indoors, fed with prepared fodder, nourishment silage or hay harvested much earlier, the level of such contamination is very low as a relatively very small amount of radioactive isotopes may enter their body with the drinking water or through inhalation.

In practice, the radioactive contamination of an animal's body is determined by its feeding habits, whether it is herbivorous or carnivorous. In carnivorous animals there is a double discrimination for radiostrontium: the level of contamination of the meat eaten is already much lower than that of plants. This difference can be demonstrated very well by analysing the ^{90}Sr contamination in predatory and herbivorous fish. This double discrimination also explains the lower Sr content and radiostrontium contamination of the somatic tissues of young animals fed on milk or recently weaned than in the older animals fed on fodder. Namely, the Sr : Ca or rather the $^{90}\text{Sr} : \text{Ca}$ ratio in milk, simply because of the effective discriminating ability against strontium, is substantially lower than in the fodder consumed by the animal.

We cannot speak of a similar discriminating ability in the case of Cs, because in animal somatic tissues the Cs : K, or rather the $^{137}\text{Cs} : \text{K}$, ratio can be higher than the value measured in the fodder plants (discrimination is referred to the K because of the similar chemical properties of Cs and K).

Since accumulation of ^{137}Cs can be expected mainly in the animal somatic tissues, scientists have been looking for methods for a long time (isotope dilution, increasing the K content of fodders, chelate formers, application of diuretics, hormone treatment), which may contribute to the speedy removal of the incorporated ^{137}Cs (Catsch 1964). For instance in Hungary, in the Animal Breeding and Feeding Research Centre (Herceghalom), based on investigations of Cs metabolism in rats and pigs, it was found out that using the iron-cyanide complex ($\text{KFe}(\text{Fe}(\text{CN})_6)_3$) as prophylactic material, the amount of Cs passed out with the urine was about 1–2% of the amount eliminated with the faecal matter. Without prophylactic material most

of the Cs was eliminated with urine. Thus, using the prophylactic substance, the absorption of cesium and of course ^{137}Cs through the intestinal membranes can be blocked. Incidentally, the prophylactic material mixed with the fodder, must fix cesium selectively and in quantity, and in addition it must not be toxic and it cannot have an adverse influence on the digestion.

Using different means of decorporation (e.g. Na-alginate) the absorption rate of strontium through the intestinal membranes can also be decreased. This effect can be explained either by the complex-forming features of the decorporating compounds, or by the blocking of the proteins participating in the strontium transport (Máté et al. 1973).

2.5.6 Possibilities for decreasing the radioactive contamination of food of animal origin.

The effect of certain processes of food production on the contamination level

I hope, it is fairly clear from what has been said so far, that under the present contamination level, similarly to foodstuffs and agricultural products of plant origin, there is no need to apply any special method of decreasing the radioactive pollution of animal tissues and foodstuffs of animal origin, because the level of radioactive contamination compared to the natural background radiation is usually low. However, it is worth of studying this issue as in the case of a higher-level radioactive contamination (nuclear explosion test, reactor disaster, etc.) it may be necessary to apply various decontamination methods. Several measures, for example, had to be taken in Hungary and in other Central European countries in 1986, in the year of the Chernobyl disaster.

As the radioactivity of animal somatic tissues is determined mostly by the radioactive contamination level of fodder and drinking water, an obvious measure is feeding the animals fodder of relatively low level of contamination. Thus the radioactive isotopes do not attain a dangerous concentration in the animals. We have seen that animal organisms discriminate well against strontium; thus the radiostrontium pollution of the different animal tissues does not reach the critical value even if fodder of relatively high level of contamination is fed. If there is ^{137}Cs pollution, using prophylactics added to fodder, e.g. iron-cyanides, the absorption of cesium from the digestive tract can be blocked and ^{137}Cs will not contaminate the animal somatic tissues (Seletskaya et al. 1976).

Just as in the case of foodstuffs of vegetable origin, in foodstuffs of animal origin there are different methods for decreasing the level of radioactive pollution. Thus the amount of active substances coming into human organisms through the nutritional chain, can be decreased. In the case of meat or meat products, the radioactive dust pollution resulting from fall-out can be eliminated by proper protective covering (e.g. polymer compounds, paper saturated with paraffin, cardboard, aluminium foil). If the pollution affects the surface of the meat, the upper layer should be removed.

Of course, when selecting the proper decontamination methods, the half-life of the polluting isotopes also should be considered. If the contamination is caused by radionuclides of relatively short half-life (e.g., ^{131}I , ^{140}Ba), there is no need for any special decontamination method; only safe storage should be ensured. In this case decontamination simply means a problem of storage and conservation. Such a solution would be not sufficient if radioactive isotopes of long half-lives cause the radioactive contamination. However, there are various other methods: for example, by passing milk through ion-exchanging resin, the amount of radiostrontium can be decreased to 2–4% of its original quantity (Braude, Glascock 1969).

We have already mentioned that through the different technological stages of food processing and preparation the level of radioactive contamination changes. For example, during cooking, a certain part of the mineral substances in meat is washed out, and together with the different mineral salts the concentration of the radioactive isotopes can also decrease significantly. During cooking, the ^{90}Sr contamination of beef decreases by 40–50% (Belova et al. 1967). During milk-processing, the decrease of contamination is very significant, as only a few per cent of the whole ^{90}Sr content is present in cottage cheese or cheese (Haraidze, Kaladiya 1974). It can be explained by the fact that most of the mineral substances in milk, including the radioactive, remain in the whey. In a similar way, only a very small part of ^{137}Cs in the milk remains in the sour cream, on the average only 4%.

The change in contamination of milk during processing can be seen, for example, in the study of Schulze-Wollgast (1972), according to which in 1968 in the GDR the mean ^{90}Sr and ^{137}Cs contamination of curd was 0.30 and 0.52 Bq/kg, respectively, whereas the mean ^{90}Sr and ^{137}Cs contamination of cheese was 1.89 and 0.52 Bq/kg, respectively. In the same year the mean ^{90}Sr and ^{137}Cs contamination of milk was 0.24 and 0.78 Bq/l, respectively. Of course, the decrease of contamination is significant

Table 80 Results of radiometrical investigations of milk products

Milk product	Activity, Bq/100 g			^{90}Sr activity mBq/g Ca
	total- β	^{40}K	^{90}Sr	
Port Salut	3.57 ± 0.15	2.86 ± 0.09	0.19 ± 0.06	223 ± 74
Edam cheese	3.09 ± 0.09	2.50 ± 0.08	0.21 ± 0.01	247 ± 17
Dry cheese	2.86 ± 0.14	2.17 ± 0.03	0.20 ± 0.03	265 ± 52
Low-fat cottage cheese	4.52 ± 0.48	4.04 ± 0.48	0.06 ± 0.02	283 ± 95

only if expressed in dry matter (or protein). For instance, if the ^{90}Sr contamination is calculated for the Ca content (see Table 80), the level of contamination does not change at all. On the basis of the data in Table 80, the results of a radiometrical survey carried out in Hungary in 1980 in the milk industry, it can be said that the ^{90}Sr contamination level of milk products, expressed as Ca content, is more or less the same, and does not differ from the contamination level of milk. In 1980 the ^{90}Sr activity (Table 60) had a value of 173 ± 135 mBq/g Ca.

2.5.7 Summary

In this subchapter the process of the buildup of natural and artificial radioactivity in the animal organism has been discussed, demonstrating its dependence on the species and the age of the animal in question. The factors influencing the radioactive contamination of the fodder–animal biological chain, the possibilities of decontaminating the foodstuffs of animal origin, and the influence of the different stages of the food-processing on the level of radioactive contamination have been analysed. The effect of the Chernobyl disaster, which increased the level of contamination, was also discussed. The most important points are the following:

1. The radioactivity of the somatic tissues of the animal is determined first of all by the level of pollution in the fodder. The amount of radionuclides entering the animal organism with the drinking water and through inhalation is much smaller. Most of the radionuclides reaching the animal organisms are of natural origin (^{40}K); the proportion of isotopes indicating contamination of artificial nature is, under the present contamination situation, usually 1–2 orders of magnitude smaller. During the period of maximum contamination due to the nuclear explosion tests, the radioactive contamination was higher by one order of magnitude than today. As is well known, in Hungary and in other European countries, the radioactivity of animal tissues reached its highest level in 1963 and 1964. However, the level of the radioactive contamination in 1986, due to the Chernobyl disaster, was in most European countries higher than the maximum values in the 1960s.

2. The relative concentration of the radioactive substances ingested by animals from fodder or other plant material changes. The degree of change is indicated by the so-called discrimination factors. Knowing these, the amount of the active substances passing on from the fodder into the somatic tissues can be estimated. For strontium, or rather radiostrontium, the fodder/animal discrimination factor can be calculated as follows:

$$D_{\text{Sr}} = \frac{{}^{90}\text{Sr} : \text{Ca in the tissues}}{{}^{90}\text{Sr} : \text{Ca in the fodder}} = \frac{\text{Sr} : \text{Ca in the tissues}}{\text{Sr} : \text{Ca in the fodder}}$$

According to our investigations, the value of the discrimination factor indicating the discriminating ability of cattle, on the basis of simultaneous radiometric measurements of milk and fodder samples is as follows:

$$D_{\text{Sr}} = 0.103 \pm 0.019$$

This value indicates that the radiostrontium contamination of animal organisms is smaller by one order of magnitude than the contamination of the food consumed by the animals. According to our measurements the radioactive contamination of the somatic tissues of the different animals is by no means the same. It mainly depends on the discriminating ability specific to each species, variety and different food habits. Based on bone measurements it can be stated that ^{90}Sr contamination increases in the different species in the following order: pig, poultry, cattle, sheep, fish.

3. The other important isotope causing lasting radioactive contamination in the

biosphere is the ^{137}Cs isotope. In this case the discrimination factor is the following quotient:

$$D_{\text{Cs}} = \frac{{}^{137}\text{Cs} : \text{K in the tissues}}{{}^{137}\text{Cs} : \text{K in the fodder}} = \frac{\text{Cs} : \text{K in the tissues}}{\text{Cs} : \text{K in the fodder}}.$$

According to our measurements, the value of the discrimination factor of the cattle is as follows:

$$D_{\text{Cs}} = 0.90 \pm 0.40$$

This value demonstrates that actually there is no discriminating effect, i.e. the Cs : K ratio in the somatic tissues and in the fodder is about the same. According to the data in the literature, the value of D_{Cs} can be even greater than 1.0, i.e., the animals are capable of accumulating cesium compared to potassium. Physiologically it means that the Cs absorption from the digestive tracts is more efficient than that of potassium.

As a consequence of the rather different discriminating abilities of the different animals in relation to Sr and Cs, in the animal tissues except the bones, ^{137}Cs activity dominates. This difference is especially apparent for ^{90}Sr and ^{137}Cs activity in the somatic tissues of carnivorous animals, where in fact, there is a double discrimination: first at the plant-herbivore step, then at the herbivore-carnivore step.

4. The contamination level of animal somatic tissues, or rather the speed of change of radioactive contamination, is determined by the biological half-life, which in turn is influenced by the species, variety, sex, age, body mass and type of utilization of the animal in question. The biological half-life of Sr, or rather that of ^{90}Sr , is a substantially longer period than the value measured for Cs, or rather ^{137}Cs . Thus radiostrontium pollution follows the change in contamination level in the fodder with a longer time-lag than does radiocesium contamination. The biological half-life indicating the rate of exchange of the substances within the organisms can be calculated as follows:

$$T_{1/2 \text{ biol}} = \frac{a}{b} \ln 2,$$

where:

a = total amount of the chemical element in question within the organism;

b = amount of daily requirement of the essential elements (e.g. K) in question, or rather the amount of daily assimilation of non-essential elements (e.g. Cs).

In young animals, because of their more intensive physiological processes, the biological half-life is substantially shorter than in grown-up or old animals.

5. In 1986, during the weeks after the Chernobyl disaster, in addition to ^{137}Cs , very high levels of ^{131}I and ^{134}Cs contamination were measured in animal tissues in numerous European countries. Fortunately, the activity of ^{131}Cs , because of its short half-life, decreased rapidly.

6. In spite of the fact that, fortunately, the present level of radioactivity does not necessitate the application of any special decontamination measure, there are special methods for foodstuffs of animal origin (e.g. ion-exchange of milk), which can significantly decrease the amount of radioactive substances endangering humans consuming food of animal origin. The different technological stages of food preparation can also alter the level of radioactive contamination. For instance, not more than

a few per cent of the original contamination of milk remain in cottage cheese or in cheese, because the majority of the mineral salts in milk, including most of the radioactive strontium and cesium, are retained in the whey.

7. The Radiological Control Network of the Hungarian Ministry of Agriculture and Food carried out measurements of radioactivity of more than 40 000 samples in 1986, including the measurements made by the special laboratories of the different food industries. The purpose of some of these measurements was to monitor the radioactive contamination in a region. However, most of the measurements were needed to control the quality of food for export. Based on these measurements, the following conclusions could be drawn:

- the degree of radioactive contamination caused by the Chernobyl disaster was extremely different in the various regions of Hungary;

- special attention was paid to the crops harvested in May, to see whether they could be used later, and certain additional measurements were necessary in order to assure a milk supply of acceptable level of radioactivity for the population;

- around the end of the year, the receptive capacity of the environment eliminated the radioactive contamination to a significant degree. The results of measurements of samples taken in autumn demonstrated that the radioactivity of the environment "returned" to the order of magnitude characteristic of the period before the disaster;

- as a result of the quality control measurements of the food for export, only 3.2% of the measured food was retained because its contamination levels exceeded the limits accepted by international standards.

2.6 RADIATION LOAD ON HUMANS AND RADIOACTIVITY OF HUMAN TISSUES

We are approaching not the longest but without doubt most important subchapter of this book. Here the issues of the radiation load on the human organism will be discussed. This is the most important part, as after all man takes his place on the top of the pyramid, man is the point where the vectors of all processes going on in the various elements of the biosphere summarize in a final compartment. The transfer processes characteristic of the parts of the biological chain, the migration of radionuclides, and the factors influencing the contamination level are not interesting in themselves; their study is not justified unless in a framework of a dynamic and complex relationship where the interactions finally lead to man. The purpose of all these researches is to gain a broad view into the whole spectrum of the biological chain, i.e., to obtain answer to the final question: what influences do the changes characteristic for each element of the air–water–soil–plant–animal chain have on man, how do they influence the radioactive isotope content of human somatic tissues.

Man cannot make himself independent of his environment, i.e., the changes of the contamination in the ambient environment (food, drinking water, air) also influence the radioactivity of human somatic tissues. The radiation load on the human organism (both external and internal) depends to a high degree on factors such as feeding

habits, way of life (working place, profession, conditions of domicile, etc.) and medical radiation load in relation to the status of health. The last questions in fact do not belong to the field of radioecology, since they are actually issues of radiobiology, radiation protection and dosimetry, which are discussed here only in a tangential manner, just to give an impression of the proportion of the radiation load coming from different sources, as well as the changes caused by them. From the point of view of radioecology, we are interested in the way radionuclides enter the human organism, and in the factors influencing the accumulation and concentration of active isotopes in the human somatic tissues.

In this subchapter the sources of the radiation load on the human organism will be discussed, data will be presented on the radioactivity of the human body tissues, and the factors influencing the radioactive pollution will be analysed. Last but not least—outlining the possibilities for decreasing the radiation load—the effects of radiation on the human organism will be briefly related, describing the essence or rather the dangers, of the somatic and genetic radiation damages.

2.6.1 Radiation load on the human organism

The human organism receives radiation load basically from two sources. The first is the incorporated radioactive isotopes, which means that the radiation of the radionuclides of natural and artificial origin come through the food chain (partly through inhalation) and are accumulated in the human organism. This is the internal radiation load. However, the organism receives radiation from external sources too.

Before relating the various components of the radiation load, we have to look at the up-to-date concept of radiation load dosage. The issues of radiation load on the population are discussed in detail, for example, by the UNSCEAR reports.

To characterize the biological radiation effect, the amount of energy absorbed in the tissues must be known. The absorbed dosage is the energy absorbed by the mass unit of their radiated material. Its unit is the gray (Gy).

However, knowledge of the absorbed dosage (D) in human tissues or organs is not always sufficient to precisely forecast the health-damaging effect of the radiation. This is the reason why in radiation protection practice the concept of the dose equivalent (H) is used, which is the absorbed dosage modified by a quality factor:

$$H = Q \cdot D$$

where:

Q = a quality factor depending on the microscopic distribution of the absorption of the radiation energy in the tissues, its value in case of X- and gamma rays is 1.

The unit of the dose equivalent is the sievert (Sv).

When the environment or rather of the human organism is contaminated with radioactive substances and the radiation load is lasting, the dosage accumulates only within a shorter or longer period of time. For these cases the concepts of dose commitment (D_c) and dose equivalent commitment (H_c) have been introduced. D_c means the time integral of the absorbed dosage intensity, H_c means the time inte-

gral of the dose equivalent intensity:

$$D_c = \int_{t_0}^{t_{\infty}} D(t) dt, \quad \text{or rather} \quad H_c = \int_{t_0}^{t_{\infty}} H(t) dt.$$

A recently introduced concept is the dose equivalent commitment produced in an organ or tissue as a consequence of a single radionuclide uptake, H_{50} (ICRP-26 1977). This is equal to the dose equivalent which accumulates within 50 years after the uptake of the radionuclide.

$$H_c = \int_t^{t+50 \text{ years}} H(t) dt.$$

The degree of the radiation load caused by radionuclides coming into the human organism can be determined knowing the amount (activity) of the radionuclide, its physical and chemical parameters, and other physiological characteristics. Instead of the rather complex calculations, tables compiled by competent international organizations (e.g. ICRP) are used in practice, making the lengthy calculations superfluous. The calculation is carried out on the basis of the following relationship:

$$D_{i,j} = U_i \cdot R_{i,j}$$

where:

$D_{i,j}$ = radiation load (Sv) caused by radionuclide i in organ or tissue j ;

U_i = the amount of radionuclide i coming into the human organism (Bq);

$R_{i,j}$ = the dosage factor of radionuclide i referred to the organ j , i.e., the radiation load produced by a unit of activity of the radionuclide i in the organ j .

Note that the dosage factor for ^{137}Cs is $1.4 \cdot 10^{-8} \text{ Sv} \cdot \text{Bq}^{-1}$ in the gonads and $1.3 \cdot 10^{-8} \text{ Sv} \cdot \text{Bq}^{-1}$ in bone marrow (medulla). For ^{131}I the dosage factor for the thyroid gland is $4.8 \cdot 10^{-7} \text{ Sv} \cdot \text{Bq}^{-1}$.

The biggest component in the external natural dosage is the gamma radiation of the soil, or rather, since civilized man spends most of his life in buildings, the gamma radiation of buildings.

The degree of dosage received depends also on the structure and material of the houses, partly because the buildings shield people from certain external radiation load (of terrestrial and, to a lesser extent, of cosmic origin), and partly because the radioactivity of the air inside the building is determined mostly by the ^{226}Ra and ^{232}Th content of the building material (Tóth, Fehér 1976; Fehér et al. 1975). In closed buildings with no ventilation the concentration of the α radiating ^{222}Rn and ^{220}Rn can be many times higher than in well-ventilated inner spaces; thus they can cause a substantial radiation load.

Another important component of the external radiation load is cosmic radiation. Its intensity is almost the same all over the surface of the Earth. It does not depend on geological circumstances, but, its degree correlates closely with the altitude, with which it increases exponentially. For instance, people living in the mountains of Caucasus and Pamir receive a dosage of 1.0–3.5 mSv (Boltneva et al. 1977). The added dosage sustained by the crews of airplanes is also not negligible: supposing 1000 flying hours at an altitude of 10 km, the added dosage due to cosmic radiation is about 1.9 mSv within one year (Maschek et al. 1981).

The share of the substances of artificial origin in the external radiation load is often higher than the dose from the natural radioactivity of the environment or the dosage due to cosmic radiation. Here first of all the radiation load of medical origin should be considered, which for part of the population means diagnostic and for another part means therapeutic radiation load. According to Gusev (1975) the radiation load of artificial origin on the human organism is on the average 90% of a medicinal character, and this is about 35% of the total radiation load.

In addition to the natural radioactive substances, our environment is also polluted by radioactive isotopes of artificial origin. Fortunately, within the present contamination situation this radiation load is substantially smaller than the dosage received from the natural background radiation, and usually has a value below 0.1 mSv per year. However, it must be mentioned that in 1963, when in consequence of the nuclear weapon tests the maximum of the contamination could be measured, the dosage thus received was, e.g. in the Soviet Union, on the average about 0.4 mSv (Marey 1980). Incidentally, in 1963 the majority of the radiation load due to the external contamination was caused by the presence of the $^{95}\text{Zr} + ^{95}\text{Nb}$ isotopes, and since 1965 the decisive part of the external artificial radiation load has been due to the radiation of ^{137}Cs .

It should be stressed that a growing part of the population works professionally with different open and closed radioactive sources and preparations or with X-ray equipments; thus they may be exposed in their working places (nuclear reactor, accelerator, isotope laboratory, etc.) to an increased dose of radiation. There are rigorous measures for those working in places with radiation danger: compulsory medical controls (monthly-bimonthly), film-dosimetrical evaluation of the dosage received. We do not want to discuss this topic in detail, since it is an issue of radiation protection and accident prevention regulations. We have only to mention that health physics regulations permit 50 mSv/year radiation dosage for people working in places with radiation danger. It is about 30 times higher than the radiation load on the normal population (not including the dosage of medical origin). Incidentally, young people and pregnant women should not work in places with radiation danger.

Among the components of the external radiation loads it is worth mentioning that, because of the X-ray radiation, there is a certain radiation load on people watching television (particularly colour television). However, this component is usually negligible, compared to the other ones, if there is a distance of at least 3 metres between the screen and the person watching the programme.

In the foregoing the external radiation load on the human organisms has been discussed. However, the dosage due to the radioactive isotopes incorporated into the organisms can be also significant.

The highest dosage on the whole body is due to ^{40}K as well as to ^{222}Rn and its derivatives (Andrási, Fehér 1969). The total radiation load due to the other radioactive isotopes of natural origin (^{87}Rb , ^{14}C , etc.) is lower. It must be stressed that we are speaking now about the mean radiation load referred to the whole body, and naturally certain organs and tissues, depending on the localization of the radioactive isotope in question, can receive significantly higher or lower radiation dose. For example, most of ^{226}Ra is accumulated in the bones, and thus the radiation load

on the bones due to ^{226}Ra is much higher, having an average value of yearly 0.5 mSv. The degree of the internal radiation load due to the incorporated radioactive isotopes also depends on the way of life. Thus, for example, the ^{210}Pb content is much higher in the bodies of heavy smokers.

Another part of the internal radiation load is of artificial origin and it is due to the radioactive isotopes contaminating the environment (^3H , ^{90}Sr , ^{137}Cs , etc.) entering into the human organism, and in certain cases to the application of radiopharmaceuticals. The radiopharmaceuticals may be defined as those open radioactive isotopes which are used for diagnostic and therapeutic purposes. There are many radioactive isotopes used for diagnostic purposes (e.g. ^{85}Sr , ^{197}Hg , ^{75}Se , ^{18}F , ^{51}Cr), the most frequently used therapeutic pharmaceuticals being ^{32}P , ^{131}I and ^{198}Au .

Among the present contamination circumstances, the internal radiation load of artificial origin, referred to the whole body, can be assessed as 0.1 mSv per year. However, in the Arctic regions, where the main food of the people is the milk and meat of reindeer, the ^{137}Cs contamination of the somatic tissues can be so high that the biggest part of the radiation load on the organism is due to the activity of the ^{137}Cs incorporated into the somatic tissues, and its value can be significantly higher even than the value of 1 mSv.

Summarizing what we have said earlier and in order to give a picture of the radiation load on the human organism, in Table 81 the proportions of radiation load on the human organism originating from different natural sources are listed (UN-

Table 81 Annual effective dose equivalent from natural sources (UNSCEAR 1988)

Source of irradiation	Annual effective dose equivalent (mSv)		
	External	Internal	Total
Cosmic rays			
Directly ionizing component	0.30		0.30
Neutron component	0.055		0.055
Cosmogenic radionuclides		0.015	0.015
Primordial radionuclides			
Potassium-40	0.15	0.18	0.33
Rubidium-87		0.006	0.006
Uranium-238 series:	0.1		1.34
Uranium-238 to uranium-234		0.005	
Thorium-230		0.007	
Radium-226		0.007	1.24
Radon-222 to polonium-214		1.1	
Lead-210 to polonium-210		0.12	
Thorium-232 series:	0.16		0.34
Thorium-232		0.003	
Radium-228 to radium-224		0.013	0.18
Radon-220 to thallium-208		0.16	
Total	0.8	1.6	2.4

SCEAR 1988). Table 82 shows the values of the effective dose equivalent commitment due to the polluting effect of the nuclear tests (UNSCEAR 1982).

The effective dose equivalent commitment calculated for the human population in the most contaminated areas of Hungary was 1 to 2 mSv as a consequence of the Chernobyl event (Andrási et al. 1986). This value is roughly equivalent to the annual average dose burden due to natural environmental radiation.

The results of calculations for the first-year committed effective dose equivalent in 34 countries are illustrated in Figure 27 (UNSCEAR 1988). The highest values are for Bulgaria, Austria, Greece and Romania, followed by other countries of northern, eastern and southeastern Europe. Countries further to the west in Europe and also countries of Asia, North Africa, North and Central America were less affected, which is in accord with the deposition pattern.

Table 82 Average values of effective dose equivalent commitment resulting from the polluting effect of the nuclear weapon tests carried out before 1980

Radionuclide	Effective dose equivalent commitment (μSv)	% proportion
¹⁴ C	2600	69
¹³⁷ Cs	540	14
⁹⁵ Zr	200	5.3
⁹⁰ Sr	120	3.2
¹⁰⁶ Ru	83	2.2
¹⁴⁴ Ce	54	1.4
³ H	47	1.2
¹³¹ I	33	0.9
²³⁹ Pu	27	0.7
¹⁴⁰ Ba	25	0.7
¹⁰³ Ru	17	0.4
²⁴⁰ Pu	17	0.4
²⁴¹ Pu	9	0.2
⁵⁵ Fe	9	0.2
²⁴¹ Am	4	0.1
⁸⁹ Sr	3	0.08
¹⁴¹ Ce	1	0.03
²³⁸ Pu	1	0.03
¹³⁶ Cs	0.06	0.002
⁵⁴ Mn	0.04	0.001
⁸⁵ Kr	0.005	0.0001
Total (rounded values)	3800	100

Note: In the case of the ¹⁴C isotope the high percentage of effective dose equivalent commitment is in relation to the relatively long half-life. Considering the time until 2000, only 7.7% of the value in the table is the effective dose equivalent commitment.

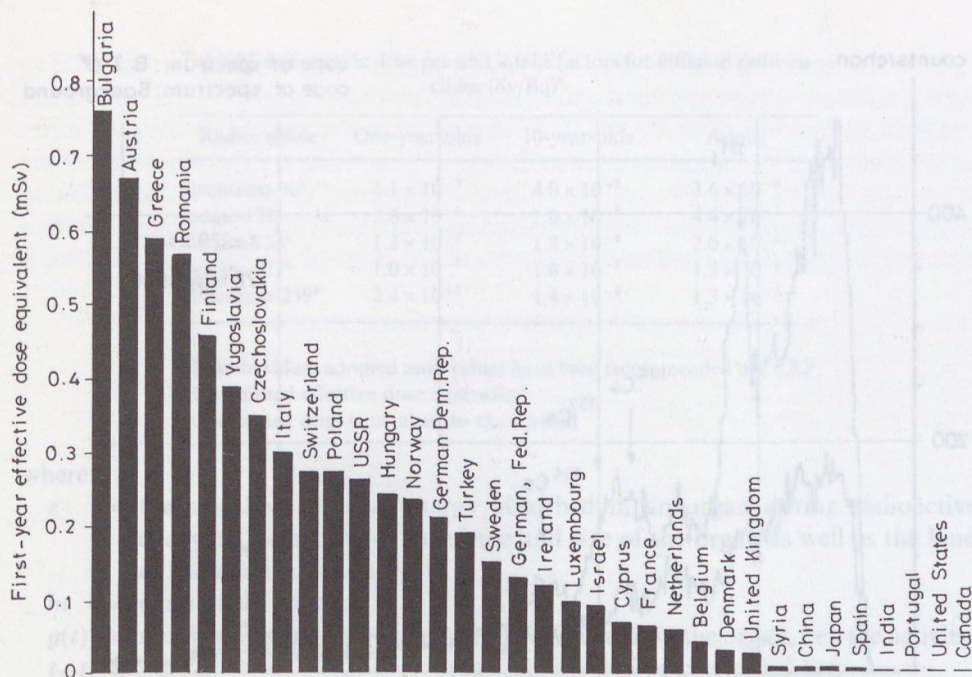


Fig. 27. Country average first-year committed effective dose equivalent from the Chernobyl accident

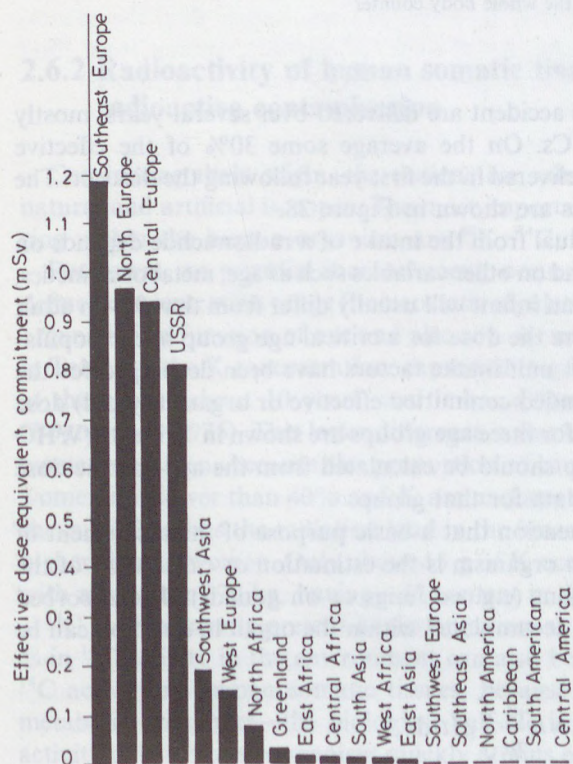


Fig. 28. Regional average effective dose equivalent commitment from the Chernobyl accident

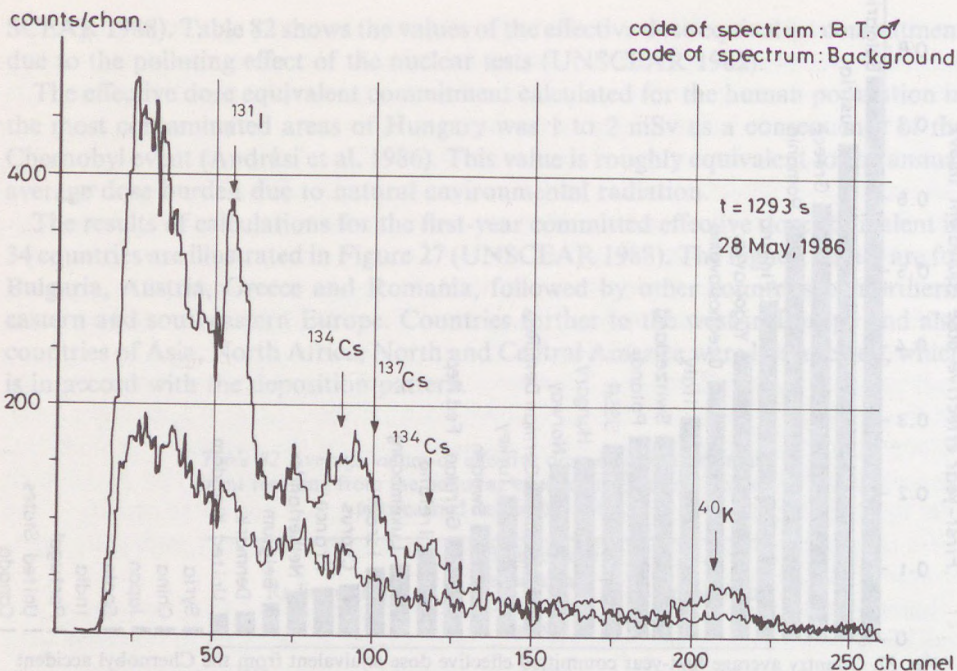


Fig. 29 Scintillation γ spectra measured by the whole body counter

The dose commitments from the accident are delivered over several years, mostly by the continuing exposure to ^{137}Cs . On the average some 30% of the effective dose equivalent commitments are delivered in the first year following the disaster. The regional average dose commitments are shown in Figure 28.

The radiation dose for an individual from the intake of a radionuclide depends on the activity of the ingested material and on other variables such as age, metabolic kinetics, size, etc. For example, the dose for an infant will usually differ from that for an adult for the same activity ingested. Where the dose for a critical age group in the population is of interest, differing dose per unit intake factors have been developed for the different age groups. The recommended committed effective or organ (thyroid) dose equivalents per unit intake (Sv/Bq) for three age groups are shown in Table 83 (WHO 1988). The dose for a critical group should be calculated from the age-specific dose factor and the food consumption data for that group.

Last but not least, we have to mention that a basic purpose of the assessment of the internal radiation burden on an organism is the estimation or calculation of the radiation load on the different organs (e.g. on lungs or on gonads). The absorbed dosage (D) due to the radioisotope accumulated within the organ in question can be calculated as follows:

$$D = k \frac{\varepsilon}{m} \int_{t_1}^{t_2} g(t) dt,$$

Table 83 Age-specific dose per unit intake factors for different radionuclides (Sv/Bq)^a

Radionuclide	One-year-olds	10-year-olds	Adults
Strontium-90 ^b	1.1×10^{-7}	4.0×10^{-8}	3.6×10^{-8}
Iodine-131 ^c	3.6×10^{-6}	1.0×10^{-6}	4.4×10^{-7}
Cesium-134 ^b	1.2×10^{-8}	1.2×10^{-8}	2.0×10^{-8}
Cesium-137 ^b	1.0×10^{-8}	1.0×10^{-8}	1.3×10^{-8}
Plutonium-239 ^b	2.4×10^{-6}	1.4×10^{-6}	1.3×10^{-6}

^a Interim values adopted until values have been recommended by ICRP

^b Committed effective dose equivalent

^c Committed dose equivalent to the thyroid

where:

- ε = the so-called effective energy absorbed in an organ during radioactive decay, as a function of the shape and size of the organ as well as the kind of radioactive isotope;
- m = mass of the organ;
- $g(t)$ = number of nuclides decayed within a time unit in the organ, i.e., the activity;
- $t_2 - t_1$ = time interval, which the dosage burden will be calculated for;
- K = constant which depends on the choice of units.

2.6.2 Radioactivity of human somatic tissues, factors influencing radioactive contamination

Considering their origin, the radioactive substances in the body tissues, can be natural and artificial isotopes. The most important internal radiation sources occurring within the human organism are ^{40}K , ^{14}C , ^{226}Ra and derivatives of the latter.

Potassium, an essential macroelement, occurs in the human organism in a well-defined concentration range (homeostatic regulation), and the K concentration of the organism is a function of age and also sex. As we can assess on the basis of the data in Table 84, the K concentration, expressed in g/body mass kg, reaches its maximum at the age of about 10 years, and it has a lower value in females than in males (Bogolyubov 1975). This latter difference is due to the different body composition of women and men, because the proportion of muscles in men is over 40%, while in women it is lower than 40% and K accumulates mostly in the muscle tissues (Gladisheva 1977). Thus the radiation load in the inner organs of men is always somewhat higher than in women. Only about 15 g of K occurs in the bones of an average man with a weight of 70 kg, having a K content in the whole body of about 140 g.

^{14}C is permanently present in the organism as part of natural carbon. The changes in ^{14}C activity in the environment can also be monitored by the measurement of ^{14}C activity in human somatic tissues, because in consequence of the continuous metabolic exchanges—the biological half-life is about 1–2 weeks—the specific ^{14}C activity of the human organism quickly attains equilibrium with the activity level of

the environment (here food). The radiation burden from the incorporated ^{14}C isotope on the human organism is substantially lower than the burden due to the presence of the ^{40}K isotope, and under the present contamination circumstances, as we have already mentioned, the additional ^{14}C activity due to the nuclear tests is already smaller than 30% of the natural background activity level. If polluting nuclear weapon tests are not continued and the emission from nuclear facilities does not contaminate further the environment, at the turn of millennium the specific

Table 84 The K content in the human organism

Age (years)	K content, g/kg body mass	
	men	women
3-7	2.72 ± 0.01	2.27 ± 0.01
7-12	2.75 ± 0.02	2.55 ± 0.07
12-17	2.36 ± 0.04	2.15 ± 0.06
17-26	2.26 ± 0.03	1.76 ± 0.04
26-45	2.08 ± 0.02	1.66 ± 0.02
45-55	1.90 ± 0.03	1.46 ± 0.04
> 55	1.80 ± 0.09	1.38 ± 0.05

activity of ^{14}C in the biosphere will only be a few per cent higher than the natural background level. For the time being, the specific ^{14}C activity coming from food and being incorporated into the human organism is about 220 mBq/g carbon.

^{226}Ra , being an alkali earth metal chemically similar to Ca, accumulates mostly in the bones. Its biological half-life is about 16 years (Bozóky 1974). The Ra content of the whole body of an average man of 70 kg is about $3.1 \cdot 10^{-11}$ g, from which $2.7 \cdot 10^{-11}$ g occurs within the bones. According to a communication by Belova and Lunkina (1967), the Ra activity of the teeth of adult persons is 3.2-7.6 Bq/kg, and their uranium content is $(1.04-7.60) \cdot 10^{-5}$ g/kg.

In addition to the ^{14}C , ^{40}K and ^{226}Ra isotopes, there are also some other radioactive isotopes of natural origin which are responsible for radiation burden. We have to mention ^{87}Rb , as well as ^{210}Pb and ^{210}Po . Rubidium occurs in human organisms in an average concentration of 10 mg/kg, and 27.85% of the natural rubidium is a radiating isotope. Its half-life is so long that the radiation load due to the ^{87}Rb isotope is smaller by 1-2 orders of magnitude than the dosage due to the ^{40}K isotope (UNSCEAR 1977). Incidentally, the chemical characteristics of rubidium, being an alkali metal, are similar to those of potassium, so rubidium participates in the metabolic processes together with potassium and it accumulates mostly in the musculature. It is not an essential element.

According to investigations carried out in the Soviet Union and Poland, the daily ^{210}Pb uptake is about 0.18-0.20 Bq (Yermolayeva-Makovskaya 1968; Jaworowski 1969). A better part of the ^{210}Pb content (as well as the stable lead) quits the organism together with the faecal matter; thus it is not taken up from the intestine during digestion (Kehoe 1964). According to the measurements of Morse and Welford (1971)

only 7.4–9.4% of the ^{210}Pb isotopes coming into the organism reach the blood. Holtzman (1963) investigated the sources of ^{210}Pb accumulated in the bones. The results of his measurements may be seen in Table 85. The data in the table demonstrate that basically there are two ^{210}Pb sources: one is inhalation and the other is foodstuffs.

Although the decisive part of ^{210}Pb , about 70%, is accumulated in the bones, the ^{210}Pb and ^{210}Po activity of the soft somatic tissues is also not negligible. Table

Table 85 Sources of the ^{210}Pb content of human bones

Sources	%
Decay of ^{226}Ra in the bones	2.3
Decay of ^{222}Rn in the body liquids	4.0
Decay of inhaled ^{222}Rn daughter elements with short half-lives	2.3
^{210}Pb inhalation	41.4
^{210}Pb activity of drinking water	12.5
^{210}Pb activity of food	37.5
Total	100.0

Table 86 The ^{210}Pb and ^{210}Po activity of soft human tissues or organs

Tissue or organ	Activity, Bq/kg	
	^{210}Pb	^{210}Po
Liver	0.12–0.65	0.16–2.13
Kidney	0.06–0.44	0.07–1.71
Spleen	0.02–0.42	0.03–0.49
Lung	0.05–0.37	0.05–0.58
Musculature	0.04–0.22	0.02–0.22
Thyroid gland	0.09–0.54	0.06–0.47

86 provides information about the ^{210}Pb and ^{210}Po content of some somatic tissues or rather organs (Yermolayeva–Makovskaya, Litver 1978).

As we mentioned before, smoking influences the amount of ^{210}Pb and ^{210}Po incorporated into the somatic tissues. According to the measurements of Little and McGandy (1966), for example, the mean ^{210}Pb content of the blood in smokers was 64 mBq/kg; at the same time in the blood of non-smokers it had an average value of 28 mBq/kg. Under the influence of smoking, the tissues of the lung receive a rather substantial radiation dosage, e.g. 40 cigarettes smoked daily mean a radiation dosage of about 15 mSv per year in the lung tissues. There is a higher risk of lung cancer, because in the bronchial tubes, in addition to the chemically carcinogenic substances, ^{210}Pb can also be accumulated (Radford, Hunt 1964). The presence of ^{210}Pb within the organism is dangerous in spite of the small additional dosage caused, because the

radioactive elements by their radioactivity—through synergism—increase the undesirable biological side-effects (toxicity, carcinogenic effect) of the different chemical components.

In the present contamination situation, of the radioactive isotopes of artificial origin, we must first of all deal with the presence of ^{90}Sr and ^{137}Cs isotopes in the human organism. The radioactive contamination of the human organism due to nuclear weapon tests reached its maximum value in 1964–1965, followed after a time-lag by changes in the level of radioactivity in other elements of the biosphere. Later it decreased for some years in a significant degree, and since the beginning of the seventies it has stabilized at a rather low level, or rather it decreased further only very little. As we have already mentioned, most of the radioactive isotopes incorporated into the organism came with food (Figure 29).

However, in 1986, because of the Chernobyl disaster, in a substantial part of the European population, serious accumulation of ^{103}Ru , ^{131}I and ^{134}Cs could be observed. Tables 87 and 88 contain the measurements carried out in Budapest (Bíró et al. 1986; Andrási et al. 1986).

Strontium or rather radiostrontium taken up from foodstuffs and drinks through the gastrointestinal tract accumulates mostly in the bones; only a smaller part reaches

Table 87 ^{131}I exposures of residents of Budapest. Averages and ranges for children (4–16 yr) and adults: 1 May–30 June 1986

Period	^{131}I activity (Bq)					
	children			adults		
	average	range	n	average	range	n
05.01.–05.04.			2	159	(60–199)	19
05.05.–05.11.	138	(46–350)	29	179	(76–326)	44
05.12.–05.18.	122	(65–216)	30	213	(71–606)	54
05.19.–05.25.	102	(39–242)	23	213	(80–552)	50
05.26.–06.01.	85	(40–214)	24	152	(50–485)	35
06.02.–06.08.	105	(53–183)	11	230	(35–385)	7
06.09.–06.30.	no data			70	(20–116)	5

n = No. of measurements

Table 88 Mean and range of the whole body ^{137}Cs and ^{134}Cs activities measured in different time intervals

Time interval	Number of measurements	Whole body activity (Bq)	
		^{137}Cs	^{134}Cs
05.05.–05.11.	3	109 (103–116)	*
05.19.–05.25.	2	160 (152–168)	100 (92–108)
05.26.–06.01.	8	248 (94–540)	142 (68–335)
06.02.–06.08.	17	344 (111–875)	171 (60–399)

* ^{134}Cs activity cannot be evaluated

the musculature and the inner organs. According to Csupka (1970), about 65% of the non-natural radiation load in the bone tissues is caused by the presence of ^{90}Sr . (The remaining 35% arise partly from other artificial radioactive isotopes (e.g., ^{89}Sr , ^{140}Ba) present in the bones in a lower concentration, and partly from the dosage burden to ^{137}Cs being present in the surrounding soft somatic tissues.)

Only part of the strontium coming into the organism with foodstuffs is taken up; thus the human organism, like the animal's body, discriminates well against strontium (Nevstruyeva et al. 1972). Therefore the difference between the $^{90}\text{Sr}:\text{Ca}$ proportion in the diet and the $^{90}\text{Sr}:\text{Ca}$ proportion in human bones is significant; the value of the

$$D_{\text{Sr}} = \frac{{}^{90}\text{Sr}:\text{Ca in the bones}}{{}^{90}\text{Sr}:\text{Ca in the diet}} = \frac{\text{Sr}:\text{Ca in the bones}}{\text{Sr}:\text{Ca in the diet}}$$

discrimination factor is according to measurements carried out in Denmark, Australia, England and the USA on the average 0.20, 0.22 and 0.27–0.28, respectively (UNSCEAR 1977). However, considering the fact the biological half-life of ^{90}Sr is a few years (Snyder, Cook 1964; Nevstruyeva et al., 1972), the amount of radiostromium accumulated in human bones follows only after a very long time-lag the changes of the radioactive contamination level in the environment, i.e., in the food. To put it in another way, the Ca–Sr system, because of the long biological half-life, virtually does not react to the changes of radioactivity levels occurring in relatively short periods. That means that ^{90}Sr is assimilated into the bone structure relatively slowly; however, its release also takes a long time. This is also demonstrated by the data in Table 89, which show the average ^{90}Sr activity of vertebral bones investigated between 1962 and 1974 in the Soviet Union, USA and Denmark (Marey, Borisov 1972, 1975). It can be stated that the maximum value was measured in 1965; later the level of ^{90}Sr contamination gradually decreased.

Table 89 Average ^{90}Sr contamination of vertebrae in adults

Year	^{90}Sr activity, mBq/g Ca		
	Soviet Union	USA	Denmark
1962	41	37	30
1963	59	59	44
1964	89	74	89
1965	115	78	100
1966	100	78	96
1967	85	70	78
1968	63	70	70
1969	59	63	48
1970	52	59	48
1971	56	52	67
1972	59	52	59
1973	59	48	56
1974	48	48	52

A much sharper change is demonstrated by the activity of ^{137}Cs in human somatic tissues (see Table 90). It is due to the relatively short biological half-life of ^{137}Cs . The data in Table 90 on the amount of ^{137}Cs reaching and accumulating in the somatic tissues of people living in Soviet towns (Marey 1980) demonstrate that the maximum value measured in the years 1964–1965 decreased in 10 years by one order of magnitude. The rapid decrease of the ^{137}Cs activity is also supported by the data of Andrási and Fehér (1969), Poulheim (1971) and Sztanyik (1980). Their data demonstrate that in 1965 the average ^{137}Cs contamination of the human somatic tissues in Hungary and the GDR was 10–15 Bq/kg; however, in 1967–1968 it decreased to a value of 3–4 Bq/kg.

Of the inner organs, the liver accumulates most of the ^{137}Cs nuclides. In the USA, the ^{137}Cs contamination of the liver of adults referred to 1 kg of original tissues was 6.22, 3.77 and 3.07 Bq in 1965, 1966 and 1967, respectively. The ^{137}Cs contamination of the lung was in the same years 4.06, 2.18 and 1.04 Bq, respectively; that of the heart was 5.55, 3.51 and 2.07 Bq, respectively.

Table 90 ^{137}Cs accumulation in the human body

Year	Daily ^{137}Cs uptake (Bq/day)	Average ^{137}Cs activity of the body	
		Bq/whole body	Bq/g K
1963	7.84	1130	8.07
1964	10.66	1540	10.95
1965	8.33	1200	8.55
1966	5.37	770	5.51
1967	3.48	500	3.55
1968	2.07	300	2.15
1969	1.44	207	1.48
1970	1.78	255	1.81
1971	1.74	252	1.79
1972	1.26	181	1.30
1973	1.07	155	1.11
1974	1.00	144	1.03
1975	0.93	133	0.95

Table 91 Cesium-137 accumulation in the test person (Vonach, Steger 1987)

Date	^{137}Cs content (kBq)	^{137}Cs intake (kBq)	Change in ^{137}Cs content (kBq)
20.10.1986	1.14 ± 0.09	0.55 ± 0.04	0.42 ± 0.14
27.10.1986	1.56 ± 0.11	0.55 ± 0.04	0.62 ± 0.17
04.11.1986	2.18 ± 0.13	0.53 ± 0.04	0.27 ± 0.19
10.11.1986	2.45 ± 0.15	0.56 ± 0.04	0.40 ± 0.23
17.11.1986	2.85 ± 0.17		

We have frequently mentioned already that as a consequence of the Chernobyl disaster, radioactive contamination of the environment and therefore of human somatic tissues increased. The degree in the ^{137}Cs accumulation is shown in Table 91, on the basis of measurements carried out in Austria (Vonach, Steger 1987).

There are many new models for characterizing the assimilation of radionuclides coming into the organism, as well as their excretion (Moskalev 1962; Kulp, Schubert 1962; Eberhardt 1967; Moiseyev, Ramzayev 1975). In general an exponential model with two components is used:

$$Q(t) = Q_0[C_1 e^{-\lambda_1 t} + C_2 e^{-\lambda_2 t}],$$

where:

$Q(t)$ = amount of the radionuclides in question within the organism at time t ;

Q_0 = original amount of radionuclides within the organism at time t_0 ;

C_1 and C_2 = proportions of radionuclides leaving the organism with different speeds;

λ_1 and λ_2 = speed constants of excretion;

t = time.

Actually, part of the radioactive substances leaves the body quickly, while another part, that assimilated within the organism, leaves the organism only with a speed corresponding to the biological half-life. For example, in the case of ^{90}Sr the degree of excretion is determined by the speed of diffusion ion-exchange and recrystallization (Norris et al. 1958; Holtzman 1962; Marey 1980). If the radionuclides are present in food for a long time, the two-component model becomes simpler:

$$\frac{DQ(t)}{dt} = -\lambda Q(t) + fc(t),$$

where:

$c(t)$ = radionuclide content of the food taken up;

f = transfer coefficient of the radioactive substance coming from the food into the human organism.

In subchapter 2.5, dealing with the radioactivity of the animal organisms or rather the study of the factors influencing the radioactive contamination, it was said that the biological half-life of an (active or inactive) element could be approximated as follows:

$$T_{1/2 \text{ biol}} = \frac{a}{b} \ln 2$$

where:

a = amount of the element in question within the organism;

b = (for an essential element (e.g. K)), the daily physiological requirement; (for a non-essential element (e.g. Cs)), the daily average of the amount of the substance absorbed in the gastrointestinal tract.

According to our calculations (Szabó 1980, 1981), the mean biological half-life of the K-Cs system in adults is about 81 days. This result was obtained by supposing that, considering the data in the literature (Lloyd et al. 1973; Bogolyubov 1975; Katz

1975), the K content of an average human organism is 140 g, and the daily K requirement is 1.2 g. Considering the above-mentioned data, the function describing the decrease of the potassium content of the organism is as follows:

$$y = 140e^{-0.00857n}$$

where:

n = number of days.

Of course this exponential function, because of the highly complex metabolism of the human organism, is only an approximation. By the way, the claim that the biological half-lives of K and Cs (having a common transport system) as well as of Ca and Sr are the same is also true only more or less.

Considering the real possibility that either the total potassium content of the organism, or the daily need of K can be changed significantly as a function of sex, body mass and age, Table 92 lists the biological half-lives rounded to integer days as a function of the whole potassium content between 80 and 170 g, and the daily K requirement between 0.8 and 1.6 g. We have to mention that the result is very similar to the average 81 days biological half-life, if the calculation is based not on the potassium content, but on the inactive Cs content of the body and on the amount of Cs coming daily into the human somatic tissues. The mean biological half-life of the K-Cs system, or rather that of ^{137}Cs , is 115 days according to Lindbom (1964), 100 days according to Marey (1980), 84 days according to Hasanen and Rahola (1971), 82 days according to Ramzayev et al. (1969), and 64 days according to measurements carried out in Hungary in the Central Research Institute for Physics.

It seems necessary to stress that generally an amount much higher than is physiologically necessary is taken up every day from the foodstuffs and drinks into the human organism. Thus, for example, according to Schelenz (1977), the amount of daily K uptake is 1.2–4.1 g; according to Ketz (1975), it can be as high as 5 g; according to Fietrzak-Flis (1969) it is on the average 2.5–2.8 g. According to our dietetic survey carried out in 1974, the daily mean potassium uptake is 2.0 g. This means that the daily mean potassium uptake is quite often substantially higher than

Table 92 Biological half-life of K-Cs system (rounded to days) as a function of the daily K need and the amount of K in the human organism

Daily need g	Amount within the organism, g									
	80	90	100	110	120	130	140	150	160	170
0.8	70	78	87	96	104	113	122	130	139	148
0.9	62	70	77	85	93	100	108	116	124	131
1.0	56	63	70	77	84	90	97	104	112	118
1.1	51	57	63	70	76	82	89	95	102	107
1.2	47	52	58	64	70	75	82	87	93	99
1.3	43	48	54	59	64	70	75	80	86	91
1.4	40	45	50	55	60	65	70	75	80	85
1.5	37	42	47	51	56	60	65	70	75	79
1.6	35	39	43	48	52	56	61	65	70	74

Table 93 Biological half-life of Ca–Sr system (rounded to days) as a function of the daily Ca need and the amount of Ca in the human organism

Daily need g	Amount within the organism, g									
	800	900	1000	1100	1200	1300	1400	1500	1600	1700
0.2	2773	3120	3466	3813	4159	4506	4852	5199	5546	5892
0.3	1849	2080	2311	2542	2773	3004	3235	3466	3697	3928
0.4	1387	1560	1733	1907	2080	2253	2426	2600	2773	2946
0.5	1109	1248	1387	1525	1664	1803	1941	2080	2218	2357
0.6	925	1040	1156	1271	1387	1502	1618	1733	1849	1964
0.7	793	892	991	1090	1189	1288	1387	1486	1585	1684
0.8	693	780	867	953	1040	1127	1213	1300	1387	1473

the need of the organism. However, this superfluous amount quits the organism directly with the faecal matter; hence it does not participate in the metabolic process-es. It does not have any influence on the biological half-life.

Since almost 100% of cesium and only a fraction of potassium is taken up by digestion, the $^{137}\text{Cs}/\text{K}$ proportion in the human organism is usually higher than in the diet. Actually, the measure of discrimination is a function of the potassium content of the food consumed. According to measurements carried out in the USA, the value of the

$$D_{\text{Cs}} = \frac{^{137}\text{Cs}/\text{K in the body tissue}}{^{137}\text{Cs}/\text{K in the diet}} = \frac{\text{Cs}/\text{K in the body tissue}}{\text{Cs}/\text{K in the diet}}$$

discrimination factor varies between 2 and 3, consequently the human organism accumulates cesium as opposed to potassium (UNSCEAR 1977).

The biological half-life of the Ca–Sr system can be calculated in a similar way to that of the K–Cs system. If, on the basis of literature data (Schulze-Wollgast 1972; Ketz 1975), the mean Ca content of an average human body is 1200 g, and the daily Ca requirement is 0.5 g, the Ca–Sr biological half-life is about 1664 days. Using these data, the function describing the decrease of the calcium content of the organism is as follows:

$$y = 1200e^{-0.000417n}$$

where:

n = number of days.

Like the K–Cs system, the Ca content of the organism and the daily Ca requirement is influenced by body mass, age, sex and physical activity. Table 93 lists the rounded biological half-lives in relation to the total Ca content between 800 and 1700 g and to 0.2–0.8 g daily Ca requirement. Just as for Cs, the biological half-lives can be also determined knowing the total inactive strontium content of the body and the amount of the daily Sr assimilation. According to our calculations (Szabó 1980, 1981), there is good agreement between the biological half-lives calculated on the basis of either the Ca content or the Sr content.

It must be mentioned that the amount of Ca coming daily from our foodstuffs and drinks into our organisms is usually higher than the physiologic requirements. The

daily Ca uptake usually has a value of about 1.0 g (Schulze-Wollgast 1972), according to our dietary survey it is on the average 0.9 g.

Table 94 lists data on the metabolism of the inactive Ca-Sr system. It can be seen that the majority of strontium is not taken up and leaves the organism together with the faecal matter. This is the basis for the good discrimination ability of the human organism against strontium. The fraction coming into the metabolism of the organism is indicated by the fraction leaving the organism with urine, and thus the biological

Table 94 Sr and Ca metabolism in adult human organism (mg)

Characteristic parameter	Sr	Ca
Amount taken daily with food	1.7	1200
Amount assimilated from food	0.3	450
Amount leaving with faecal matter, by-passing absorption	1.4	750
Amount leaving daily with urine	0.3	250
Half-life in body fluid (days)	2-3	3-4
Average half-life in bones (days)	2300	2770

half-life can be calculated. For cesium, which has an important role in the long-lived contamination, the situation is different, because it is easily taken up from the gastrointestinal tract. It takes part in the metabolism, and it leaves the organism mostly with urine, according to its biological half-life.

The radioactive isotopes accumulated within the organism usually leave together with urine. From the amount of ^{90}Sr leaving the organism with the urine the total amount of ^{90}Sr in the organism can be calculated. The relationship is as follows:

$$A = a + bx^{1/3}$$

where:

A = activity, within the organism;

x = amount of activity leaving the organism daily with the urine;

a and b = constants.

Age has a significant influence on the rate of uptake of radioactive isotopes coming into the organism, their accumulation within the organism, and their excretion. For example, in infants and small children the ^{90}Sr uptake and exchange is relatively fast: the biological half-life is shorter than 1 year. However, in adult age the exchange slows down. The biological half-life of Sr is many years—according to our calculations it is about 5 years; according to Table 94 it is about 6-7 years. Because of the short biological half-life and the more intensive assimilation (there is no equilibrium between the amounts of mineral substances taken up from the foodstuffs and eliminated by faecal matter or urine), it may happen that the $^{90}\text{Sr}/\text{Ca}$ proportion in the bones of small children is substantially higher than the same proportion in adults. It can be explained by the radioactive isotopes being quickly assimilated into the organism of small children in response to the increasing radioactive contamination of foodstuffs. However, the adult organism reacts practically only to a prolonged increase of contamination.

On the other hand, it is an interesting observation that the ^{90}Sr activity of the bones in infants is surprisingly low. It can be attributed to the fact that the organism of the mother discriminates well and the $^{90}\text{Sr}/\text{Ca}$ proportion of breast-milk is lower by one order of magnitude than in the food consumed by the mother (Calapaj, Ongaro 1970). The $\text{Sr}:\text{Ca}$ proportion of breast-milk is about 12% of the $\text{Sr}:\text{Ca}$ proportion of the food.

Food habits determine the amount of radionuclides coming into the organism in a decisive way. The radiostrontium content of milk and milk products, meat and meat products, egg and food prepared with flour is low, and in contrast the ^{90}Sr contamination of vegetables, greens and fruits is many times higher. In spite of the fact that about 50–80% of our daily Ca need is covered by milk and milk products, the ^{90}Sr activity coming into the organism with vegetable food plays a determining role.

The proportion of the $^{90}\text{Sr}/\text{Ca}$ quotient in the total diet and in the milk has been measured in a large number of countries. Between 1963 and 1975, in Norway its mean value was 1.3, in France 1.4, in Denmark 1.5, in the GDR 1.7, in Japan 2.7 and in the Soviet Union 3.1 (UNSCEAR 1977). The higher the proportion, the lower the milk consumption.

The situation is different in the case of ^{137}Cs . The ^{137}Cs contamination of food of animal origin is not lower than in plants; actually, it may be even several times

Table 95 Average ^{90}Sr and ^{137}Cs activity of food between 1963 and 1973 in the Soviet Union, in the urban population

Year	Activity, Bq/day		$^{90}\text{Sr} : ^{137}\text{Cs}$
	^{90}Sr	^{137}Cs	
1963	1.41	7.84	0.18
1964	2.26	10.66	0.21
1965	1.59	8.33	0.19
1966	1.07	5.37	0.20
1967	0.78	3.48	0.22
1968	0.70	2.07	0.34
1969	0.59	1.44	0.41
1970	0.67	1.78	0.38
1971	0.67	1.74	0.39
1972	0.59	1.26	0.47
1973	0.56	1.07	0.52

higher. According to the measurements of Schulze-Wollgast (1972), 21–25% of the ^{90}Sr content of the diet and 47–56% of the ^{137}Cs content of the diet can be attributed to milk and meat.

Table 95 shows the average values of ^{90}Sr and ^{137}Cs coming with daily food consumption into the human organism for the urban population, on the basis of measurements carried out in the Soviet Union (Marey 1980). Tables 96 and 97 list the percentile proportions of ^{90}Sr and ^{137}Cs in the organism originating from different

Table 96 Percentage distribution of ^{90}Sr taken up from different kinds of food by the urban population in the Soviet Union between 1963 and 1973

Year	Cereals	Milk	Potato	Vegetables	Fruits	Meat	Fish
1963	56.0	19.6	4.5	5.3	2.0	7.1	5.5
1964	60.4	15.7	6.7	5.3	2.6	4.0	5.3
1965	59.0	15.6	7.1	6.4	2.1	4.0	5.8
1966	48.8	17.4	8.5	6.9	2.1	5.0	11.3
1967	40.4	17.1	10.6	9.0	2.6	6.5	13.8
1968	33.7	17.5	11.7	9.8	3.1	5.9	18.3
1969	35.9	20.5	12.2	9.6	3.8	7.1	10.9
1970	34.3	19.4	13.1	10.3	3.4	7.4	12.1
1971	35.6	20.1	12.6	9.8	5.2	7.5	9.2
1972	32.1	21.6	10.8	11.8	5.0	6.2	12.5
1973	32.0	19.0	12.2	13.1	3.4	6.8	13.5

Table 97 Percentage distribution of ^{137}Cs taken up from different kinds of food by the urban population in the Soviet Union between 1963 and 1973

Year	Cereals	Milk	Potato	Vegetables	Fruits	Meat	Fish
1963	43.3	24.7	9.4	3.8	1.8	15.3	1.5
1964	44.5	17.4	13.0	4.3	2.3	16.2	2.3
1965	40.6	12.0	10.0	4.8	3.6	16.2	12.8
1966	35.7	13.8	7.6	4.6	2.2	21.8	14.3
1967	33.2	14.3	8.6	4.8	1.9	20.6	16.6
1968	28.0	18.8	10.3	5.6	1.9	20.0	15.7
1969	25.9	20.7	13.2	6.2	2.4	21.2	10.4
1970	25.6	19.3	11.7	9.0	1.9	21.8	10.7
1971	27.1	23.0	10.2	7.9	3.0	19.6	9.2
1972	27.7	26.5	9.4	6.3	3.5	20.6	6.0
1973	29.0	25.9	11.2	7.7	2.1	16.1	8.0

kinds of food. It can be seen from these tables that in the Soviet Union, because of the relatively high bread consumption, cereals are the main sources of contamination, both for ^{90}Sr and ^{137}Cs . The proportions in other countries with different nutrition customs can be quite different. In countries where milk and meat consumption is high, the decisive part of ^{137}Cs consumption comes into the human organism through the consumption of foodstuffs of animal origin.

We have to mention here that for 5 days each in February, July and October, 1974, we investigated the whole diet in a secondary school in Győr (Hungary). According to our measurements the gross β activity of the daily diet was 74 ± 28 Bq, the ^{40}K activity was 66 ± 16 Bq, the ^{90}Sr activity was 0.63 ± 0.22 Bq, the ^{137}Cs activity was 3.44 ± 2.41 Bq, or rather the ^{90}Sr activity was 0.70 ± 0.37 Bq/g Ca, and the ^{137}Cs activity was 1.72 ± 1.26 Bq/g K. These values are in good accordance with the data of contamination measured in other European countries (Miserez 1963–1980; Schulze–Wollgast 1969, 1972; Fietrzak–Flis 1969; Mattern et al. 1969; Porzucek 1972;

Moldenhawer et al. 1971; Csupka 1974). It can be stated that the radioactive contamination of the environment and the foodstuffs in Hungary is similar to the data characteristic of other European countries.

In addition to the above-mentioned natural and artificial isotopes, we have to mention tritium, which occurs in substantial concentrations in human somatic tissues. Since most of ^3H comes into the human organism with drinks and drinking water, consequently the mean specific tritium activity of the human body is similar to that of drinking water, since on average 70% of the human body mass is water. The biological half-life is short (1–2 weeks); thus the organism quickly follows the changes in the contamination of the environment (here chiefly the drinking water). In spite of the fact that the concentration of ^3H in our foods is comparatively high in relation to the other radioactive isotopes, because of its very low radiation energy the radiation dosage received from tritium represents a low dose level. According to Miserez (1980), for example, the radiation dose received from ^3H is smaller by 4 orders of magnitude than the dose due to ^{40}K .

We have seen that in the air–water–soil–plant–animal–man biological chain the relative concentration of the different radioactive isotopes changes from one member of the chain to another. The degree of change is indicated by the discrimination factors. For instance, considering only the stages following each other, the connection between the radioactivity in the soil and in the human somatic tissues can be characterized by a common discrimination factor:

$$D = D_1 \cdot D_2 \cdot D_3 \cdot x_1 + D_1 \cdot D_3 \cdot x_2$$

where, for the radioactive element in question, D_1 , D_2 and D_3 mean the discrimination factors for the plant : soil, animal : plant and man : food steps, respectively, and x_1 and x_2 ($x_2 = 1 - x_1$) stand for the average proportion of the animal and vegetable part of the diet, respectively.

Of course this relationship has only theoretical importance; actually there is no close correlation between the radioactivity of the soil and human somatic tissues. On the one hand, radioactive substances can also come into the human (and animal) organism through drinking water and inhalation; on the other hand, the plants can take up radioactive isotopes not only from the soil but also through their leaves from the air and rainwater. Last but not least, the value of the discrimination factor is in turn determined by several other factors (plant and animal species, age, food habits in man, etc.).

Although we have mentioned it in the sub-chapter dealing with the radioactivity of animal organisms, we feel it is necessary to stress again that the speed of release of radioactive substances, and at the same time the degree of radiation load on the organism, is determined by the effective half-life ($T_{1/2 \text{ eff}}$), which in turn can be calculated from the physical half-life ($T_{1/2 \text{ phys}}$) and the biological half-life ($T_{1/2 \text{ biol}}$) as follows:

$$T_{1/2 \text{ eff}} = \frac{T_{1/2 \text{ phys}} \cdot T_{1/2 \text{ biol}}}{T_{1/2 \text{ phys}} + T_{1/2 \text{ biol}}}$$

It can be seen that for a relatively long physical half-life

$$T_{1/2 \text{ eff}} = \frac{T_{1/2 \text{ phys}} \cdot T_{1/2 \text{ biol}}}{T_{1/2 \text{ phys}}} = T_{1/2 \text{ biol}}$$

i.e., there is almost no difference between the effective and biological half-lives. If, for example, in the case of an adult the biological half-life of the K-Cs system is supposed to be 100 days, the effective half-life will be 99.1 days.

2.6.3 Possibilities for decreasing the radiation burden

The human organism receives external and internal radiation loads. Therefore, when discussing the possibilities of decreasing the radiation burden, we have to treat separately, on the one hand, the possibilities of reducing the radiation load from external sources, and on the other hand, those which are devised to reduce the internal radiation load due to the radioisotopes incorporated into the human organism.

When discussing the different components of the radiation dosage affecting the human organism, we mentioned that a significant part of the radiation load is due to the ^{222}Rn content of air (mostly affecting the tissues of the lung). Since the ^{222}Rn activity of the air in the inner spaces of closed buildings can reach rather high levels (due to the ^{226}Ra decay in the walls) the dosage on the organism can be decreased by frequent ventilation. Of course, the radon is not the only radioactive substance in the air and through inhalation even other radioactive particles, e.g. radioaerosols of artificial origin, may come into the organism and may partly be incorporated there. A detailed survey of calculating methods of radioactive radiation due to radionuclides coming into the organism through inhalation has been compiled, e.g. by Zarayev (1975).

Another factor in the external radiation load is the altitude of residence. The higher the altitude, the greater the radiation dosage due to cosmic radiation. In certain cases the influence of geological factors is also not negligible. Where the uranium content of the lithosphere or rather the soil is high, the living organisms receive greater radiation load due to the higher natural background level.

The distance from surface nuclear weapon tests also plays a decisive role, as for example in certain islands of Micronesia in the Pacific Ocean, where American nuclear weapon tests were carried for a decade from 1952 onwards, the level of radioactivity is so high even today that the islands are uninhabitable. Serious radioactive contamination could be measured in the immediate environment of the Chernobyl nuclear reactor, too. As it is well known, in a region of a radius of 30 km the population had to be relocated and thus more than 100 000 persons were forced to change residence. Figure 30 shows the geographic location of Chernobyl.

Efforts should be made to decrease the amount of radiation load of medical character (with a diagnostic purpose) to a minimum. Only a very limited number of X-ray pictures should be prepared and only in the most justified cases. The different sensitivities of the different organs must be stressed (e.g. the hand is much less sensitive than the gonads).

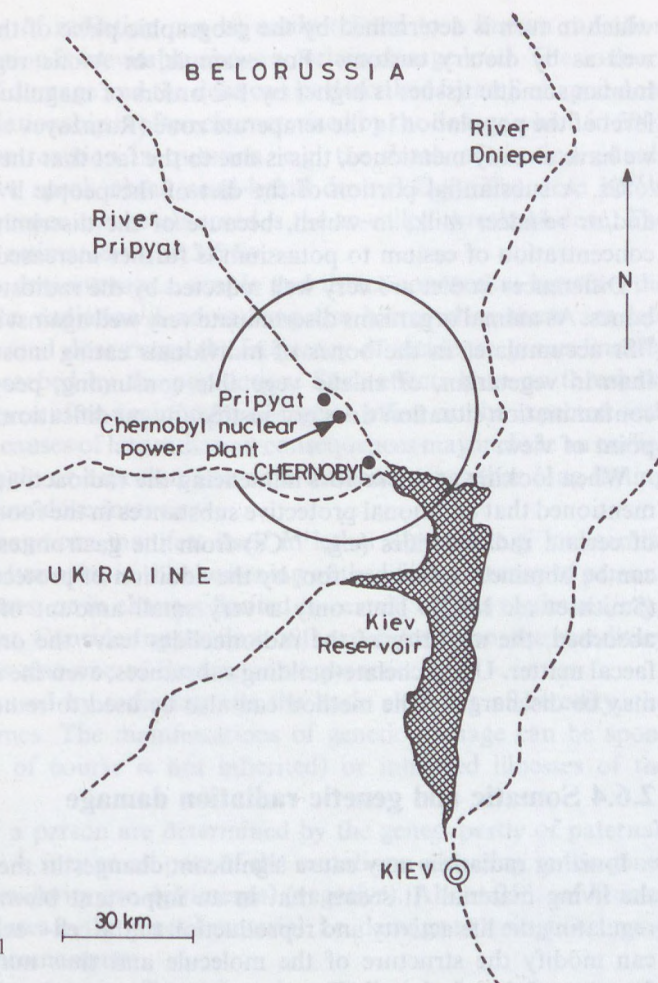


Fig. 30. The site of the Chernobyl nuclear power plant

More and more people receive radiation from external sources as a consequence of their daily work. The great importance of complying with the regulations of radiation protection and health physics as well as the efforts to decrease the amount of radiation exposure cannot be stressed enough. Some somatic and even genetic damage may be caused even by small doses, so utmost care and prudence is requested from those who work with open and closed radioactive isotopes, nuclear facilities, and X-ray equipment. It is no coincidence that, according to researches carried out in different countries, the prevalence of leukaemia among radiologists is higher than in the general population (Ichimaru, Ishimaru 1977). This fact points to unsuitable measures of radiation protection being used during earlier decades.

The internal radiation load is caused by radioisotopes incorporated into and accumulated in the human organism. The amount of radioactive substances in the organisms is determined chiefly by the activity of the consumed foodstuffs and drinks,

which in turn is determined by the geographic place of the radiation in question, as well as by dietary customs. For example, in Arctic regions the ^{137}Cs activity in human somatic tissues is higher by 1–2 orders of magnitude than the contamination level of the population in the temperate zone (Ramzayev et al. 1973; Marey 1980). As we have already mentioned, this is due to the fact that the Arctic vegetation enriches ^{137}Cs . A substantial portion of the diet of the people living there is reindeer meat and/or reindeer milk, in which, because of the discrimination ability, the relative concentration of cesium to potassium is further increased.

Differences in diet are very well reflected by the radiostrontium content of human bones. As animal organisms discriminate very well against strontium, significantly less ^{90}Sr accumulates in the bones of individuals eating mostly foods of animal origin than in vegetarian, or chiefly vegetable consuming, people. However, the present contamination situation does not justify any modification of dietary habits from this point of view.

When looking at the factors influencing the radioactivity of animal organisms, we mentioned that additional protective substances in the food may block the absorption of certain radionuclides (e.g. ^{137}Cs) from the gastrointestinal tract. Similar results can be obtained in humans too, by the addition of protective substances to the food (Smith et al. 1982). Thus only a very small amount of radioactive substances is absorbed; the majority of the radionuclides leave the organisms together with the faecal matter. Using chelate-building substances, even the incorporated radionuclides may be discharged. The method can also be used to remove toxic heavy metals.

2.6.4 Somatic and genetic radiation damage

Ionizing radiation may cause significant changes in the structure and function of the living material. It seems that in an important biomolecule—e.g. in the DNA regulating the life activity and reproduction of the cell—even a single ionization event can modify the structure of the molecule and thus may disturb the harmonious function of the whole cell. Thus it may be supposed that even the smallest radiation dose can cause biological damage.

On the one hand, the damage caused by radioactive radiation can manifest itself in the destruction of cells assuring the survival of the individual. This is called somatic damage. On the other hand, damage to the cells responsible for reproduction and heredity is genetic. The severity of biological reaction is determined by the measure of radiation, i.e., the dose absorbed in the organism, the type and energy of radiation, the duration and intensity of radiation, the size of the part of the body under radiation influence and the biological importance of the tissues and organs in question, as well as their radiation sensitivity and regeneration capacity. Usually, the greater the radiation dose, the more serious is the damage caused.

The early consequences of radiation are manifested within only hours or days after the effect; the late consequences can be observed usually months or years after the radiation dose.

The early consequences of radiation can be easily related to a known radiation effect, and their manifestation is inevitable above a certain dosage level. The connection between the degree of damage and the dose can be described by an S-shaped dose efficiency diagram. The relationship is often characterized by the dosage which in 50% of the cases produces the reaction in question, e.g. the death of the individuals irradiated. In this case we speak about semi-lethal dose (LD_{50}). The dose below which virtually no consequence can be observed is the so-called threshold dose. The value of LD_{50} for man is estimated to be 3.5 Sv.

It is usually difficult to demonstrate a simple and direct connection between the degree of damage and the radiation load causing the harm when there are late consequences. Within a broad dose range the frequency of incidence grows linearly with the average dose absorbed by the population. Such effects have no threshold levels; as a consequence, even after a minimum dosage an effect can be expected with a certain probability. The causes of late radiation consequences may include an earlier acute radiation effect, single or repeated low-dose influence or rather long-lasting radiation influence with low-dose intensity.

The late radiation damage can manifest itself in lethal illnesses, e.g. leukaemia, cancer or other malignant tumours, in the shortening of the life expectancy of persons suffering from other diseases, or in changes limited to certain organs or tissues, e.g., chronic dermatitis, cataract. Considering the time of the manifestation, the hereditary damage of the offspring is also one of the late consequences.

The genetic damage caused by radiation hits the basic elements of heredity, the genes and the chromosomes. The manifestations of genetic damage can be spontaneous abortion (which of course is not inherited) or inherited illnesses of the offspring.

The genetic features of a person are determined by the genes, partly of paternal, partly of maternal origin. A very small part of the mutations occurring in the genes are useful (positive); the majority are detrimental (negative). About 25% of human diseases are caused by failures in the genetic material, i.e. damage to a single or many genes, or to one of the chromosomes.

In the evaluation of the genetic effect of a given radiation dose, the so-called doubling dose is an important parameter. This is the dose which doubles the natural incidence of hereditary diseases. Knowing this value, it is possible to assess the risk of genetic damage. Different authors, on the basis of different assumptions, put the value of the doubling dose somewhere between 0.3 and 2 Sv. According to UNSCEAR (1982), it is 1 Gy (in the case of radiations with small LET).

As the probability of having a child quickly decreases after the age of 30, the "genetic importance" of the gonad radiation burden is not the same at all ages. (Incidentally, in Hungary the average age of parents, both mother and father, is 29 years.) Therefore when assessing the danger of genetic injuries caused by radiation on a given population, the age distribution should also be considered. Knowing that even the smallest radiation dose can cause both somatic and genetic damage, and that the frequency of genetic mutations is a function of the dose received, all superfluous radiation load is to be avoided and the received radiation dose should be kept at the lowest possible level.

It should be mentioned that the issues of the somatic and genetic effect of radiation were discussed in detail, for example, in the proceedings of the ICRP conference in Amsterdam in 1983 (Broerse et al. 1983).

2.6.5 Biopositive effect of low-level radiation?

It seems that finding the real way of extrapolation from high-dose range to solve the effect of low-level radiation need not necessarily be the only approach to the problem. In any case, because it will surely come to a below-level conclusion, which will bring the "threshold theory" on the scene, the influence of doses below the "threshold" would still remain a question.

The expression "radiostimulation" originates from *Breslavets*, who coined the term for the positive biological influence of radiation treatments on the organism. It is known that stimulation with an optimum radiation dose has favourable effects, e.g. on plants (e.g. faster germination, yield-increase), (Berezina 1975; Simon, Bhattachariya 1977).

Figure 31 shows the stimulation effect of low-dose radiation (Luckey 1980). The responses to low doses of radiation suggests that the ambient (natural background) radiation level is insufficient for the optimum supply of the essential functions. Therefore radiation doses somewhat above the ambient levels should be beneficial for many physiological functions of the living organisms.

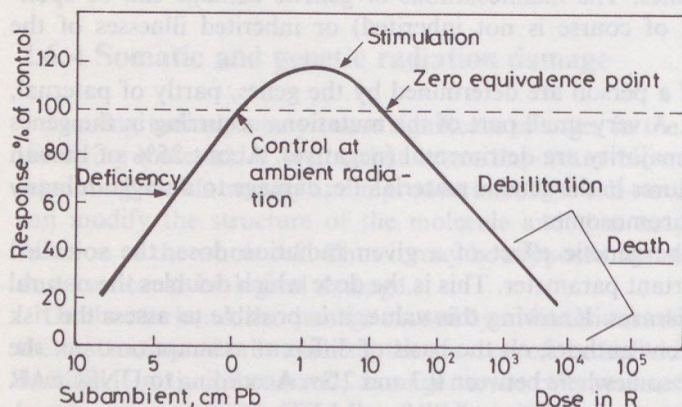


Fig. 31. Complete dose-response curve for ionizing radiation. The zero equivalence point (ZEP) shows there is no detectable response at this dose according to the parameter being tested

The body of evidence for the beneficial effects of small doses of radiation is growing (e.g. Luckey 1982), and this could not be explained by any extrapolation techniques. On the other hand, the indirect mechanism is bringing not only a new view, but also some explicit aspects like LET dependence of ionization rate, and slight, but definite, evidence of dose-rate effect, etc. At higher doses the direct effect is probably predominant, while within the low-dose range the indirect effect takes place and both effects seem strictly different in their consequences. While direct action is somatically and genetically harmful, the indirect influence may be generally beneficial. The graphic

image of this idea is given in Figure 32 for humans (Spurny 1984). It may be seen that below 0.01 Gy there is room for only biopositive influence.

Finally, I would like to mention the observations made in Hungary in May and June 1986 after the Chernobyl disaster. Various vegetables grew very rapidly. Perhaps this was due to the favourable weather conditions (high rainfall); or it is also possible that the increased radiation level had a biopositive influence on the growth of plants. The dose burden of the plant tissues in Hungary was in the range of stimulation, and not in that of inhibition (Szabó 1987, 1989).

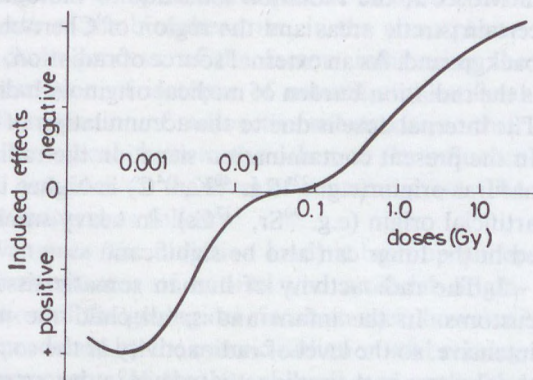


Fig. 32. Graphic scheme of the suggested mechanism

The question whether the radioactive contamination due to the Chernobyl accident had any stimulation effect on the plants was discussed and analysed at the XVIIIth annual ESNA (European Society of Nuclear Methods in Agriculture) meeting in Stara Zagora, Bulgaria, 30 August–4 September 1987 (Szabó 1987). This problem was discussed in 2 different working groups (W.G.2 “Radiation induced stimulation effects in living organisms” and W.G.7 “Environmental pollution”). Many specialists, e.g. Eriksson (1987) from Sweden and Dmitriev (1987) from the Soviet Union, reported rather similar observations, i.e., a significant increase of the green biomass of various plants, probably in consequence of the higher radioactive contamination level.

Of course, to test this hypothesis (the biopositive influence of the radioactive contamination from Chernobyl), it would be necessary to carry out a statistical analysis. The problem is, what to consider as a control for comparison. First of all the excess radiation level was rather different in the different locations of the country. Secondly, growth and yield depends also on other parameters (e.g., weather conditions, fertilization), so it is not possible to carry out a proper analysis.

2.6.6 Summary

In this subchapter the main sources of the radiation load on the human organism and the proportions of the different doses were discussed. Data were presented on the amount of radioactive substances coming into the human organism, and on the radioactive contamination of the different human somatic tissues. The possible mea-

tures of decontamination were related, and last but not least, the issues of somatic and genetic damage and the possible biopositive influence of low-dose radiation (radio-stimulation) were analysed. The most important statements are as follows:

1. There is external and internal radiation load on the human organisms, both of which may be of natural and artificial origin. Most of the external radioactive radiation is due to terrestrial radiation, to radiation from buildings or rather the air within them, and to cosmic radiation. As the environmental contamination caused by nuclear weapon tests has decreased since the partial Nuclear Test Ban Treaty in 1963 in Moscow, the radiation load due to radioactive contamination (not considering certain Arctic areas and the region of Chernobyl) is usually lower than the natural background. As an external source of radiation, most of the dosage of artificial nature is the radiation burden of medical origin with diagnostic and/or therapeutic purpose. The internal dose is due to the accumulation of radioactive isotopes in the organism. In the present contamination situation the radiation load from the radionuclides of natural origin (e.g. ^{226}Ra , ^{40}K , ^{14}C) is higher than the dose due to the isotopes of artificial origin (e.g. ^{90}Sr , ^{137}Cs). In heavy smokers the amount of ^{210}Po accumulated in the lungs can also be significant.

2. The radioactivity of human somatic tissues is a function of age and dietary customs. In the infant and small child the rate of assimilation and exchange is intensive, so the level of radioactivity in the somatic tissues follows relatively quickly the changes in the radioactivity level in the consumed food. In the adult the biological half-life becomes longer, the rate of metabolism slows down, and the organism reacts only with a relatively long time-lag to the possible lasting changes in the radioactivity of food. Dietary customs have an influence on the proportion of the amounts of ^{90}Sr and ^{137}Cs coming into the organism. The strontium and radiostrontium content of foodstuffs of animal origin is significantly lower, because of the very good discriminating ability of animal organisms against strontium, than that in foods of vegetable origin. Strontium accumulates mostly in the bones, cesium in the muscles.

3. The human organism, like animal organisms, discriminates against different chemical elements or radionuclides in different ways. In the human somatic tissues, the Sr:Ca proportion is usually significantly lower, and the Cs:K proportion is usually significantly higher than the proportions measured in the food consumed.

4. In consequence of external and/or internal radiation loads on the organism, both somatic and genetic damages may be incurred. As the radiation effect has no threshold dose, it is very important that the radiation load on the organism should be kept at the very minimum. In the present contamination situation, the radioactive contamination of the foodstuffs and thus of the human somatic tissues, or rather the radiation burden, is lower than the natural background, so there is no need to apply any decontamination methods. The contamination level in Hungary is similar to that in other European countries, the radioactive contamination of human tissues was at the beginning of the 1980s lower by one order of magnitude than the contamination maximum in the period 1964–1965. In 1986, as a consequence of the Chernobyl disaster, radioactive contamination of human tissues increased significantly in several European countries, and among the radionuclides with long half-lives mostly ^{134}Cs and ^{137}Cs were accumulated.

5. A very interesting field of radiation biology is the radiation effects of small doses. Supposing that there is a stimulation and inhibition range as a function of dose, it is quite possible that the increased radiation load due to the Chernobyl disaster (also) caused biopositive effects for some living organisms.

2.7 SUMMARY

In this chapter the intention was to give a comprehensive picture of the levels of radioactivity in the different components of the biosphere, i.e., in the atmosphere–hydrosphere–soil–plant–animal–man chain process, of the factors influencing the concentration, distribution and accumulation of isotopes of natural and artificial origin. Special attention was paid to the migration of radioactive substances, to their transport from one component of the biological chain to the other, to the influence of the agricultural and food industrial activities on the level of contamination. The most important statements can be summarized as follows:

1. The level of radioactivity in a given area (the radiation load) is built up under the common influence of radioactive substances of natural and artificial origin. Natural radioactivity is mostly a function of the geological circumstances; however, the amount of radioactive substances produced by the cosmic radiation and coming from the atmosphere to the other components of the biosphere is not negligible. Among the isotopes of natural origin, based on their concentration, half-life, biological role and dangerousness, ^{14}C , ^{40}K and uranium (and its daughter elements) and radium (and its daughter elements) are the most important. Among the isotopes of artificial origin, which are emitted into the atmosphere as a consequence of nuclear weapon tests or from nuclear power plants or other nuclear facilities, ^{90}Sr and ^{137}Cs are of paramount importance. However, even the presence of ^3H and ^{85}Kr is not negligible considering the radioactive situation of the environment. Usually the importance of the other radionuclides (e.g. ^{89}Sr , ^{95}Zr , ^{106}Ru , ^{131}I , ^{134}Cs , ^{140}Ba , ^{239}Pu) is substantially smaller, if the area in question is quite distant from the epicentre of the nuclear weapon test.

2. Because of the continuous and dynamic interaction among the elements of the biosphere, radioactive substances (just like the circulation of the other elements) migrate continuously, being exchanged among the parts of the biological chain, and thus their concentration changes under the influence of numerous factors. The parameters determining the migration of radioactive substances interact in a highly complex manner, and thus the radioactivity of every member of the biological chain is formed as a consequence of the close interconnections with every other component, a really many-sided interaction. Thus for example, it is not only the soil that influences the radioactivity of the vegetation, but the latter also influences the soil, e.g. the water movement and water balance, which in turn modify the migration of the radioactive isotopes.

3. The radioactivity, or rather the radioactive contamination of a given area, is influenced by many factors. The most important ones discussed in this book, are as follows:

- geological circumstances;
- geographical location of the given area, the relief of the terrain, the hydrographic situation;
- climatic, meteorological features;
- mechanical structure, chemical composition of the soil, the applied agrotechnical and meliorative methods;
- species and vegetation period of the plants;
- species, age, feeding of animals;
- age, habitation, dietary customs of Man.

4. The ion uptake of ^{90}Sr from the soil through the root into the plant, is of paramount importance, as it causes long-lived radioactive contamination in the plants, and later in animal and human organisms. At the same time most of the ^{137}Cs comes not from the soil, but in a direct way, through the leaves into the plant tissues. The amount of radioactive substances taken up by the plants from the soil can be significantly decreased by the application of suitable agrotechnical methods, e.g. by liming. The level of radioactive contamination is in close correlation with the amount of precipitation, as the better part of the radioactivity coming from the atmosphere comes through wet deposition (rain-out, wash-out) on the surface.

5. The ^{90}Sr and ^{137}Cs activity in agricultural products and in vegetables is usually of the same order of magnitude; however, in animal somatic tissues the ^{137}Cs activity is usually dominant. The reason is that the animal (and human) organism discriminates very well against strontium, i.e., the $\text{Sr}:\text{Ca}$ proportion in the body is smaller by one order of magnitude than the value measurable in the fodder (or food). Thus the radioactive contamination of animal (and human) somatic tissues is determined mostly by the circumstances of feeding, or rather the dietary customs. The level of radioactive contamination is also influenced by age as at younger age the biological half-life is shorter and the rate of exchange is more intensive, so the organism reacts faster to the changes in contamination level in the ambient environment (here fodder, aquatic environment, food). ^{137}Cs , being mostly in the musculature, is exchanged faster than ^{90}Sr , which occurs mostly in the bones.

6. At present the level of radioactive contamination is low: natural radioactivity usually represents a significantly higher activity than the artificial contamination. The contamination level measured in Hungary is similar to that measured in other European countries. The maximum contamination due to the atmospheric nuclear weapon tests was between 1961 and 1964, when the radioactive contamination of the air-water-soil-plant-animal-man biological chain was higher by one order of magnitude than the values measured at the beginning of the 1980s. Unfortunately, in consequence of the nuclear power plant disaster in Chernobyl, the contamination level increased significantly in most European countries, far surpassing the values measured at the beginning of the 1960s. In spite of the fact that at present the degree of the radioactive contamination fortunately does not render it necessary to apply any special decontamination methods, it must be mentioned that the different stages of the processing of agricultural products and the processes of food preparation (e.g. extraction, washing, cooking) further decrease the level of contamination.

3. RADIOECOLOGICAL RESEARCH AND ORGANIZATIONS CARRYING OUT RADIOECOLOGICAL RESEARCH IN HUNGARY

I think it will perhaps be of some interest to give a short survey of the radioecological research programme in Hungary and to relate its objectives, detailing in what institutes and research laboratories these investigations are carried out, and to what other researches they are linked.

A short summary will be given of the research programme carried out in radioecological and related subjects in the Hungarian research institutions. Finally, the activities of the Radiometrical Monitoring Network of the Ministry of Agriculture and Food will be presented.

3.1 RADIOECOLOGICAL RESEARCH

In Hungary planned research activity is coordinated at national or ministerial level, within the framework of either the Hungarian Academy of Sciences or the National Committee for Technical Development with funding for the various research programmes depending on their nature and importance, and on the financial aid allocated to the different programmes. The major, long-term tasks, the more important priorities of research, are included in the National Plans for Long-Term Scientific Research, and the central development programmes. Covering shorter terms are the National Plans for Medium-term Research and Development, which specify the research and development objectives in a more precise wording. Research priorities considered of special importance by a ministry (e.g., Ministry of Health, Ministry of Agriculture and Food Industry) constitute the subject of the ministerial level research programme. (Topics of lesser importance, on which, however, sufficient emphasis is to be laid to warrant their treatment as programme in their own right, are elaborated within the frameworks of divisional, sectorial, or institutional research programmes.)

Of course, it is necessary that programmes at the different levels be connected to each other and be completed by research and development activity of an adaptational nature (licence, know-how); these should cover the major areas in a given field of science both at basic and applied research levels.

There are two programmes, or main directions of research, at the level of the

National Plan for Long-term Scientific Research which also includes research activities of radioecological nature, and these are as follows:

1. Protection of the Human Environment,
2. Widening the Selection of Foodstuffs, New Directions in Food Processing and Preservation.

The Ministry of Environmental Protection and Water Management is responsible for the programme entitled Protection of the Human Environment. The main subjects include protection of the purity of air and water, decontamination and utilization of wastes, protection of ecosystems and genetic resources, and environmental protection aspects of agricultural production within the framework of which researches of a radioecological nature are also performed.

The programme entitled "Widening the Selection of Foodstuffs, New Directions in Food Processing and Preservation" is managed by the Central Research Institute for Food Industry. Of the major topics of the programme the determination of the quality of foodstuffs is the one which is connected to the radioecological field. To comply with the more and more stringent specifications of the export market, the development and application of up-to-date methods of analysis are necessary, by means of which the residues of pesticides, additives and other contaminants dangerous to health present in the products may be determined with high accuracy.

In the National Plan for Medium-term Research and Development there are two programmes in which radioecological research is also carried out:

1. Research and Development Priorities for the Safe Operation of Nuclear Power Plants,
2. Research on Regional Environmental Protection of Lake Balaton.

The National Atomic Energy Committee is responsible for the co-ordination of the National Plans for the Medium-term Research and Development programme entitled "Research and Development Priorities for the Safe Operation of Nuclear Power Plants". The sub-programmes, in which radioecological research is also performed, are as follows: safety analysis for the forecasting of the quantity and distribution of radioactive contamination in emergency situations, the utilization and storage of wastes and burnt-out heating units of nuclear power plants, the analysis of the radiation conditions of normal operation and in emergency in nuclear power plants and their environment, and medical and biological research required for the safe operation of nuclear power plants and for safeguarding against radiation accidents.

The Ministry of Environmental Protection and Water Management is responsible for the programme entitled Research on Regional Environmental Protection of Lake Balaton. Research of a radioecological nature is carried out in the following sub-programmes:

- analysis of the biological processes in the lake;
- the scientific foundation and research tasks of the control of water-quality;
- methods of prevention and neutralization of materials and effects harmful to human health.

One of the major medium-term directions of research under the Hungarian Academy of Sciences is entitled "The Protection of Man and His Natural Environment

(Biosphere)". Research of a radioecological nature is carried out in the following main directions of the research programme: comparative research on terrestrial ecosystems, research on cultivated ecosystems, or those affected by cultivation, and their contaminations, and research on aquatic ecosystems.

Within the framework of the Ministry of Agriculture and Food, researches of an ecological nature are also conducted in connection with the development of analytical methods under the titles "Researches for the Foundation of the Comprehensive Development of Food Industry", and "Quality Improvement in the Production of Meat-based Foodstuffs", both connected with the long-term strategic programmes.

The programme supervised by the Ministry of Industry, entitled Environmental Protection, is connected to the field of radioecology. In this strategic programme the neutralization and safe storage of dangerous waste materials is of primary importance.

Several subjects of a radioecological nature are included in the main research direction of the Ministry of Health entitled Medical-Biological Research Related to the Application of Ionizing Radiation and Radioisotopes. The major subjects related to the above are as follows: cytopathological analysis of radiation damage, survey of the changes in environmental radioactivity and radiation load on the population after the Paks nuclear power plant went into operation; migration of the more important radionuclides in the terrestrial and aquatic food chain; radioecological analysis of the Hungarian section of the Danube; and radiation load of the employees of the nuclear power plant.

There is a ministerial research programme also under the Ministry of Environmental Protection and Water Management. Of the major research topics in this research programme hopefully the results obtained in the following fields will also pertain to radioecology: protection of air purity; the prevention of waste-production; the utilization and neutralization of wastes; and faunistic and floristic research studies of ecosystems.

In addition to the national or ministerial research sub-programmes listed above, research activities are also pursued in several other, lower-level research programmes which are either directly connected to radioecological subjects, or the results of which may also be utilized in radioecology. Thus for example, under the auspices of the Ministry of Agriculture and Food, research related to radioecological subjects is conducted within the framework of the programmes entitled Agrochemical Researches and Radiation Techniques in Agriculture and Food Industry. In the first programme the intake of minerals by plants, ion uptake through the roots and leaves and interdisciplinary research of the biochemical role of the micro-elements are the major subjects, best utilized in the subfield of radioecology. One of the major subjects of the second programme is the influence of radioactive contamination on crop development and the important quality indicators of certain plants.

Naturally a wide scope of research work is being carried out on the subject of radioecology in several countries of the world, and a number of international research programmes have been set up. Thus, for example, Hungary is also a participant in the research programme entitled "The Problems of Radioecology of the River Danube", initiated for the survey of radioactive contamination of the Danube and its environ-

ment and co-ordinated by the International Atomic Energy Agency. Within the framework of this programme, extensive research is carried out in the National Research Institute for Radiobiology and Radiation Hygiene.

The investigation on the radiation safety of the River Danube and its environment became necessary because of the nuclear power development programmes of the countries situated along the Danube, especially nuclear power plant construction and development in Hungary. The aim of the research is to follow the migration of the radionuclides released into the hydrosphere and to analyse the radiation-hygiene risks brought about by the utilization of the river's resources. It must be mentioned that coordinated research on the radioactive contamination of the Danube was also pursued within the framework of the Council for Mutual Economic Assistance.

3.2 INSTITUTES CARRYING OUT RESEARCH OF A RADIOECOLOGICAL NATURE

Research and investigations of a radioecological nature are carried out in a number of Hungarian institutes. Instead of listing the different institutions and their research programmes, it is more practicable to group the institutes according to the nature of their research or monitoring tasks. Three ministries, the Ministry of Health, the Ministry of Agriculture and Food, and the Ministry of Environmental Protection and Water Management, and their several institutes, research laboratories, and field stations take part in the environmental radiation monitoring work in Hungary. The structure of the environmental radiation protection control system directed by the authorities is shown in Figure 33.

a) Atmosphere

The following sub-section itemizes institutes dealing with problems connected with the radioactivity of the atmosphere.

Under the auspices of the National Meteorological Service as a central organ, data on the level of fall-out and natural radioactivity are collected by the local meteorological stations coordinated by the Central Institute for Atmospheric Physics as a base institute.

Radioactive aerosol and fall-out measurements and medical researches are carried out in the National Research Institute for Radiobiology and Radiation Hygiene.

The radioactive contamination of the atmosphere is also monitored by research laboratories belonging to the Hungarian Academy of Sciences, i.e., by the Department for Radiation Protection of the Central Research Institute for Physics and the Institute for Nuclear Research in Debrecen.

The analysis of the air-contaminating emissions of the nuclear reactors is the task of the Nuclear Training Reactor of the Technical University of Budapest and the Nuclear Power Plant Company of Paks.

Finally, the activity of the Ministry of Environmental Protection and Water Management should also be mentioned, which, through the Institute for Environmental

Protection, plays the role of the base institute in the protection of air purity and directs and coordinates the work of the local offices in the counties.

b) Hydrosphere

For the official control system of the radioactive contamination of waters and aquatic ecosystems (aquatic plants, algae, mud, fish), the central organ is the Ministry for Environmental Protection and Water Management and the base institute is the Scientific Research Institute for Water Management. Measurements are made by county directorates of environmental protection and water regulation.

On behalf of the Ministry of Health, the National Research Institute for Radiobiology and Radiation Hygiene is the base institute.

Radioactive emissions of nuclear power plants causing the contamination of the aquatic environment are investigated by the Nuclear Training Reactor of the Technical University of Budapest and by the Nuclear Power Plant Company of Paks.

Measurements of radioactivity in water are also made by the Radiometrical and Monitoring Network of the Ministry of Agriculture and Food with the coordination of the Veterinary and Food Control Service as a base institution.

c) Soil

Analyses of the natural and artificial radioisotope content of the soil are carried out within the framework of the Ministry of Agriculture and Food by the Network of the Veterinary and Food Control Stations in the various counties, with the base organ being the Veterinary and Food Control Service.

Analysis of the radioactive contamination of the soil within the jurisdiction of the Ministry of Health is dealt with by the National Research Institute for Radiobiology and Radiation Hygiene.

Of the institutes of the Hungarian Academy of Sciences, the Research Institute for Soil Science and Agrochemistry and the Institute for Nuclear Research are engaged in research on the problems of the radioactivity of the soil.

d) Flora and fauna

Analysis of the radioactivity of flora, fauna and agricultural products and foodstuffs is primarily the task of the Ministry of Agriculture and Food. Measurements are taken by the veterinary and food control stations in the counties and by the research institutes of the different industrial sectors, with the direction and coordination of the Veterinary and Food Control Service as the base organ.

The radioactive isotope uptake of plants and questions of a radioecological nature in plant physiology are investigated in the isotope laboratories and irradiation gardens of the several universities for agricultural sciences and in the Research Institute for Irrigation. The animal physiological researches in connection with radioecology are allocated to the Stockbreeding and Feeding Research Centre and to the Veterinary University.

In addition to the Ministry of Agriculture and Food, the Ministry of Health is also engaged in the analysis of the radioactive contamination of foodstuffs, the analyses being performed partly by the public health and epidemiological stations, and partly by the National Institute for Nutrition and Dietetics.

e) Man

Analysis of the radiation load affecting the human organism from different sources,

analysis of the radioactive isotopes incorporated into the human organism, direction of the radiation control in working places and the research activity in connection with these are primarily the task of the Ministry of Health, the base institution of which is the National Research Institute for Radiobiology and Radiation Hygiene. Measurement and research of this nature are carried out in the National Oncological Institute, the Institute for Biophysics of the Medical University of Pécs, and the Radiological Clinic of the Semmelweis Medical University.

Dosimetric research is pursued in several laboratories of the Hungarian Academy of Sciences, such as the Central Research Institute for Physics, the Institute for Nuclear Research, and the Isotope Research Institute.

Questions as to the population-level radiation exposure to natural radioactive isotopes are also investigated by some other institutions, e.g. the Health Service of the Mining Company of Mecsek.

f) Radioactive wastes

Issues related to the treatment and deposition of the radioactive wastes produced in nuclear facilities, and the development of monitoring methods for radiation protection are dealt with, on the part of the Ministry of Health, by the Public Health and Epidemiological Station of Budapest, and of the research institutes of the Hungarian Academy of Sciences, by the Isotope Research Institute and the Central Research Institute for Physics.

g) Measurement technique

The majority of the research related to the measurement technique of radioactive isotopes and the development of new radiation protection equipment is carried out in several research laboratories, the Central Research Institute for Physics, the Isotope Research Institute and the Institute for Nuclear Research of the Hungarian Academy of Sciences, and in the jurisdiction of the Ministry of Health, in the National Research Institute for Radiobiology and Radiation Hygiene, the Institute for Biophysics of the Semmelweis Medical University and the National Oncological Institute. In the field directed by the Ministry of Agriculture and Food, methodological research is pursued by the Veterinary and Food Control Service.

Mention should be made of the research and development activity of the Gamma Works (development of measuring systems), and the work done by the National Office for Metrology (standardization, preparation of reference standards) and by the National Office for Standardization.

h) Other issues

In addition to the research laboratories listed previously, there are several institutes and university departments in Hungary, whose research is closely connected to radioecology. This is basically in two directions, one in nuclear measurement technique and the biology of radiation effects, and the other on the subject of inactive microelements and toxicology. The first direction requires no special explanation, since it is obvious that the application and development of the systems measuring and evaluating ionizing radiation (detectors, analysers, spectrometers, computerized spectrum evaluation, etc.) of the radioanalytical and isotope technical processes, and studies of the radiobiological and dosimetric methods are closely related to the science of radioecology. Research and analyses of this kind are pursued by the departments

of radiochemistry, analytical chemistry, nuclear physics, or radiology of the medical and technical universities, and the universities of sciences. The other direction is the field of inactive microelements. It has already been mentioned that a radioactive isotope is practically not different from the inactive isotope of the same element as regards its chemical character, i.e., chemically the stable and the active isotope behave in an essentially identical way. Consequently, the information available on the migration, transport, and accumulation of the inactive materials (primarily microelements) within the biological chain process can be directly referred to the radioactive isotopes of these elements, that is, they can also be used in the field of radioecology. There is also a close link with the toxicological and radioecological knowledge and research related to the heavy metal microelements. To this second direction join for example the analyses and research conducted by the departments of chemistry, plant physiology, or animal physiology of the agricultural universities, and the research and development activities carried out within the framework of the Plant Protection and Agrochemical Network.

Finally it must be mentioned that the operational and official radiation protection control data of the environment of the Nuclear Power Plant Company of Paks are transmitted to a so-called data bank. Thus, in case of emergency, for example, increasing contamination, reactor accident, or nuclear war, the National Headquarters of Civil Defence have sound information at their disposal to take the required precautionary measures.

3.3 RADIOLOGICAL AND MONITORING NETWORK OF THE MINISTRY OF AGRICULTURE AND FOOD

The Radiological and Monitoring Network of the Ministry of Agriculture and Food operates within the framework of the National Radiation Monitoring System, and its primary task is to analyse the radioactivity of environmental elements pertaining to the scope of authority of the ministry. The basic task of the network is to analyse regularly and continuously radioactive contamination of the agricultural products and foodstuffs, and to supply information on the results of laboratory measurements. In addition to this major function, it carries out investigations, even if occasionally or on a smaller number of samples, for the survey of radioactive contamination of the soil, surface waters, and certain wild plants and animals.

The network carries out investigations throughout the country. Sampling locations are shown in Figure 34. The regular monitoring work was started in 1960. For the last ten years the 30 km environment of the Nuclear Power Plant of Paks has been included in these investigations with special emphasis (see Figure 35). Up to the date the nuclear power plant went into operation, a survey of the background radiation level was carried out in the Paks region. The measurements supplied information on the level of the actual contamination and the fluctuations of the contamination level,

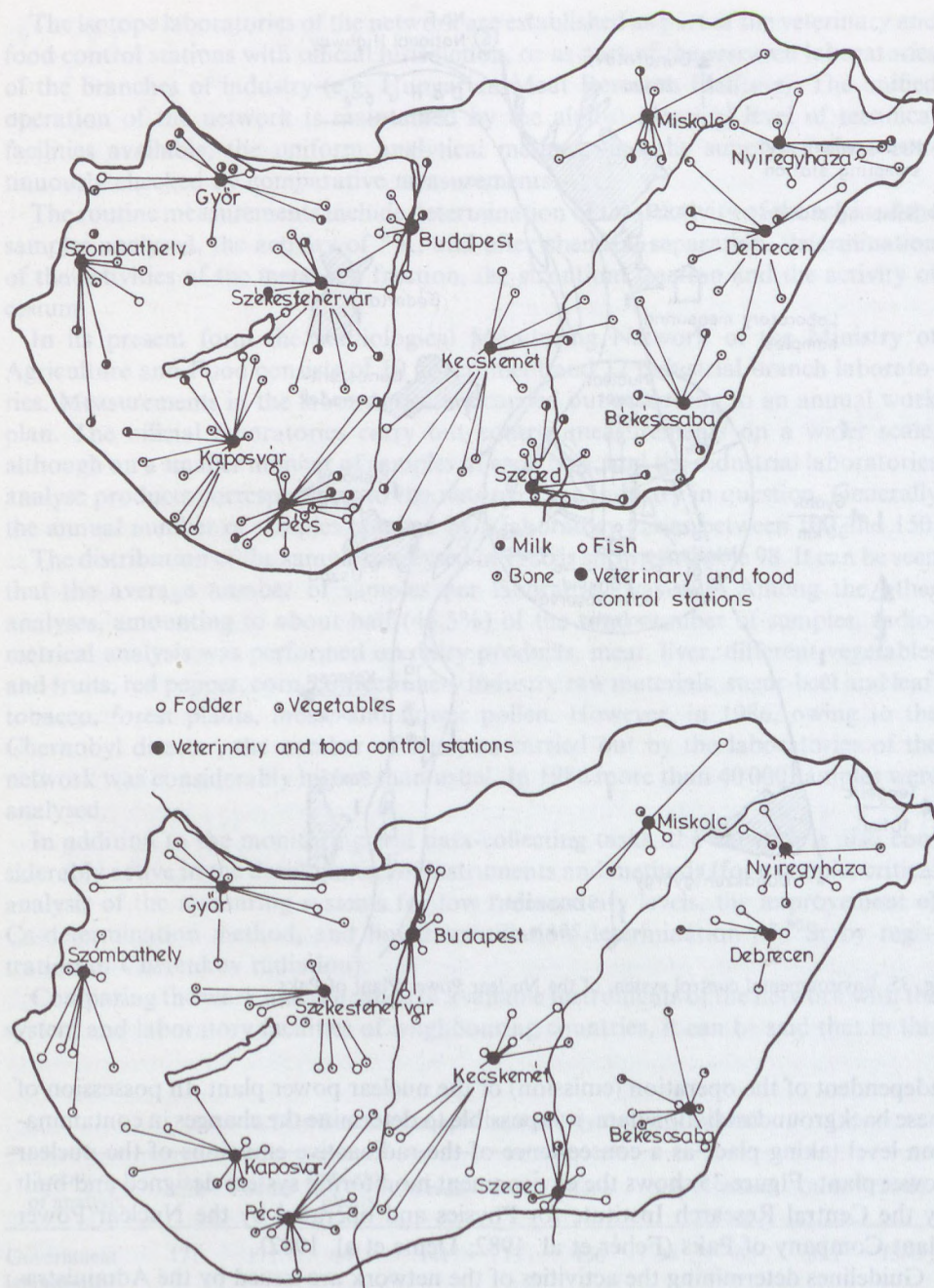


Fig. 34. Sampling areas for measuring radioactive contamination

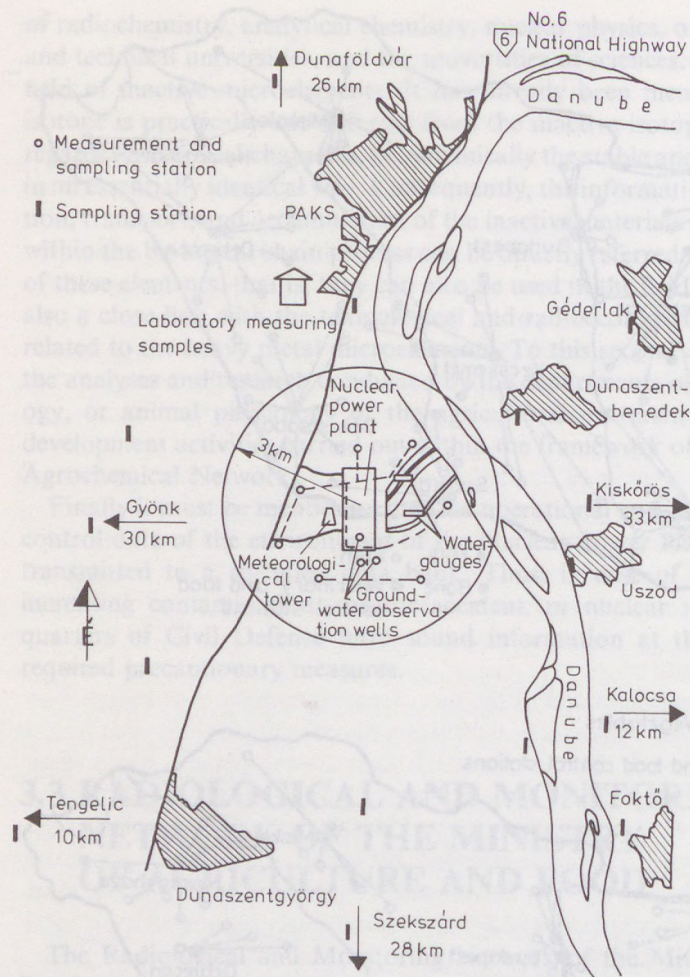


Fig. 35. Environmental control system of the Nuclear Power Plant of Paks

independent of the operation (emission) of the nuclear power plant. In possession of these background radiation data, it is possible to determine the changes in contamination level taking place as a consequence of the radioactive emissions of the nuclear power plant. Figure 35 shows the environment monitoring system designed and built by the Central Research Institute for Physics and operated by the Nuclear Power Plant Company of Paks (Fehér et al. 1982; Deme et al. 1982).

Guidelines determining the activities of the network are issued by the Administrative and Legal Department of the Ministry of Agriculture and Food, in agreement with the Veterinary and Food Control Department and with the approval of the National Headquarters of Civil Defence. Operational control is the responsibility of the Veterinary and Food Control Service of the Ministry of Agriculture and Food.

The isotope laboratories of the network are established as part of the veterinary and food control stations with official jurisdiction, or as part of the research laboratories of the branches of industry (e.g. Hungarian Meat Research Institute). The unified operation of the network is maintained by the almost identical level of technical facilities available, the uniform analytical method, and the supply of data continuously checked by comparative measurements.

The routine measurements include determination of total activity of the ashes of the samples analysed, the activity of ^{40}K , and after chemical separation, determination of the activities of the metal-ion fraction, the strontium fraction and the activity of cesium.

In its present form the Radiological Monitoring Network of the Ministry of Agriculture and Food consists of 10 government and 12 industrial branch laboratories. Measurements in the laboratories are carried out according to an annual work plan. The official laboratories carry out control measurements on a wider scale, although on a smaller number of samples of each type, and the industrial laboratories analyse products corresponding to the nature of the industry in question. Generally the annual number of samples handled by a laboratory varies between 100 and 150.

The distribution of the samples analysed in 1980 is shown in Table 98. It can be seen that the average number of samples per laboratory was 122. Among the other analyses, amounting to about half (46.5%) of the total number of samples, radio-metrical analysis was performed on dairy products, meat, liver, different vegetables and fruits, red pepper, corn, confectionery industry raw materials, sugar-beet and leaf, tobacco, forest plants, moss, and flower pollen. However, in 1986, owing to the Chernobyl disaster, the number of analyses carried out by the laboratories of the network was considerably higher than usual. In 1986 more than 40 000 samples were analysed.

In addition to the monitoring and data-collecting tasks, the network is also considerably active in the development of instruments and methods (for example, critical analysis of the measuring systems for low radioactivity levels, the improvement of Ca-determination method, and liquid scintillation determination of ^{90}Sr by registration of Cherenkov radiation).

Comparing the work and the range of available instruments of the network with the system and laboratory facilities of neighbouring countries, it can be said that in this

Table 98 Distribution of samples analysed by the RCN in 1980

Location of analysis	Milk	Fodder	Indicator plants	Weeds	Fish	Bone	Soil	Water	Other	Total
Government laboratories	171	171	341	144	73	156	94	65	335	1550
Industrial laboratories	36	0	0	0	0	139	41	0	908	1124
Total	207	171	341	144	73	295	135	65	1243	2674

respect Hungary does not play a subordinate role in the field. The laboratory instruments and investigation processes which have been introduced and standardized so far are in compliance with the current European standards.

In Hungary a large number of food and environmental samples are taken regularly and systematically and analysed in order to monitor their radioactive contamination. Only regular contamination control analyses can safeguard against any possible new nuclear weapon test, radioactive emission or contamination resulting from special meteorological conditions or from the peaceful use of nuclear energy, being undetected. The necessary measures can thus be taken without delay depending on the nature and concentration of the contamination.

3.4 SUMMARY

In this chapter an attempt has been made to give an overview of the radioecological research and research programmes in Hungary, and to provide a survey of the institutions engaged in research of a radioecological nature and in radiometric analysis, and of the activities of the Radiological Monitoring Network of the Ministry of Agriculture and Food. In a brief summary the following statements can be made:

1. Since radioecology is specifically an interdisciplinary field of science, there are a large number of research and development programmes in the framework of which research of a radioecological nature is also carried out. As far as their significance is concerned, the programme entitled "Research and Development Priorities for the Safe Operation of Nuclear Power Plants", marked off by the National Medium-Term Research and Development Plan and coordinated by the National Nuclear Energy Committee, and the major research initiative of the Ministry of Health entitled "Biomedical Researches Related to the Application of Ionizing Radiation and Radioisotopes" are the ones of particular importance.

2. The task of monitoring the radioactivity of the environment is basically fulfilled by three ministries, the Ministry of Health, the Ministry of Agriculture and Food and the Ministry of Environmental Protection and Water Management and their institutes, research laboratories and field stations, where a considerable proportion of the research work of a radioecological nature is performed. Significant research work is carried out in similar fields also in the different institutes of the Hungarian Academy of Sciences (Central Research Institute for Physics, Nuclear Research Institute, Isotope Research Institute, Research Institute for Soil Science and Agrochemistry); and important work is also carried out in some sub-fields by other institutes, companies and research laboratories (e.g. Nuclear Teaching Reactor of the Technical University of Budapest, Nuclear Power Plant Company of Paks, Health Service of the Mining Company of Mecsek).

3. The basic function of the Radiological Monitoring Network of the Ministry of Agriculture and Food is the regular and continuous monitoring of radioactive contamination of agricultural products and foodstuffs. The operational work of the network is performed by the Veterinary and Food Control Service.

4. WHAT HAS THE FUTURE IN STORE FOR US?

The primary aim of this chapter is not so much to convey radioecological information to the reader, but rather to draw attention to the subject, to emphasize the necessity of a sense of responsibility for our common future, and to present the wide range of possibilities of the application of nuclear technique. This chapter is divided into three short sub-sections. In the first part, questions related to military engineering, nuclear weapons and nuclear warfare are analysed and the responsibility of ecologists is pointed out. An adequate answer is sought in the second part for the question: are nuclear power plants dangerous to the environment? The third part was written with the aim of giving a short survey of those fields of nuclear technique which can successfully be applied in the different fields of agriculture and the food industry.

4.1 MILITARY ENGINEERING, NUCLEAR WEAPONS, NUCLEAR WARFARE, AND THE RESPONSIBILITY OF ECOLOGISTS

It is a commonplace statement that during the past few decades a revolutionary change has taken place in military engineering; the destructive, annihilating power of the RBC weapons in many times that of traditional military equipment. The strategic rocket nuclear weapons (the so-called intercontinental ballistic missiles equipped with a nuclear warhead), owing to their long effective range, manoeuvrability and speed (in 30 minutes they can reach a target at a distance of 15 000 km) can strike a crushing blow on military, industrial and civilian targets. In addition to the intercontinental rockets, a strategic air force equipped with long-distance bombers of long effective range and a nuclear-powered submarine fleet developed for carrying and launching medium-range rockets are also important parts of the strategic attacking capacity.

The early atomic bombs detonated over Hiroshima (uranium) and over Nagasaki (plutonium) were the so-called single-phase (fission, or F-) bombs, operating on the principle of the chain reaction. In these the chain reaction, besides the neutron source, could be started by uniting with a detonator, the two nuclear charges, both of a mass which in itself is smaller than the critical one. Very large amounts of heat develop as a consequence of the chain reaction, and the temperature of several million degrees

Celsius renders possible the fusion, and thus the development of the two-phase atomic bomb (FF-bomb, fission + fusion). Here, the detonator is essentially a single-phase atomic bomb which is surrounded by the agent of the H-bomb (deuterium, tritium or lithiumdeuteride). Since in the explosion of the H-bomb a great amount of high-energy neutrons are generated, this makes it possible to use ^{238}U also for nuclear fission. If a two-phase bomb is surrounded by another jacket made of natural uranium in which nuclear fission is produced, then we can speak of a so-called three-phase (FFF-, i.e. fission + fusion + fission) nuclear weapon.

It is mentioned here only for the sake of comparison that, expressed in TNT-equivalents, the first atomic bombs were about of 20 kt (recently the destructive force of the Hiroshima and Nagasaki atomic bombs was assessed to have been 12.5 kt and 22 kt, respectively), and the upper limit of the active force of single-phase nuclear weapons is about 300 kt. For the two-phase nuclear weapons this value can be several Mt, while for the three-phase ones it can reach several hundred Mt. To obtain a picture of the destructive power of nuclear weapons, it should be mentioned here that, for example, the total explosive compound of the bombs dropped in the air raids by the Allies during the Second World War amounted "only" to 1.4 Mt, the equivalent of a single, medium-sized, two-phase bomb. Thus it may be said without any exaggeration that in a nuclear war the whole of mankind could be exterminated and human civilization could be wiped from our planet.

The destructive power of nuclear weapons is exerted in the form of heat radiation, shock wave and ionizing radiation. Since the effect of heat radiation and ionizing radiation both decrease quadratically as a function of distance and the force of the shock wave decreases cubically, so a nuclear weapon of 1 Mt destructive power can practically be replaced by three 100 kt ones. Therefore, in order to increase the destructive power of a single rocket, rockets supplied with several warheads have been constructed.

Military engineering is in constant development, and the improvement and modernization of nuclear weapons is going on. For example, a neutron bomb has been constructed which has a smaller destructive effect on buildings; however, it can produce very intensive neutron radiation, consequently greatly enhanced radioactivity. It should also be mentioned that, owing to the increased purity of fissile materials, and to improvement in the construction, design and production of new, artificial fissile materials (for example, californium having a critical mass of about 1.5 g), theoretically it is possible to construct nuclear charges which can be used in firearms.

The effect of the application of nuclear weapons and of nuclear weapon tests (threatening to destroy the whole mankind) lies in the contamination of the environment by radioactive materials. These fission products as we have been seen in detail in the preceding chapters, enter the plant-animal-man biological chain from the air, water, or soil, and accumulate in living organisms causing radiation damage. That is precisely the fact that should be the reason why peace-loving people in the world, especially ecologists who possess knowledge of the biological chain process and the ecological factors and are thus able to realize the imminent danger, should raise their protest against the application of nuclear weapons and the continuation of nuclear weapon tests.

The motto of the environmental conservation strategy of the world, "The Earth is not inherited from our parents, but it is borrowed from our children" (UN Environment Programme), is well known. Since we are aware of the dangers arising as a consequence of radioactive contamination of the biosphere, we ecologists have to promote the idea and raise our voices in all possible forums that nuclear weapons must disappear once and for all from the arsenal of the world.

4.2 ARE NUCLEAR POWER PLANTS DANGEROUS TO THE ENVIRONMENT?

The above question is justified, especially so after Chernobyl. Of course, Chernobyl was not the first reactor accident of all, but it was the first to have really serious consequences. Only three earlier reactor accidents resulted in significant releases of radionuclides: the first at Chelyabinsk (USSR) in 1957, the second at Windscale (United Kingdom) in October, 1957, and the third at Three Mile Island (United States) in March, 1979 (UNSCEAR 1988). While it is very difficult to estimate the fraction of the Windscale radionuclide core inventory released to the atmosphere, it has been estimated that the accident released twice the amount of noble gases released at Chernobyl, but 2000 times less ^{131}I and ^{137}Cs . The Three Mile Island accident released approximately 2% as much noble gases and 0.00002% as much ^{131}I as the Chernobyl accident did. The contaminating effect of the Chelyabinsk disaster was approximately 1 order of magnitude smaller than that of Chernobyl (Gudkov 1990).

It is generally known that nuclear reactors and nuclear facilities, even during their normal operation, release a certain, and occasionally not negligible, amount of radioactive material to the environment, that is, they pollute the biosphere with radioactive materials. The tendency in the energy supply structure of our planet appears to be self-evident: the application of nuclear energy will grow and gradually replace energy obtained from fossil fuel (Figure 36) (Jones 1988). This undoubtedly will be made possible only by the construction and operation of further nuclear power plants, which will be accompanied by an ever-increasing amount of radioactive emission. Consequently, more and more radioactive material will be released into the environment; the amount of radioactive contamination will constantly be on the increase. Is that really what can be expected?

So far numerous data have been published on the environment-polluting emissions of the reactors, and there is a considerable amount of data available on the radioactive contaminating effect of nuclear power plants and nuclear facilities in the course of their ordinary operation (e.g. Nelson et al. 1972; Gusev 1976).

There are a large number of publications analysing the issue of technological safety of nuclear power stations, and the inherent dangers at the population level. (For example, both the so-called Rasmussen-report (U.S. Nuclear Regulatory Commission 1975) and the so-called Birkhofer-study (Deutsche Risikostudie Kernkraftwerke 1980) are widely known.) In judging risk in the operation of a nuclear power plant the

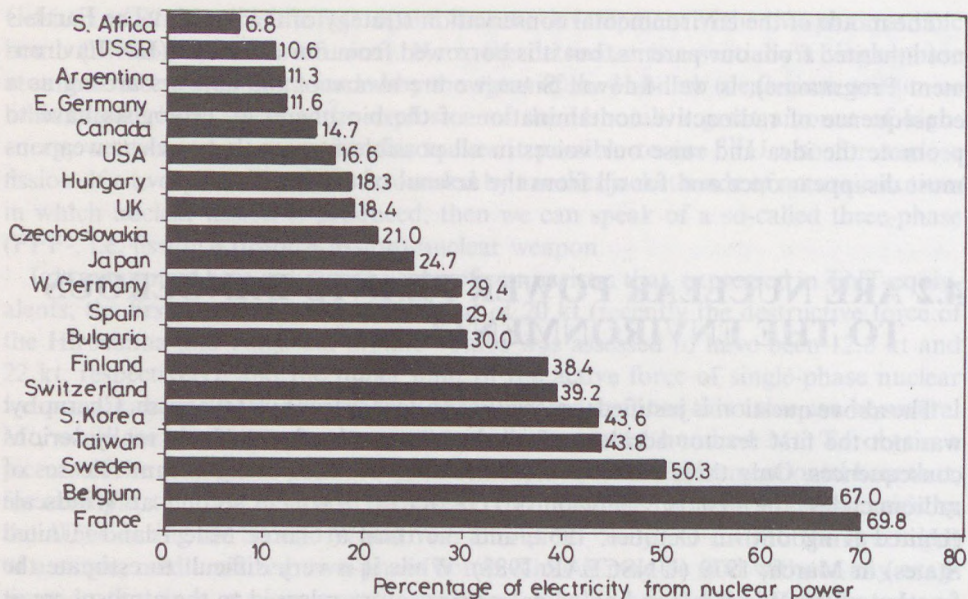


Fig. 36. The percentage of electricity generated from nuclear power plants in various countries throughout the world in 1986

consequences of accidents and the probability of their occurrence should be taken into account simultaneously. Risk assessment has its elaborate mathematical methodology based on probability calculations.

In the case of nuclear power plants and nuclear facilities, in principle the normal, operational emission is distinguished from the radioactive release taking place during a breakdown. When the site of the prospective plant is selected and it is put into operation, it is a generally accepted prerequisite that the biological equilibrium of the environment should not be endangered by the polluting effect of these facilities. At the same time there are nuclear power plants also which emit some pollution into the environment; however, the costs of construction and operation still remain within the margin of competitiveness. In any case a certain compromise is necessary: theoretically it is possible to build a nuclear power plant of any predetermined low level of emission of radioactive materials; however, it is always the natural background radiation level that must be taken into account and to this must the level of contamination be compared.

The failure or breakdown emission of nuclear power plants can cause serious contamination in the environment of the reactor. Therefore nuclear power plants must be designed, constructed and operated with a high degree of safety. The main point is that a highly reliable safety system must be installed (for example, for stopping the reactor immediately in the event of failures, and for minimizing their consequences). In short, a safety system is required, which on the one hand can guarantee that the possibility of failures leading to breakdowns is very small, and on the other hand, can provide acceptable margins of risk even in the case of possible

unforeseen failures or breakdowns. Of course, the possibility of a breakdown cannot be excluded even with the most consistent observation of the most severe safety measures. For example, in 1979 in Harrisburg (USA), there was a serious breakdown in the Three Mile Island nuclear power plant (McDermott 1982), although it is true that in spite of the breakdown the contaminating effect on the environment was relatively small.

In recent years several data have been published in the UNSCEAR-, ICRP-, and NRPB-reports on the frequency of fatal malignant illnesses, leukaemia, carcinomata, among persons whose body, or part of their body, had been exposed to radiation. According to these reports, the radiation exposure of one million people to a dose of 0.01 Gy, that is a collective dose of $1 \cdot 10^4$ persons $\cdot$ Gy, results on the average in 120 fatal malignant tumorous illnesses. In addition to these, also about 100 such tumorous illnesses can be expected which by now can already be treated successfully (e.g. tumours of the thyroid or the breast).

The collective radiation load of the population of Hungary is $3 \cdot 10^2$ persons $\cdot$ Gy/year, an average individual radiation load of $2.5 \cdot 10^{-5}$ Gy/year. Calculated for one million persons, a maximum of 0.3 fatal and 0.25 medicable malignant tumours per year are expected. Relative to the total Hungarian population assumed to be 10–12 million at the turn of the millennium, this would mean 3–4 fatal and 3 medicable tumours per year. The genetically significant individual radiation load of the one million group of the population is a dose of 10^{-5} Gy/year, as a consequence of the nuclear power energy programme, which in the first two generations may cause 0.1 significant genetic damage per year, i.e., one case annually in the whole population.

The health risks of the Hungarian nuclear power plant construction programme are shown in Table 99. In the table the expectable health risks are calculated for a population of 12 million, based on 1 GWe and 12 GWe electric energy output obtained from a nuclear power plant. From the data of the table the conclusion can be drawn that at the turn of the millennium the supply of the additional electric power requirement of the Hungarian population from nuclear energy could cause 3–4 fatal and 3–4 medically treatable illnesses, 5–6 fatal, and 700–1 000 non-fatal accidents or occupational diseases, and 1–2 serious genetic damages every year.

Only to illustrate the orders of magnitude and for the sake of comparison, it is mentioned here that the annual number of casualties in Hungary due to traffic, industrial or home accidents is about 5 000 persons, which, referred to a population of one million persons will make the number of annual casualties to about 500. Deaths in the population as a consequence of malignant tumours may be assessed as about 2,400/million persons. As for the genetic damages, it must be mentioned that about 6% of infants have some kind of genetic disorder, and about 1% of these are of a dominant hereditary character. The number of these latter genetic damages by the millennium may exceed 2 000 per year.

I think that nowadays the dangers of nuclear power plants are brought up unjustifiably too often. On the whole the truth is that with the present technology of nuclear power plants, assuming an accident-free operation, the expectable radiation load is only a small fraction of the natural background radiation. Of course it cannot be

Table 99 Health risks of the nuclear power plant construction programme

Source and type of possible damage	Incidence in a population of 12 million	
	1 GW _e year	12 GW _e years
Fatal radiation		
total body irradiation	0.3	3.6
alpha-irradiation of the respiratory tract	0.002–0.004	0.025–0.05
irradiation of individual organs	0.005	0.06
total	0.307–0.309	3.69–3.71
Medicably treatable radiation damage		
total body irradiation	0.25	3.0
irradiation of individual organs	0.05	0.6
total	0.30	3.6
Genetic radiation damage		
in the first two generations	0.1	1.2
in all other generations	0.1	1.2
total	0.2	2.4
Fatal accidents		
uranium mine	0.03	0.36
ore mill	0.01	0.12
power plant construction	0.3–0.4	3.6–4.6
power plant operation	0.05	0.6
transportations	0.01	0.12
total	0.40–0.50	4.80–6.00
Non-fatal damages		
accident	55–65	660–780
occupational diseases	5–25	60–300
total	60–90	720–1080

stated that nuclear power stations and nuclear facilities have no contaminating effect (the danger lies primarily in ^{85}Kr and ^{131}I), but the global contamination level is very low. In practice this small amount of radioactive emission will also be eliminated in all likelihood by the beginning or the middle of the 21st century, by putting fusion power plants into operation, which from the point of view of the relative biological hazards are 10^6 – 10^7 times more advantageous than the present fission power plants.

The high-activity waste produced in nuclear power plants, nuclear facilities and isotope laboratories must be stored in such a way that they should not contaminate the environment. This question has already been dealt with when discussing the subject of the radioactive contamination of the soil. In addition to that we mention here only one further point. The newest, and from the point of view of possible further contamination also the safest way for storing radioactive waste is casting it, after suitable pre-treatment, into glass (vitrification). The chemical characteristics of glass are such that under natural circumstances practically no radioactive material can be released from it. The radioactive glass cast into the appropriate form in moulds can then be stored safely, either in the depth of the ocean or underground, and there is

no need to worry about the radioactive waste material being released and polluting the environment. The disadvantage of this process is that it is rather costly.

Part of the truth is that apart from radioactive emissions, the so-called thermal contamination of nuclear power plants is also a factor that cannot be disregarded. Because of their considerable thermal output, nuclear power plants must be sited along large rivers which provide cooling water. However, the experience of operating nuclear power plants for about four decades appears to be sufficient so that the thermal effects of new nuclear power plants to be constructed in the future can be adequately assessed, and putting the new plants into operation will not significantly upset the equilibria of the aquatic ecosystems.

Finally, another fact which has already been mentioned before, it is not only nuclear power plants that produce radioactive emission. The fly ash of coal-burning power plants is also very rich in different kinds of natural radioactive isotopes, that is radioactive contamination is also caused by energy production utilization of the traditional methods based on fossil fuel. Of course, from the point of view of environmental protection, in the case of thermal (fossil fuel) power stations the air-polluting compounds (SO_2 , CO, NO, etc.), and not radioactive emission, stand at the focus of attention.

4.3 NUCLEAR TECHNIQUE — A BLESSING OR A CURSE?

This is perhaps a somewhat too resounding title; however, it calls attention to a problem of primary importance for the future of mankind. It is a fact that by nuclear energy one can both destroy and build; with its help, both paradise and hell can equally be created. Is it then a blessing or a curse? Unfortunately no definite answer can be given as yet to this question. The sad examples of Hiroshima, Nagasaki or Chernobyl still haunt us and the danger of nuclear war has not passed either. For several decades it will be possible to detect contamination from the nuclear weapon tests or reactor accidents, and there are numerous localities, e.g. the Bikini Islands and the immediate vicinity of Chernobyl, where the area is contaminated even now to such an extent that staying there could result in a radiation load higher than the acceptable risk level.

This is one side of the question. And what about the other side? There are a great number of arguments that can be enumerated here. For instance, the energy obtained from nuclear power plants plays an ever-increasing role meeting the energy requirement of the world. Radioactive isotopes are widely used in several fields of scientific research, medical diagnosis and therapy, industrial process control, and the analytical application of the nuclear measurements techniques. Irradiation techniques can also be extensively used in agriculture and the food industry. This is a long list of advantageous applications. It must be mentioned, of course, that up to now both in the USA and in the Soviet Union several underground nuclear detonations with civilian objectives were carried out (Schuricht 1974; Zhuravlev, Katsapov 1976;

Morohov et al. 1977). The aims of these controlled explosions, among others, include the construction of large hydrotechnical establishments and canals, extinguishing gas bursts, oil exploitation, the construction of underground storage spaces, etc. Table 100 shows the dimensions of a cavity that can be formed in a layer of natural salt by underground nuclear detonations of different explosive power (Kruglov 1976).

On the basis of the above, it probably will not seem too difficult to answer the question whether it is a blessing or a curse. It is a blessing, if the nuclear technique

Table 100 Characteristics of underground nuclear weapon explosions

Power of explosion (kt)	Depth (m)	Diameter of cavity produced (m)
1.1	161	13.0
3.4	365	18.7
5.3	828	17.4
25.0	600	32.0

is used for the sake of mankind, and it is a curse if it is used against Man. As far as the direction of this activity is concerned, it is ourselves who can make the choice, the responsibility rests with us. And if we do not succeed? Even then it is not the nuclear technique which is to be blamed, but rather the people who turned nuclear power against the human race. Of course it is impossible to predict whether there will be a single survivor to be accused of causing such a catastrophe.

In the following, after this somewhat subjective allusion which may even appear somewhat too pessimistic, let us see how and where nuclear technique is or can be applied in the practice of agriculture and in the food industry or in scientific research. Since this subject is connected to radioecology only very indirectly, a cursory account only will be given.

1. Application of isotope technique, tracer technique

This is undoubtedly the widest field in the application of radiation techniques, comprising, with the use of *in vivo* and *in vitro* methods, almost all modern fields of biology, biochemistry, immunology, endocrinology, and several other related branches of science. The development of the *in vitro* immunobiological method, the radio-immunoassay (RIA) technique and the different activation analytical methods suitable for the determination of the elemental composition (mainly microelements) are of outstanding importance. (As a matter of fact the first activation analysis was carried out by George Hevesy, a scientist of Hungarian origin in 1936, and later he was awarded the Nobel prize for his achievements in, among others, the application of radioactive isotopes as tracer technique in biology.)

2. Radiomutation

The radiomutation improvement methods have been used with various plant species for many years. Recently, in addition to the γ -irradiation treatment used

previously, irradiations with neutrons and charged particles have become more and more common.

3. Radiostimulation

Small-dose irradiation can be used widely for the stimulation of life processes. Although the method is primarily used for the presowing treatment of sowing-seed, according to the most recent researches, the results are expected to be used also in animal breeding, e.g. radiation stimulation of fish sperm, eggs.

4. Disinfection, sterilization, food-irradiation

Ionizing irradiation is remarkably suitable for prolongation of the storability of agricultural products and foodstuffs. Depending on the applied dose the treatment is called radication, radurization, or radappertization. By irradiation, drinking water, or a culture medium can be sterilized, and the method can be used very well also for the disinfection of sewage-sludge or dung water. Irradiation can be used with success also in biological plant protection against insects (the so-called male-sterile technique).

5. Nuclear measurement and control technique

Nuclear technique renders possible the measurement and control of a number of characteristic parameters (e.g. measurement of water-content, thickness, level measurement) in several fields. These systems can be readily automatized, and in all likelihood—in the control and regulation technique of the food industry—their application will be even more extensive.

4.4 THE NUCLEAR ACT IN HUNGARY

In 1980 the Hungarian Parliament passed a law on nuclear energy, the Nuclear Act of 1980, and the Council of Ministers passed Decree No. 12/1980 (IV. 5.) on the execution of Act No. I. of 1980 on Nuclear Energy. Summarized briefly below are the contents of the Act and the Decree of execution with special emphasis on their radioecological aspects.

It is stated in the Act that the application of nuclear energy can be utilized efficiently and economically on a large scale (industry, agriculture, sanitation, scientific research) for meeting the social demand. Nuclear energy is utilized on an ever-widening scale also in Hungary. It plays an important role in providing energy for the country.

However, nuclear energy can be used as a mass destruction weapon as well. Since in Hungary nuclear energy can be used for peaceful purposes only, the Hungarian Republic makes every efforts to stop the armaments race and advocates necessary measures to achieve nuclear disarmament.

The application of nuclear energy and radioactive isotopes for peaceful purposes may also impose hazards and inappropriate application can contaminate the environment, and endanger people's health. Consequently the utilization and safe application of nuclear energy requires great professional expertise, utmost care and well-organized and executed work.

From all radiation sources, the total annual radiation load on employees working

with nuclear energy and on the population may not exceed the limit value permitted by relevant specifications which take into account the recommendations of international and Hungarian professional organizations. This specification adequately prescribes the maximum amount and concentration of radioactive materials which may be emitted, and also the mode of emission. The maximum amount and concentration of the radioactive materials which may be emitted to the environment and the conditions of the emission are determined by the Minister of Environmental Protection and Water Management in agreement with the Minister of Health. For the use of water in connection with the application of nuclear energy and for the protection of water supplies against radioactive contamination, the decrees of law on water conservancy and the decrees issued by the Minister of Environmental Protection and Water Management in agreement with the Minister of Health apply.

Nuclear energy must only be used in such a way that it does not endanger human life, or the health of the present and future generations, and that it is not harmful to material goods. In order to achieve this, nuclear energy may only be applied in the way determined by the legal decree, and under regular control of the government. It must be guaranteed that the application of nuclear energy meets the requirements of work safety, health physics, and environment protection. The employees working in places exposed to radiation hazards must regularly undergo checks of health physics and medical supervision, and it must be assured that they receive and properly use the safety and protection devices available to them. The permitted dose limit values for the employees working with nuclear energy, and for the population are determined by the Minister of Health. However, within a specific permitted dose limit value the radiation load should be kept at the minimum level that can be reasonably achieved.

The user of nuclear energy is obliged to monitor radiation levels regularly. If the level of ionizing radiation, the radioactive contamination or the amount of radioactive materials emitted to the environment exceeds the permitted limit, the appropriate measures must be immediately taken. The facilities used for the application of nuclear energy must be constructed in such a way that any emergency situation can be adequately dealt with and that the radioactive material released in an accident or some other emergency can be localized. For the protection of the population in the area around the nuclear facility, safety zones must be provided. In the case of emergencies, e.g. reactor accidents, for the safety of the population it may become necessary to evacuate a settlement (or part of it), to introduce different limitations, prohibitions, and other similar measures, and to carry out decontamination.

Although even the peaceful use of nuclear energy may impose some danger, by adequate safety measures and by adherence to the regulations these hazards can be reduced to a minimum. Therefore the basic question in the application of nuclear energy is how to protect the persons employed in this work, as well as the population and the environment, from harmful effects.

4.5 SUMMARY

In this chapter the issues related to the dangers of nuclear war, the application of nuclear weapons and the responsibility of ecologists have been outlined. I have also tried to answer the question whether the nuclear power plants are dangerous to the environment. Finally, the applicability of nuclear technique in agriculture and food industry has been presented. The important statements reflecting the author's opinion can be summarized as follows:

1. The demolishing-destructive power of modern nuclear weapons exceeds that of traditional weapons by several orders of magnitude. In a nuclear war the whole of mankind may perish. Even a limited nuclear war could cause such radioactive contamination that it would render healthy human life impossible on Earth for a long time. Therefore the prevention of a nuclear war is a vital issue for mankind. Similarly, it is a key issue to decrease as much as possible the current contamination level of the environment (by the complete cessation of atmospheric, underwater and underground weapon tests). The responsibility of an ecologist lies in calling the people's attention to the movement of radioactive materials in the elements of the biosphere, to the dangers of radioactive contamination as a special type of pollution of our environment. Ecologists should raise their voices in scientific, social and political forums and demand the banning of the application of nuclear weapons and nuclear explosions.

2. Nuclear power is the source of energy which mankind must utilize to an ever-increasing extent. It is also a fact that a not-negligible amount of radioactive material is emitted by nuclear power plants into the surface waters and into the atmosphere, even during ordinary operation. Theoretically it would be possible to decrease the radioactive emission to any optionally small value; however, energy obtained in such a way, because of its price, could not be competitive either with the energy obtained from fossil fuel, or with that generated by hydroelectric power plants. Full adherence to the stringent rules of work safety and health physics is the assurance that the increase of that part of radiation load of the environment which can be attributed to the radioactive emission of nuclear power plants still remains within an acceptable margin of risk from a sanitary point of view. Furthermore, it is a vital interest of mankind that in the future no event similar to the Chernobyl disaster could occur. As a prerequisite for this, construction design must strive for maximum safety and the operation of the facility must be both responsible and professional.

3. Nuclear technique can be utilized in a wide range of applications in several fields of research and agricultural production and the food industry. Its most important areas are the isotope technique, tracer technique, radiomutation, radiostimulation, disinfection, food irradiation and measurement and control technique.

5. CONCLUSION

Now, at the end of the 20th century, the possibilities before mankind (new fuels, modern telecommunications, computerized production control, high-degree automation, genetic engineering, etc.) which could transform our planet into a paradise have already been outlined; but so also have those which lead to destruction (nuclear war, ecological crisis, civilizational diseases, etc.). We live in an agitated, changing age and in a world, full of contradictions, often completely separated from the supposedly natural way of living, surrounded by a flood of information, amidst considerable tension on our nervous system, suffering from and at the same time enjoying what is given to the man of our times by civilization, science and modern technology.

The contamination of the biosphere by radioactive isotopes is one of the special factors of environmental pollution. It is a factor, the effect of which could be fundamental to the life of generations yet to come. It is the vital interest of mankind to put an end to atmospheric nuclear weapon tests, to prevent nuclear war, and to reduce the emissions from nuclear power plants to a minimum.

Protection against radioactive contamination is an integral part of environmental protection, an activity which is essentially pursued in order to restore the biological equilibrium of nature upset by human activity. Consequently, nature shall not be conquered, but instead, by common sense and careful economy, and in the knowledge of ecological interactions, it shall be utilized in such a way that it offers a clean and healthy environment to present and future generations alike.

APPENDIX

A very short summary is given below of the basics of radioactivity as a physical phenomenon, in relation to the analysis and measurement technique of radioactive isotopes, and also of the different forms of interaction between ionizing radiation and matter. This section is not an organic part of the book, but it will help readers unfamiliar with the isotope technique and enable them to obtain a picture of radio-analytical methods, and the equipment necessary to carry out radioecological and research investigation on the interaction between radiation and matter. At the end of this section, under the title Glossary, all the technical terms which may not be well known to the general reader, are listed and each of them is briefly explained.

Radioactivity as a physical phenomenon

Of the almost 2000 isotopes known today the number of stable isotopes is only about 300. The cause of radioactive decay is the nuclides trying to attain the minimum energy state, and to establish a proton:neutron ratio corresponding to the minimum energy. If, for example, the nucleus has a neutron in excess, it can be discharged by the emission of β -radiation, because then the mass number remains unchanged; however, the atomic number (the number of the protons) increases.

The most important characteristics of radioactive isotopes is the probability of their decay. The decay law, describing the course of radioactive transformation related to time, is of a statistical nature, and is valid only for a large number of radioactive atoms. The number of atoms decaying during a time interval is proportional to the number of atoms which are capable of decay.

The above may be formulated as

$$\frac{dN}{dt} = -\lambda N,$$

where:

λ = denotes the decay constant.

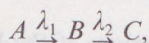
Solving the differential equation we obtain

$$N = N_0 e^{-\lambda t}$$

that is the number of radioactive atoms (and thus radioactivity) decreases exponentially as a function of time. The following relation exists between the half-life and the decay constant:

$$t_{1/2} = \frac{\ln 2}{\lambda}, \quad \text{and} \quad \lambda = \frac{\ln 2}{t_{1/2}}.$$

Within a system the concentration of active matter continuously decreases as a consequence of radioactive decay taking place spontaneously, while at the same time that of the decay products gradually increases. If the decay product is also radioactive, an equilibrium state of the activity ratios may ensue. If, for example, a certain element A decays to element B, and then the element B decays to element C



the change in the quantity of the parent element A may be given by the relation

$$\frac{dN_1}{dt} = -\lambda_1 N_1,$$

while the change in the quantity of the element B in relation to time may be written as

$$\frac{dN_2}{dt} = \lambda_1 N_1 - \lambda_2 N_2.$$

If $\lambda_2 \gg \lambda_1$ (the so-called secular equilibrium), then the activity of the parent element may be practically considered as identical with that of the daughter element. This is the case for example for ^{226}Ra (half-life 1622 years) and ^{222}Rn (half-life 3.8 days), or for ^{90}Sr (half-life 28 years) and ^{90}Y (half-life 50.4 days).

During the α -decay, two protons and two neutrons leave the nucleus, thus the relative amount of the neutrons in the nucleus increases since in the case of elements of a larger atomic number the n:p-ratio is $\gg 1$. The heavy nuclei (high atomic number) are the ones which are capable of emitting α -particles. The energy of the α -radiation is discrete. For the α -radiating isotopes found in nature, the Geiger-Nuttel rule is valid, which establishes a relation between the decay constant (λ) of the α -radiating isotope and the penetration range (R) of the α -radiation:

$$\lg \lambda = a + b \lg R.$$

During β -decay the radioactive atomic nuclei reach a stable state by disposing of excess neutrons by means of electron radiation (negative β -decay, β^-), or if there are protons in excess, by positron radiation (positive β -decay, β^+). The energy spectrum of the β -radiation is continuous, for the characterization of the radiation the maximum energy is given.

γ -radiation belongs to the electromagnetic radiations of discrete energy; it is often concomitant with β -decay. X-ray radiation is similar to γ -radiation, depending on the nature of the energy spectrum we may speak of continuous and characteristic (discrete energy) X-ray radiation.

It should be mentioned that some of the isotopes of certain elements of high atomic number (U, Cf), similarly to the other kinds of radioactive decay, may also be broken up in a spontaneous way into two or three atomic nuclei of smaller mass number and few neutrons. This phenomenon is called spontaneous fission.

Analysis and measurement technique of radioactive isotopes

The application of the analytical processes and chemical methods for the measurement or quantitative determination of the different radioactive isotopes depends to a decisive extent on the available measuring equipment. Using up-to-date γ -spectrometric equipment (semi-conductor detector of good energy resolution, multi-channel analyser, computerized spectrum evaluation) even non-destructive tests not requiring previous chemical treatment can be performed. Of course, only γ -radiating isotopes can be measured in this way.

If no excellent energy resolution measuring system is available, previous chemical separation is also required in the case of the γ -radiating isotopes, which mostly means the application of gravimetric, extraction, ion-exchange, or chromatographic methods. The NaI(Tl) scintillation detectors are widely used for the detection of γ -radiation, and recently the very good energy solution semi-conductor Ge(Li) and Si(Li) detectors and the high purity Ge detectors have become more common.

A considerable proportion of the radioactive isotopes are β -radiating: for the detection of β -radiation primarily GM-tube detectors, with β -sensitive scintillation measuring heads are used. During recent years the liquid scintillation measuring technique has become more and more widespread.

Some of the heavy radionuclides are α -radiating. Since the range of the α -particles is exceedingly short, their detection requires special measuring techniques (e.g. surface barrier detector in vacuum chamber, solid state track detector).

The routine radiometric analysis of agricultural products of plant or animal origin, and of foodstuffs is as follows:

- the β -activity of the ash of the sample gives the so-called total- (or gross-) β -activity;
- the potassium content of the sample (this can be measured e.g. from the hydrochloric solution of the ashes by flame-photometric method) makes it possible to calculate the ^{40}K activity;
- the difference of the total- β -activity and the activity of ^{40}K gives the so-called residual activity, the value of which provides information on the level of the radioactive contamination;
- the so-called metal ion-fraction activity (containing metals of the groups I to IV) can be separated as an oxalate from the hydrochloric solution of the ashes.

If the residual activity is many times that of the metal ion-fraction activity, it is probable that the sample is contaminated by a significant quantity of ^{137}Cs or ^{134}Cs isotope. In such cases it is justified to determine the ^{137}Cs activity, which can be performed either by β -measurement following a previous chemical separation (e.g.

by hexachloroplatinate, or nickel-ferrocyanide) or of the ashes or perhaps of dried original matter by a direct γ -spectrometric analysis.

If, in addition to the activity of the metal ion-fraction, knowledge of ^{90}Sr activity is also necessary, by using an appropriate chemical separation Y can be separated from Sr, and the activity of ^{90}Sr can be determined by measuring the radioactivity of ^{90}Y , being in radioactive equilibrium with the former. If the activity of the so-called Sr-fraction is made up of the activity of ^{89}Sr and ^{90}Sr (^{90}Y) only, then the ^{90}Sr content, due to the very different half-life of ^{89}Sr and ^{90}Sr , can be calculated also on the basis of the two activity measurements taken at different times.

For routine analyses a lead tower, a detector of adequate sensitivity, efficiency and resolution (e.g. scintillation measuring head), and a recording equipment (scaler, spectrometer) are necessary. To measure the radiation level of the environment a dosimeter and a dose-intensity (dose-rate) meter are required.

The interaction of ionizing radiation with matter

During the interaction of ionizing radiation with a matter, changes are produced in the condition of the radiating particles (energy, direction, absorption), and in the atoms and molecules of the matter participating in the interaction (ionization, excitation, nuclear reaction, and as secondary effect, chemical reaction). Basically the detection methods of the different kinds of radiation also rely on the nuclear radiation-matter interaction effect. The interactional cross sections (the probability of the interaction occurring) depend also on the atomic number, and hence on the chemical composition, of the matter, so they are suitable for the development of analytical methods. The interactive effects may be grouped according to the particles generating the interaction (α , β , γ), or by the part of the matter taking place in the interaction (shell electron, atomic nucleus, the Coulomb field of the atomic nucleus).

α -radiation, consisting of $^4_2\text{He}^{2+}$ atomic nuclei and generally of 3–8 MeV energy, by entering into interaction with the shell electrons, nuclei and electric field of the atoms brings about ionization, excitation (chemical transformation) and perhaps nuclear reaction. As a consequence of the interaction the α -particles are damped, scattered and absorbed. The relation between the range of penetration (R) and the energy of the α -radiation (E_α) is as follows:

$$R = a \cdot E_\alpha^n$$

in the case of air $n \approx 3/2$.

Since the specific ionizing capacity of the α -radiation is very large, its penetration range is exceedingly short, in biological materials it is only of the 10 μm order of magnitude. Ionization does not only take place directly due to the α -rays, but the shell electrons pushed out by them are also capable of producing further ionizations, i.e., the so-called secondary ionization.

When there is interaction between β -radiation and matter, the atoms or molecules of the latter are excited and ionized, and the electrons constituting the β -radiation, because of the collisions with the shell electrons or the effect of the field of the atomic

nuclei, change their direction of travel. As a consequence of the interaction, part of their kinetic energy is transferred to other electrons, or they lose some of their energy by slowing down in the electric field of the atom and simultaneously emit X-ray photons. Some of the electrons leave the matter participating in the interaction with a change of direction and energy (damping, scattering), while the rest lose their energy completely in the interior of the matter and are absorbed.

The absorption characteristics of the matter taking part in the interaction relative to β -radiation are shown by the absorption coefficient (μ). When penetrating through a layer of matter of d thickness, the intensity of I_0 decreases approximately to a value of

$$I = I_0 e^{-\mu d}$$

If the β -radiation consists of positrons, then the damped positron enters into an interaction with a shell electron, and two γ -photons each of 0.51 MeV energy are produced. This is the so-called annihilation radiation.

During the interaction of γ -radiation and matter, a great variety of phenomena take place. The photons of the radiation can enter into an interaction with the atomic nuclei, the shell electrons of the matter, and with the Coulomb field of the nuclides, and as a consequence of this they can be absorbed and flexibly or inflexibly scattered, causing ionization and excitation in the matter. Absorption primarily takes place either during the interaction with the shell electrons (photo effect), or during the interaction with the Coulomb field of the nucleus (formation of pairs). In the case of scattering the most important characteristic parameter is the so-called Compton-scattering. The photo effect, the formation of pairs (the opposite of annihilation), and the Compton-scattering are proportionate to the fourth, second, and first power of the atomic number, respectively. By the increase of the γ -energy the probability of the photo effect decreases, that of the formation of pairs increases, while the Compton-scattering has its maximum at around 0.4 MeV.

The absorption of the γ -radiation may also be described by the Lambert-Beer-type equation:

$$I = I_0 e^{-\mu x}$$

Of the ionizing radiations, γ -radiation is of the most penetrating capacity (a small fraction of it can penetrate through a layer of matter of any thickness) and consequently the values of the specific ionizing capacity and the specific energy loss (dE/dx) are the smallest in this case.

GLOSSARY

ABIOTIC FACTORS The inanimate factors of the environment, e.g. soil, air, temperature, illumination. The abiotic factors mostly exert a complex synergistic effect (a combination of one another), i.e., the individual effects of the various factors are exerted vectorially.

ACTIVATION ANALYSIS An analytical method in nuclear physics for the analysis of the elemental composition of a matter. Activation by irradiation is followed by the detection of the generated radiation.

ACTIVE MATTER, RADIOACTIVE MATTER Radiating matter which consists of atomic nuclei capable of decaying. To a smaller or greater extent all matter in our environment is active, that is, it emits ionizing radiation.

ACTIVE TRANSPORT Transport of matter through a membrane, often against a concentration gradient.

ACTIVITY The number of nuclei decaying during a time unit. At present the unit used is Bq (1 Bq=1 decay/sec). Previously the unit Ci was used (1 Ci=3.7 · 10¹⁰ decay/sec, about equivalent to the activity of 1 g of radium).

AEROSOL Solid colloidal dispersion system floating in gaseous medium, in which the aerodynamic equivalent diameter of the floating particles is in the sub- μ m range.

ANABOLISM All building processes in a living organism during which the materials taken up from the environment are incorporated into the living substance of the organism. The name of the process acting in the opposite direction is catabolism.

ANTHROPOGENIC EFFECT A biotic factor related to the activities of Man in changing nature. An example is the ecological crisis brought about by extensive environmental pollution.

ATOMIC BOMB TEST, NUCLEAR TEST, NUCLEAR WEAPON TEST Unregulated nuclear reaction, which involves a chain reaction taking place in a fraction of a second with a mass larger than that of the critical one. Depending on their mode (location), they can be grouped as stratospheric, atmospheric, surface, underground or underwater tests.

ATOMIC ENERGY, NUCLEAR ENERGY Energy released, or that which may be obtained, by fission of heavy atomic nuclei (e.g. ²³⁸U), or in the fusion of light atomic nuclei (e.g. ²H), exceeding chemical energy by several orders of magnitude. In a nuclear reaction (atomic nucleus transformation) a loss of mass will occur, and on the basis of the equivalence formula ($E=m \cdot c^2$) energy is released.

ATOMIC NUCLEUS The central part of the atom having a positive charge, and consisting of protons and neutrons. The number of protons is the atomic number, the sum of the number of the protons and neutrons gives the mass number. The mass of the atom is concentrated in the atomic nucleus, the radius of the nucleus is only 10^{-13} – 10^{-12} cm, so its density is very great, about 10^{14} g/cm³.

BECQUEREL The derived SI unit of activity of a radionuclide (radioactive). 1 Bq = 1 decay/sec.

BIOCOENOLOGY A special field of ecology, analysing the interaction between the communities of organisms (biocoenoses) and the living areas (biotopes). Differentiation can be made between phytocoenology (coenology of plants), and zoocoenology (coenology of animals).

BIOCOENOSIS The organisms of a habitat (biotope), that is the integral of the plants and animals occurring in it.

BIOGENIC ELEMENTS Elements building up living organisms, playing an important role in the biological processes (e.g. H, C, O, N, S, P, Na, K, Ca, Mg, Fe, Cu).

BIOGEOCHEMISTRY The branch of science studying the interrelations of living organisms and the geosphere which deals with biological processes taking part in the geological accumulation or migration of different elements.

BIOGEOCOENOSIS The aggregate of the biotope and the biocoenosis.

BIOGEOCYCLE The full cycle of certain matters (e.g. water, nitrogen, carbon) taking place in nature during the interactions of living organisms and inanimate matter.

BIOLOGICAL EQUILIBRIUM Such a state of equilibrium of the fauna and flora in the area in question, and within them of the species and individuals which correspond to the local conditions, although it may temporarily be broken as a consequence of random (and small-scale) alteration of the forces determining the equilibrium. However, it will return as a consequence of the self-regulating capacity of the biocoenosis. Prolonged, one-sided, extensive intervention causes the disruption of the equilibrium.

BIOLOGICAL HALF-TIME Period of time referring to the rate of replacement or removal of the different elements or compounds from the organism, during which half of the element or compound in the organism are exchanged by metabolism, that is, the original quantity is reduced to half.

BIOLOGICAL RADIATION EFFECT An effect brought about in biological systems by ionizing radiation. The radiation effect, depending on the dosage received, can be reversible or irreversible; the latter is called radiation damage. The radiation effect in organisms can be somatic (occurring in affected individuals) and genetic (affecting the gametes, observable in descendants).

BIOSPHERE The part of the Earth's atmosphere in which life can exist, the conjunction of the lithosphere, hydrosphere and atmosphere, the integral of the ecosystems.

BIOTIC FACTORS The effect of living organisms exerted on other organisms (e.g. plant-plant effect).

BIOTOPE Habitat.

CATABOLISM Decomposition, the opposite of anabolism.

CONTAMINATION Pollution.

COSMIC RADIATION High-energy radiation from space reaching the Earth in identical intensity from all directions. The primary cosmic radiation consists mainly of protons and high-energy photons.

CRITICAL MASS Minimum mass of the fissible material necessary for the occurrence of a nuclear chain reaction.

CRITICAL ORGAN Organ, the injury of which (due to increased sensitivity to radiation, or the accumulation of a radioactive isotope in it) entails the maximum health risk of an individual or that of his descendants.

CROSS SECTION A characteristic indicating the probability of the occurrence of a process (e.g. nuclear reaction).

CURIE The previously used unit of radioactivity. $1 \text{ Ci} = 3.7 \cdot 10^{10} \text{ decay/sec.}$

DECAY CONSTANT Refers to the velocity of radioactive decay, written as the formula

$$\lambda = \frac{\ln 2}{T_{1/2}}.$$

DECONTAMINATION Release or decrease of radioactive pollution.

DECORPORATION Excretion or removal of the matter in question from the organism.

DECORPORATION AGENTS Chemical substances facilitating the removal of the matter in question (e.g. radioactive element, toxic heavy metal) from the organism.

DISCRETE ENERGY Radiation energy of determined energy levels (non-continuous energy distribution).

DISCRIMINATION FACTOR A ratio which shows the relative concentration charge of radioactive isotopes during the food chain process. For example, in the case of strontium in the animal-plant system $D_{\text{Sr}} = \frac{\text{relative Sr concentration in milk}}{\text{relative Sr concentration in fodder}}$

DOSE Ratio of the energy transferred to the volume part of the radiated matter and the mass of the volume part expressed in units of gray (Gy) $\cdot 1 \text{ Gy} = 1 \text{ J/kg}$ (formerly in units of rad, $1 \text{ rad} = 10^{-2} \text{ Gy}$).

DOSE EQUIVALENT Product of the absorbed dose and the quality factor, expressing its biological effectiveness. Its unit is the sievert (Sv). 1 Sv is the dose which for a human organism has a biological effect equivalent to 1 Gy of dose absorbed from X-ray radiation. The formerly used unit was rem: $1 \text{ rem} = 10^{-2} \text{ Sv}$.

DOSE EQUIVALENT COMMITMENT The integral of the dose equivalent as a function of time (for prolonged radiation load).

DOSE INTENSITY Dose per unit of time, in the case of a punctual source of radiation, written as a formula:

$$D_i = K_p \frac{A}{l^2}$$

where:

K_p = dose constant;

A = activity;

l = distance.

DOSE RATE See **DOSE INTENSITY**.

DOSIMETRY Measurement of dose, a special field of health physics.

DRY DEPOSITION Gravitational fall-out, deposition, dispersion taking place in dry weather from the atmosphere in compliance with Stokes' law of deposition.

DRY-OUT Dry fall-out, the part of fall-out not due to precipitation.

ECOLOGY Environmental biology. The study of the natural environment, particularly the interrelationships between organisms and their surroundings.

ECOLOGICAL FACTORS Biotic and abiotic factors determining the interactions between the environment and the organisms.

ECOSYSTEM Part of nature where continuous metabolism goes on between the inanimate nature and the biocoenoses of plants and animals evolving in it.

EFFECTIVE HALF-LIFE A characteristic showing the decrease of the radioactivity of an organism, which can be calculated as a function of the physical half-life ($T_{1/2 \text{ phys}}$) and the biological half-life ($T_{1/2 \text{ biol}}$) formulated as:

$$T_{1/2 \text{ eff}} = \frac{T_{1/2 \text{ phys}} \cdot T_{1/2 \text{ biol}}}{T_{1/2 \text{ phys}} + T_{1/2 \text{ biol}}}$$

ELECTROMAGNETIC RADIATION Waves propagating in vacuum at the velocity of light ($3 \cdot 10^8$ m/sec). It consists of γ -radiation, X-ray radiation, ultraviolet, visible and infrared light, and also radio waves.

ENERGY SPECTRUM In nuclear technology the spectrum showing the distribution of the intensity of the detected particles as a function of energy.

FALL-OUT Falling out, dispersion, deposition, the sum of the wet and the dry deposition.

FISSION Nuclear fission, the decay of a large mass nucleus to smaller nuclei accompanied by release of energy.

FISSION PRODUCTS The radioactive nuclides produced as the result of fission in a nuclear test or during the operation of a nuclear power plant.

FLUX The number of particles passing through a surface unit during a time unit.

FUSION A nuclear reaction between light nuclei as a result of which a heavier nucleus is formed and a large quantity of nuclear energy is released.

GAMMA-SPECTROMETRY Record of the energy spectrum of γ -radiation by means of an appropriate detection equipment and analyser for the determination of the γ -radiation emitting isotopes.

GENE MUTATION A hereditary transformation of the genes occurring spontaneously or by induction. The majority of the mutations of the genes are of a recessive nature.

GENETIC RADIATION DAMAGE Deleterious transformation brought about in the structure of the material of inheritance by the effect of radiation.

GONAD DOSE A dose equivalent affecting the sexual organs.

GRAY The derived SI unit of absorbed dose: $1 \text{ Gy} = 1 \text{ J/kg}$.

H-BOMB A two-phase (FF) atomic bomb.

INCORPORATION Admission to the body, essentially a process following contamination, during which the radioactive isotope takes part in the metabolism, or is built into the body of the organism.

INDICATOR PLANTS Plants (sorrel, spinach, lettuce) analysed regularly—in Hungary by the Radiometrical Monitoring Network of the Ministry of Agriculture and Food—for the indication of the radioactivity level of the vegetation.

INDICATOR ORGANISMS Organisms indicating well the changes (e.g. alteration of the level of radioactive contamination) in the environment.

INDUCED RADIOACTIVITY Active materials generated in the substances of the immediate environment (especially neutron radiation) of a nuclear reaction caused by a nuclear test.

INHALATION Breathing in, respiratory incorporation.

IONIZING RADIATION Radiation that is capable of causing ionization, either directly or indirectly.

ISOTOPE EFFECT Difference resulting from the different neutron (and mass) number of the atoms, which results in somewhat different physical, chemical and biological characteristics.

ISOTOPES Different forms of an element in question (e.g. ^1H and ^3H) characterized by different number of neutrons. Found on the same place (*isos topos*) of the periodic table of elements.

LET Linear energy transfer, the average energy transferred to a certain material by a charged particle of a given energy along a short section of its path, divided by the length of the path unit.

METABOLISM Material exchange in living organisms.

METAL ION-FRACTION ACTIVITY An activity ratio referring to the existence of radioactive contamination of artificial origin. The metal ion-fraction consists of metals belonging to groups I to IV, e.g. can be separated as oxalate from the hydrochloric solution of the ashes of the analysed sample.

MeV The unit of the energy of radiation: $1 \text{ MeV} = 10^6 \text{ eV}$.

MIGRATION Movement, transport, flowing, transfer.

MINERALIZATION Biodegradation, the decay of organic substances by means of the saprophytic microorganisms of the soil.

NATURAL RADIATION LEVEL Level of cosmic radiation of natural origin, and that of the radiation of the radioactive substances found in the biosphere.

NEUTRON WEAPON A new, special version of the atomic bomb which has a smaller destructive effect (heat and shock wave); however, it produces very intensive neutron radiation, destroying living organisms.

NUCLEAR ENERGY See **ATOMIC ENERGY**.

NUCLEAR MEASUREMENT TECHNIQUE Technique requiring nuclear measuring equipment, developed for the determination of some physical or chemical characteristics (e.g. thickness, elemental composition), and applied for the detection of radiation or radioactive materials. Some typical fields of the application of this technique are e.g. radioactive tracer technique, measurement of radiation pollution, activation analysis, thickness analysis by radiation.

NUCLEAR MEASURING EQUIPMENT Equipment consisting of several units (e.g. detector, photomultiplier, amplifier, analyser, scaler, computer) depending on the nature of the task (e.g. measurement of dose, dose level or contamination).

NUCLEAR MEDICINE Branch of science dealing with the medical application of nuclear technique, primarily through the utilization of radioisotopes in diagnostics and therapy.

NUCLEAR POWER PLANT A power plant for primarily power generation purposes, in which the thermal energy is produced by a high-power nuclear reactor. The nuclear power plant at Paks, Hungary, consists of blocks of 440 MWe.

NUCLEAR REACTION Process taking place in the atomic nucleus spontaneously or by some external effect, which results in the change of the atomic nucleus (e.g. fission, β -decay).

NUCLEAR REACTOR An equipment which is used for the generation of different radiations or isotopes, and for the production of nuclear energy. The essence of the operation of the reactor is regulated chain reaction. At present only fission reactors operate; however, in the 21st century in all probability fusion reactors will also be built. The first nuclear reactor (atomic pile) was put into operation by Fermi in 1942; figuratively speaking this was the second lighting of fire made by mankind. It was in 1959 when the first reactor in Hungary was put into operation in the Central Research Institute for Physics.

NUCLEAR TEST See **ATOMIC BOMB TEST**.

NUCLEAR TEST BAN TREATY In 1963 the nuclear powers signed in Moscow the so-called limited nuclear test ban treaty, in which they agreed to stop all nuclear tests on the surface of the Earth, in the atmosphere and the stratosphere. Countries who are not party to the convention (e.g. China), unfortunately continue with their nuclear tests. The Moscow treaty does not apply to underground tests.

NUCLEAR WAR War which, instead of the traditional military equipment, is fought with nuclear weapons of very powerful destructive capacity.

NUCLEAR WEAPON A weapon which, compared to the traditional ones, is of a very powerful destructive capacity (radiation, light and thermal effect, air pressure wave), the force of which is characterized by the TNT-equivalent. The force of a three-phase nuclear weapon, the so-called FFF bomb, can reach the value of even several hundred Mt TNT.

NUCLEAR WEAPON TEST See **ATOMIC BOMB TEST**.

PHYSICAL HALF-LIFE Period of time during which the activity of a radioactive isotope in question is reduced to one half its previous level. It is inversely proportional to the decay constant:

$$T_{1/2} = \frac{\ln 2}{\lambda}.$$

PLUTONIUM BOMB Single-phase atomic bomb with ^{239}Pu charge.

POLLUTION Contamination, vitiation, fouling.

POTASSIUM ACTIVITY Activity ratio which can be attributed to the presence of ^{40}K isotope. It constitutes a significant part of the natural radioactivity of the biological substances.

QUALITY FACTOR (QF) Factor indicating the biological effectivity of the dose. Its value is 1 for X-ray and γ -radiation, 3–5 for slow neutrons, 10 for protons and fast neutrons, and 20 for α -radiation.

RAD The former unit of absorbed dose of ionizing radiation. $1 \text{ rad} = 10^{-2} \text{ Gy}$.

RADAPPERTIZATION High dose radiation treatment (20–60 kGy) to guarantee full storability and germ-free condition of a foodstuff.

RADIATION BIOLOGY Radiobiology.

RADIATION DOSE Radiation load affecting an inanimate substance or a living organism (organ, tissue, etc.) from an exterior or interior source of radiation. Its unit is Gy, or Sv. See **DOSE**, **DOSE EQUIVALENT**.

RADIATION INHIBITION Inhibition of physiological processes by means of radiation treatment. In the case of onion or potato for example the inhibition of germination, and thereby the prolongation of the storage period, can be attained by it.

RADIATION STERILIZING Disinfection by means of radiation, the destruction of microorganisms by radiation.

RADICIDATION Radiation treatment of foodstuffs and agricultural products of vegetable origin against different plant and animal pests (e.g. insects, microorganisms) harmful to health.

RADIOACTIVE Emitting radiation, active.

RADIOACTIVE CONTAMINATION Radiocontamination, radioactive pollution.

RADIOACTIVE DECAY Transformation of the nuclei of radioactive atoms to the nuclei of other elements, accompanied by the emission of radioactive radiation.

RADIOACTIVE EQUILIBRIUM May occur between a parent and a daughter element if the daughter element is also radioactive (e.g. ^{90}Sr – ^{90}Y).

RADIOACTIVE MATERIAL See **ACTIVE MATTER**.

RADIOACTIVE RADIATION Ionizing radiation emitted by radioactive isotopes.

RADIOBIOLOGY, RADIATION BIOLOGY A special field of biology closely related to biophysics, medical radiology and health physics, the subject of which is the analysis of the effect of exterior and interior sources of radiation on biological objects.

RADIOCESIUM Radioactive cesium (e.g. ^{137}Cs).

RADIOCARBON Radioactive carbon, the radiating isotope of carbon (^{14}C).

RADIOCHEMISTRY Radiation chemistry, an interdisciplinary branch of science incorporating nuclear chemistry, isotope technique and analytical chemistry.

RADIOECOLOGY Radio-environmental biology, radio-environmental science.

RADIOGRAPHY Photoemulsion picture of radioactive isotopes, or emitted radiations.

RADIOIMMUNOASSAY (RIA) Very sensitive radioimmunological measuring technique based on the determination of the radioactivity of the immune serum.

RADIOMETRY Measurement of radiation level, radioactivity or radioactive contamination.

RADIOMUTATION Mutations produced by radiation treatment.

RADIONUCLIDE Radioactive isotope, radioactive atomic nucleus.

RADIONUCLIDE MONITORING See **INDICATOR ORGANISMS**.

RADIOPHARMACOLOGY A special field of pharmacology analysing the route of medicines in the organism by means of labelled (radioactive) compounds, a special field of diagnostics and therapy.

RADIOPHARMACONS Open radioactive products used for medical diagnostics and therapy.

RADIOSTIMULATION Promotion of physiological processes by means of radiation. In the case of plants for example, increase of yield can be attained.

RADIOSTRONTIUM Radioactive strontium (e.g. ^{90}Sr).

RADIOTOXICITY Depending on their hazardous character (half-life, radiation energy, etc.) the radioactive isotopes can be divided into different groups of dangerousness. Group no. I (exceedingly dangerous) includes ^{90}Sr , ^{210}Pb , ^{226}Ra , group no. II (very dangerous) includes ^{89}Sr , ^{95}Zr , ^{106}Ru , ^{131}I , ^{137}Cs , ^{146}Ba , ^{144}Ce , group no. III (moderately dangerous) includes ^7Be , ^{14}C , while group no. IV (slightly dangerous) holds ^3H .

RADIOTOXICOLOGY Branch of science dealing with the effect of radioactive substances taken up (incorporated) by an organism, a special field of toxicology.

RADIUM Alkaline earth metal, a natural radioactive element which transforms into radon by means of α -decay, one of the elements of the ^{238}U decay series.

RADON Decay product of ^{226}Ra , radioactive inert gas.

RADURIZATION Increase of the storage period, the improvement of the conservation quality of foodstuffs by means of a radiation dose below 10 kGy.

RAIN-OUT Falling out by precipitation, part of wet deposition, transfer of radioactive (and other) substances present in the cloud layer to the rainwater.

RBC WEAPON Radiological, biological and chemical weapon.

REM Former dose unit (dose equivalent) affecting the human organism: $1 \text{ rem} = 10^{-2} \text{ Sv}$.

REPROCESSING Processing the burnt-out fuel rods of reactors in order to utilize material capable of fission.

RESIDUAL ACTIVITY Difference of the total β -activity and the ^{40}K activity in ashes of materials of biological origin.

REUTILIZATION Recycling, it can be explained by the translocation of the different elements or components within the organism.

SOMATIC RADIATION INJURY Radiation load affecting an individual.

SPECIFIC ACTIVITY Activity related to an element of its own.

SPECIFIC IONIZATION The number of pairs of ions per unit length along the path of an ionizing particle.

SPECTRUM Spectral range which can be obtained by the resolution of light as electromagnetic radiation according to its wavelength (energy). See also **ENERGY SPECTRUM**.

STRATOSPHERE Cloudless layer of the atmosphere at an altitude of over 10 km.

Sv (SIEVERT) The SI unit of dose equivalent (equivalent dose for man).

TERRESTRIAL RADIATION Part of the natural radiation which originates from radioactive rocks, and the radioactive isotopes occurring in the soil.

THRESHOLD DOSE Limit value necessary as a minimum dose to induce effect. In the case of radiation dose for example, the genetic effect has no threshold dose, that is, even the smallest radiation dose may be sufficient to produce a gene mutation.

TNT EQUIVALENT Trinitro-toluene (trotyl) equivalent indicating the destructive power of a nuclear weapon.

TOTAL- β -ACTIVITY Sum of the radioactive isotopes emitting β -radiation of natural or artificial origin from the analysed sample (generally in ashed form). In the

present state of contamination a large amount of the total- β -activity of the biological samples originates from the ^{40}K content (gross-activity).

TOXICITY Toxic character, poisonous effect.

TROPOSPHERE Cloudy layer of the atmosphere below the stratosphere.

URANIUM BOMB A single-phase nuclear weapon with a ^{235}U charge, based on the phenomenon of nuclear fission.

WASH-OUT Washing out, part of wet deposition, decontamination taking place in the volume of air below the clouds by the effect of precipitation.

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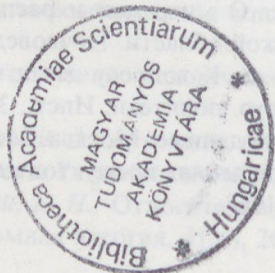
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RADIOECOLOGY AND ENVIRONMENTAL PROTECTION

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This book reviews the occurrence of varying concentrations of different radio-isotopes in successive steps of biological chain processes. It considers the factors which affect the transfer of these radio-isotopes, their changes in concentration, and the possibilities for preventing or decreasing the level of contamination.

The material covers a wide range of topics and emphasizes environmental protection, giving useful information regarding the relationship between radiation protection, agricultural practices and dietary intakes. The book contains detailed data accrued from the Chernobyl disaster, as well as other important Eastern European literature, to illustrate points made in the text. The book covers the entire biological chain of radioecology, stressing the interactions between the different parts of the biosphere experiencing radioactive contamination.

Readership: Researchers, postgraduates and senior-level undergraduates, and all those working in the areas of ecology, environmental science, physics, radiation biology, radiation protection, and dosimetry. Biological scientists, those involved in plant, animal and human physiology. Agricultural and food industries, soil scientists.

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