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Feature article

Human health damages due to ionising radiation in life cycle impact assessment

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Abstract

This paper describes the assessment of the human health damages related to the man-made routine releases of radioactive material to the environment as reported in Life Cycle Assessment (LCA). The fate and exposure analyses are based on site-specific modelling of the French nuclear fuel cycle, from which generic exposure factors are derived. The effect analysis is based largely on epidemiological studies. The damage analysis relies on the concept of disability adjusted life years (DALY). Cultural theory is used in the damage assessment to create two value-compatible assessment scenarios. Two sets of damage factors for damage-oriented and two sets of equivalency factors for effect-oriented impact assessment methods are presented. An assessment of human health damages of different electricity supply systems shows that low-dose ionising radiation contributes to 13% and 80–99% of total human health damages of nuclear power production, applying an individualist and an egalitarian perspective, respectively.

Keywords: Ionising radiation; Human health; Nuclear power; Life-cycle impact assessment; Cultural theory; Life Cycle Assessment; Eco-indicator

1. Introduction

The nuclear fuel cycle, phosphate rock extraction, coal power plants, and even oil and gas extraction are man-made sources of air- and waterborne radionucleide releases to the environment. Up to now, such emissions have seldom been considered in life cycle assessment, due to a lack of appropriate

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operationalised impact assessment models. However, ionising radiation has been mentioned in LCA literature for some time (see, e.g., [1], p. 76ff; [2], p. 33ff; [3]). In 1994 and 1996 the energy systems database "Ökoinventare von Energiesystemen" [4,5] published emission data for a large number of radionucleides emitted within the nuclear fuel cycle and by coal power plants. In 1995, the first ExternE reports included a detailed assessment of the health effects of nuclear facilities in France [6]. Based on a draft version, Martin [7] created an impact category for radioactive releases compatible with impact-oriented characterisation method (e.g., [1,8]. Frischknecht [9, p. 129ff] further refined this approach and made it compatible with the Eco-indicator 95 [35].

However, human health damages due to radionucleides and due to airborne pollutants such as SO_x, particulate matter, or carcinogenic chemicals have not yet been commensurable in the context of LCA. This paper bridges this gap and aims for a damage-oriented assessment for human health effects of radionucleides emitted by European nuclear fuel cycles. The assessment is mainly based on site-specific fate and exposure models for French nuclear facilities. For our purpose (i.e., its application in life cycle assessment), data have been generalised to render the assessment site-independent.

We describe the human health effects related to the routine releases of radioactive material to the environment. Health effects due to eventual large accidental releases are not considered, because they are outside the methodological framework of LCA. Health effects due to occupational exposure are not considered, due to consistency reasons[1]. Health impacts due to radioactive waste disposal in underground facilities are disregarded because no data for radionucleide releases into groundwater are provided in the energy systems' database [4, 5]. Furthermore, the approach described in this paper does not cover effects on ecosystems. A proposal for how to assess radiological impacts on the natural environment is, for instance, given by Solberg-Johansen et al. [11].

2. Methods

Starting from atmospheric and liquid discharges into the environment, their dispersion in different media (atmosphere, rivers, lakes, the ocean and soil) is modelled. Thereby, the main pathways from the points of release to the receiving population are considered. These fate and exposure analyses are based mainly on Dreicer et al. (1995), who use French production sites as the main data source. The health effects (i.e., different kinds of cancer and severe hereditary effects) due to the exposure of human beings to irradiation are then calculated statistically. The severity of these different health effects is weighted by applying the concept of disability

[1] Pneumoconiosis of coal miners, for instance, is not considered in the human health damage category of Eco-indicator 99 either [10].

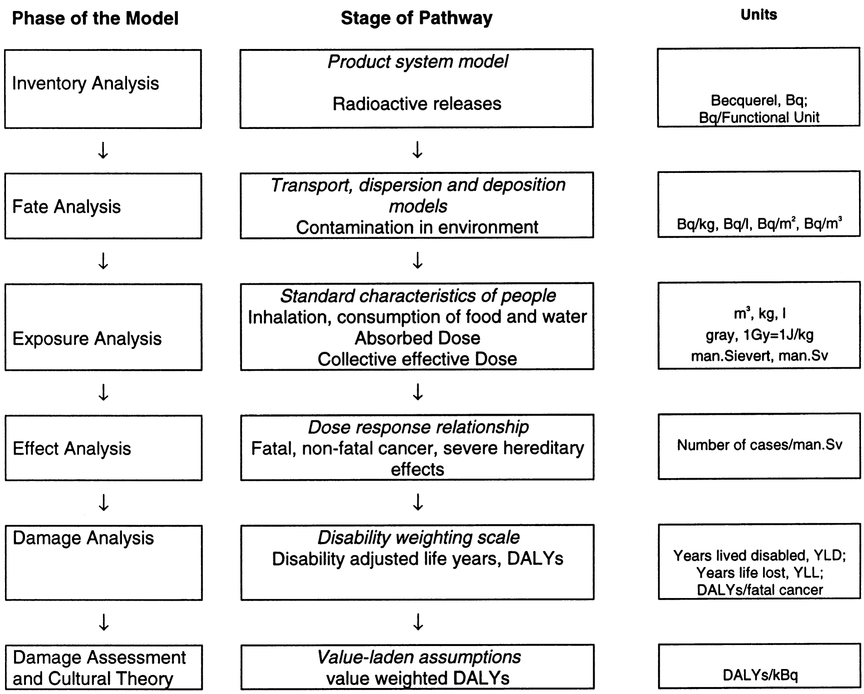

Fig. 1. Overview of impact pathway stages of radioactive releases in the assessment described in this paper, based on Dreicer et al. [6, p. 19] and Hofstetter [14]. See text for an explanation of radiation-related terms and units. Units used in the damage analysis are explained in Section 5.

adjusted life years (DALY), developed by Murray and Lopes [12,13] for the World Health Organisation and the World Bank. Furthermore, cultural theory is used to consistently model value-laden assumptions necessary in the impact assessment models and at points in the inventory analysis. Uncertainty is expressed in 95% confidence intervals, which are estimates, based on the literature reviewed. Figure 1 gives an overview of the entire assessment method of health damages due to ionising radiation introduced in this paper.

The assessment method is designed for an LCA sensu stricto as described, for example, by Hofstetter [14, p. 12][2]. Therefore, site-specific information must be generalised. Because of its high influence on the fate and exposure of liquid effluents, information about the kind of receiving body (rivers and lakes or the ocean) is maintained in this damage assessment. The assessment

[2] The following attributes of an LCA sensu stricto are especially relevant in our context. It includes no detailed information on the time pattern, nor on the site of releases, and it relies on fully quantitative information on a ratio scale, is based on simple linear or linearised models, and follows the 'less is better' approach.

is developed in view of damage-oriented impact assessment methods such as the Eco-Indicator 99 [10]. However, it also perfectly suits effect-oriented characterisation approaches (e.g., [1,8]) by simply introducing a reference substance such as iodine-129 that has been chosen by Frischknecht [9].

The new approach presented in this paper is a compromise between generic impact assessment data still required in most of today's LCAs, site-dependent data and site-specific data published by, e.g., Dreicer et al. [6]. As a first attempt, site-specificity is judged to be of minor importance, due to similarity of site-specific exposure factors and the similarity in emission patterns of nuclear fuel cycles. A reduced site-dependency is considered by distinguishing between exposure factors for liquid releases into sea and into fresh water bodies (e.g., rivers). The coefficients given in this paper are applied in a generic way to inventory tables of different electricity supply systems. The coefficients are, of course, open to changes due to future improvements of the models. However, the main structure of the proposed approach is judged to be maintainable.

The following terms and units are used in this paper:

- Becquerel (Bq) is the unit for measuring radioactive decay: 1 Bq is equivalent to one radioactive decay per second. (The former unit for Bq was Curie (Ci), with 1 Ci = $3.7 \cdot 10^{10}$ Bq). Becquerel is a measure at the point of emission.
- Gray (Gy) is the measure of absorbed dose, not considering the different reaction types of body tissues. It is the energy absorbed per unit mass of the irradiated material (1 Gy = 1 J/kg) and is the fundamental dosimetric quantity in radiological protection.
- Sievert (Sv) is the unit for measuring the effective dose, based on human body equivalence factors for the different ionising radiation types (α-, β-, γ-radiation; neutrons). 1 Sv = 1 J/kg body weight. Sievert is a measure at the point of immission. Sv contains physical data on energy doses and biological data on the sensitivities of different body tissues. It is also used to quantify the committed effective dose (effective dose integrated over 50 years) and of the average individual dose (committed effective dose received by an average individual).
- Man Sievert (man.Sv) is the collective dose, calculated by multiplying the average individual dose representative of the population by the number of people affected and integrating it over a specified time horizon.

3. Fate and exposure analyses

3.1. The ExternE model

In the ExternE model, routine[3] atmospheric and liquid discharges of all steps in the French nuclear fuel cycle are considered (Dreicer et al. 1995).

[3] Routine emissions: Emissions due to normal operation excluding severe accidents.

It is assumed that these emissions occur on a continuous basis throughout the year. For each radionucleide, the collective dose has been calculated considering all relevant pathways.

3.1.1. Fate analysis

Data of discharges from the sites (mining and milling, conversion, enrichment, fuel fabrication, electricity production, and reprocessing) and of the surrounding conditions (population density, lifestyles of that population, meteorology, etc.) refer to the French and European situation. The models use a time horizon of 100,000 years, to take into account the longevity of some radionucleides[4]. For the assessment of long-term global exposure, the world population is assumed to remain at a constant 10^{10} people for 100,000 years.

For the dispersion of atmospheric discharges, a Gaussian plume model is used[5]. The estimated levels of uncertainty of the atmospheric dispersion are a factor of 2 to 4 for the local and a factor of >4 for the regional dispersion [6, p. 28].

For liquid releases into rivers, a simple box model is used, dividing the river into several sections and assuming instantaneous mixing in each section. The radionucleide concentration in a compartment is represented with a differential equation. The estimated level of uncertainty is a factor of 5. For liquid discharges into the sea, a model of the European sea is used (including the northern European waters and the Mediterranean sea). The concentration in each compartment is time-dependent, as are the transfer rates. The estimated level of uncertainty is a factor of 2 to 3.

For globally dispersed radionucleides (i.e., tritium (H-3), carbon-14, krypton-85, and iodine-129], simplified models over a time horizon of 100,000 years are applied. For H-3 the global hydrological cycle is modelled dynamically based on seven compartments. For C-14 four environmental compartments are used in a dynamic model. For Kr-85 a dynamic model with two compartments (for the two hemispheres) is used. For I-129 a dynamic model with nine compartments is applied. The models and results used by Dreicer et al. [6] were published initially by IAEA [15].

The confidence in the results of the global assessments for tritium, carbon-14, iodine-129, and krypton-85 is low, "due to the extremely general models that are used and the propagation of very small doses over a large population for very long periods of time" [6, p. 310]. Dreicer et al. judge the uncertainty in the estimates made for assessing the global impacts

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[4] With half lifes of 1.6×10^7 years for iodine-129 or 7.1×10^8 years for uranium-235, additional impacts are to be expected beyond 100,000 years. According to Dreier et al. [6, p. 52], only about 15% of the collective effective dose of iodine-129 occur during the first 100,000 years, compared with an assessment until infinity.

[5] Although not valid for distances above 10 km, Dreicer et al. [6, p. 28] use this model for regional impacts as well. According to them errors are minor, due to fairly uniform population and agricultural production distribution.

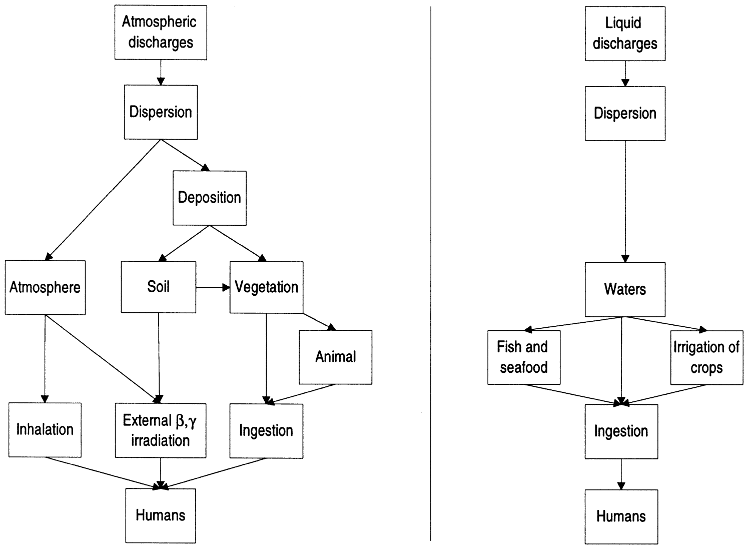

Fig. 2. Pathway for atmospheric and liquid release, respectively, of radionucleide into terrestrial (left) and aquatic environment (right), according to Dreicer et al. [6, p. 20].

probably to be greater than an order of magnitude, except in the case of carbon-14, where the global carbon cycle is quite well known.

3.1.2. Exposure analysis

To assess the exposure of humans, an increase in radiation in air, water, soil, and vegetation is determined based on the radionucleides' transport, dispersion, and deposition. The pathways for atmospheric and liquid discharges as applied by Dreicer et al. [6] are shown in Figure 2. The aquatic pathway is further subdivided in pathways for river and marine releases, respectively. Furthermore, the increase in radiation in air and soil leads to additional, external β and γ irradiation. Exposure occurs via direct inhalation, external irradiation via air and soil, and ingestion of plants (including irrigated crops) and animals (including fish and seafood), leading to additional collective doses in human beings.

The main pathway of radionucleides relevant in the damage assessment of electricity supply systems, shown in Section 7, is intake through the diet for C-14 and I-129, external irradiation for Kr-85 [15], and inhalation for Rn-222 [6].

3.2. Exposure factors

For the assessment of the exposure from radionucleides released by processes of the European nuclear fuel cycle, data from Dreicer et al. [6] will be used and partially completed with data from the reports of the

Table 1
Exposure factors (collective dose per activity released) of radionucleides[a]

Exposure factor (man.Sv/kBq)	Atmospheric releases	Liquid releases into rivers	Liquid releases into the ocean	Reference[b]
Silver-110 (Ag-110m)	—	3.3×10^{-10}	—	[6]
Americium-241 (Am-241)	—	—	2.1×10^{-8}	[6]
Carbon-14 (C-14)	—	—	7.8×10^{-10}	[6]
Curium alpha (Cm alpha)	—	—	3.8×10^{-8}	[6]
Cobalt-58 (Co-58)	2.8×10^{-10}	2.7×10^{-11}	—	[6]
Cobalt-60 (Co-60)	1.1×10^{-8}	2.9×10^{-8}	2.6×10^{-10}	[6]
Cesium-134 (Cs-134)	7.9×10^{-9}	9.5×10^{-8}	5.2×10^{-11}	[6]
Cesium-137 (Cs-137)	8.9×10^{-9}	1.1×10^{-7}	5.2×10^{-11}	[6]
Iodine-131 (I-131)	1×10^{-10}	3.3×10^{-10}	—	[6]
Iodine-133 (I-133)	6.2×10^{-12}	—	—	[6]
Manganese-54 (Mn-54)	—	2.1×10^{-10}	—	[6]
Lead-210 (Pb-210)	1.0×10^{-9}	—	—	[18]
Polonium-210 (Po-210)	1.0×10^{-9}	—	—	[18]
Plutonium alpha (Pu alpha)	5.5×10^{-8}	—	4.9×10^{-8}	[6]
Plutonium-238 (Pu-238)	4.4×10^{-8}	—	—	[6]
Radium-226 (Ra-226)	6.0×10^{-10} [c]	8.5×10^{-11} [d]	—	
Radon-222 (Rn-222)	1.6×10^{-11}	—	—	[6]
Ruthenium-106 (Ru-106)	—	—	9.5×10^{-11}	[6]
Antimony-124 (Sb-124)	—	5.4×10^{-10}	—	[6]
Antimony-125 (Sb-125)	—	—	9.8×10^{-12}	[6]
Strontium-90 (Sr-90)	—	—	2.7×10^{-12}	[6]
Thorium-230 (Th-230)	3.0×10^{-8}	—	—	[18]
Uranium-234 (U-234)	6.4×10^{-8}	1.6×10^{-9}	1.5×10^{-11}	[6]
Uranium-235 (U-235)	1.4×10^{-8}	1.5×10^{-9}	1.6×10^{-11}	[6]
Uranium-238 (U-238)	5.4×10^{-9}	1.5×10^{-9}	1.5×10^{-11}	[6]
Xenon-133 (Xe-133)	9.4×10^{-14}	—	—	[6]

[a] The squared geometric standard deviation σ_g^2 for each exposure factor is 10 (assumption based on qualitative information). Dividing and multiplying the best estimate by σ_g^2 spans the 95% confidence interval.

[b] [6] Dreicer et al. 1995: Factors are derived from emissions per year and collective dose per year. [18] UNSCEAR 1993: Factors are taken directly from the reference.

[c] Based on the assumption that the Ra-226-emission of 2 kBq/kg natural uranium released during mining and milling [5, part VII, p. 56] leads to the Ra-226-concentration in rivers of 40 Bq/m^3 used by Dreicer et al [6, p 109].

United Nations Scientific Committee on the Effects of Atomic Radiation, UNSCEAR [16–18]. Compared with UNSCEAR assessments, which are based mainly on North-American data, the assessment done by Dreicer et al. [6] is more appropriate to European situations because it reflects human health effects caused by the French nuclear fuel cycle.[6]

The collective doses per activity of selected radionucleides released are shown in Table 1 for atmospheric and liquid releases and in Table 2 for globally dispersed radionucleides. For details about models and parameters, readers are referred to the original sources.

The effective dose factor for long-term radon-222 emissions from uranium mill tailings is highly disputed [19,20]. From the SENES report [21] exposure factors for Rn-222 are derived that are between 7 and 45 times

[6] In UCPTE-Europe, French nuclear power plants contribute more than 50% to nuclear electricity.

Table 2
Exposure factors (collective dose per activity released) of radionucleides[a]

	Local and regional exposure and exposure from global dispersion during 100,000 years		Local and regional exposure and exposure from global disperion during 100 years	
	Exposure factor (man.Sv/kBq)	Assumed standard deviation σ_g^2	Exposure factor (man.Sv/kBq)	Assumed standard deviation σ_g^2
Atmospheric releases				
Carbon-14 (C-14)	1.4×10^{-7}	10	1.3×10^{-8}	10
Tritium (H-3)	9.5×10^{-12}	20	9.5×10^{-12}	20
Iodine-129 (I-129)	6.2×10^{-7}	50	1.9×10^{-7}	20
Krypton-85 (Kr-85)	9.3×10^{-14}	20	9.3×10^{-14}	20
Liquid releases				
Tritium (H-3) into rivers	3.0×10^{-13}	20	3.0×10^{-13}	20
Tritium (H-3) into the ocean	4.6×10^{-14}	20	4.6×10^{-14}	20
Iodine-129 (I-129)	6.6×10^{-8}	50	1.5×10^{-8}	20

[a] Based on Dreicer et al [6], including global collective doses with different time horizons. Dividing and multiplying the best estimate by σ_g^2 spans the 95% confidence interval.

lower than the value assessed by both UNSCEAR [18] and Dreicer et al. [6]. The most important causes for the large difference are much lower site-specific population densities compared with UNSCEAR [18] and Dreicer et al. [6], as well as a lower dose-conversion factor, compared with Dreicer et al. [6]. Because data for one-third of the actual uranium production capacity and of recently abandoned mines are missing, we use the value assessed for the French mine in Lodève and the reference mine from UNSCEAR reports [17,18], respectively.

As we will show in Section 7, radon-222 may have a large influence on the total human health damage, despite its relatively short half-life time of 3.82 days and its limited temporal and geographical reach. However, the half-life time of its precursor radionucleide, thorium-230, is 80,000 years and the anthropogenically caused release of radon-222 will diminish only according to this slow decay. In the LCI-example in Section 7 we assume that the exposure factor of radon-222 is constant during the emission period of 80,000 years considered in mining and milling uranium.

Average exposure factors based on fate and exposure analyses for different sites (i.e., different steps in the nuclear fuel cycle) have been used only for selected airborne radionucleides released by the power plant and the reprocessing plant (carbon-14, tritium, iodine-129 and krypton-85).

The exposure factors used are based on assessments done with the use of French sites and their specific meteorological and demographic situation. Limitations of radiological impacts of radionucleides with short half-lives are taken into account. In Dreicer et al. [6, p. 310] it is estimated that "for the 67% confidence interval, the results are considered to be correct within an order of magnitude." If we assume that the values are distributed lognormally [14, p. 230 f] the above information results in a squared geometric standard deviation σ_g^2 of 10 for the exposure factors of radionucleides with local and regional impacts.[7]

Whereas for the isotopes listed in Table 1 the local and regional exposure are decisive, there are isotopes (see Table 2) for which the global exposure is far more important. The global impacts (e.g., for the release of Carbon-14) are based on the integration of small individual doses over a very large time and geographical scale (100,000 years for an assumed constant global population of 10^{10} people).

Because of the high uncertainty related to globally dispersed radionucleides carbon-14, tritium, iodine-129, and krypton-85, Table 2 shows two different exposure factors. In a first case, global exposure is considered for the first 100,000 years; in a second case, global exposure is considered for the first 100 years only.

The global impacts of tritium and krypton-85 do not increase after 100 years, and therefore the exposure factors of these two radionucleides do not increase when the larger time frame of 100,000 years is applied. Based

[7] Dividing and multiplying the best estimate by σ_g^2 spans the 95% confidence interval.

on uncertainty considerations cited in section 3.1, the squared geometric standard deviation is assumed to be 10 for carbon-14, irrespective fo the time horizon. For H-3 and Kr-85 the squares geometric standard deviation is assumed to be 20; for I-129 a value of 50 is assumed for its long-term impacts.

4. Effect analysis

4.1. Introduction

Ionising radiation transfers energy into the body tissue and may thereby interfere in the structure of molecules (see [22] or [23]). In living organisms, this energy transfer may disturb or destroy cellular functions (somatic effect: i.e., fatal and nonfatal cancer) or it may change the genetic code of cells (hereditary effect).

Concerning the probability of cell changes, two types can be distinguished: *deterministic* and *stochastic* damages. For deterministic damages the severity of the damages is proportional to the dose (with threshold) and for stochastic effects the probability but not the severity is proportional to the dose. In this paper, only stochastic effects are considered, because only routine releases are included in the impact assessment (which excludes severe accidents at nuclear power plants).

4.2. Carcinogenic effects

An important discussion is whether, and how, epidemiological findings at medium and high exposure can be extrapolated to low doses. Linear, supralinear, sublinear, threshold models and even beneficial effects of low radiation levels thanks to a hormetic[8] effect have been suggested [24, p. 351]. Despite the fact that all hypotheses are supported by some experts, most international advisory boards assume a linear–no threshold (LNT) behaviour for low doses of ionising radiation [18, 24, 25]. The slope including high dose rates can be best described as S-shaped, and the section where no acute effects are observed is supposed to follow a linear–quadratic function (curve A in Figure 3).

Most of the epidemiological information is available from the quadratic section. A linear extrapolation of this high dose rate data to the origin would lead to Curve B with the slope α_L. However, Curve D with the slope α_1 gives an approximation of the slope of Curve A for low dose rates. The ratio between α_L and α_1 is the so-called *dose and dose-rate effectiveness factor* (DDREF) which was found to be between less than 2 and 10 [25]. A DDREF of, e.g., 5 means that the risk increase per man.Sv observed at high doses is divided by 5 to assess risks at low doses. All higher DDREF values stem from animal tests. Epidemiological data on the association

[8] Hormetic effects are effects stimulating the resistance system of an organism.

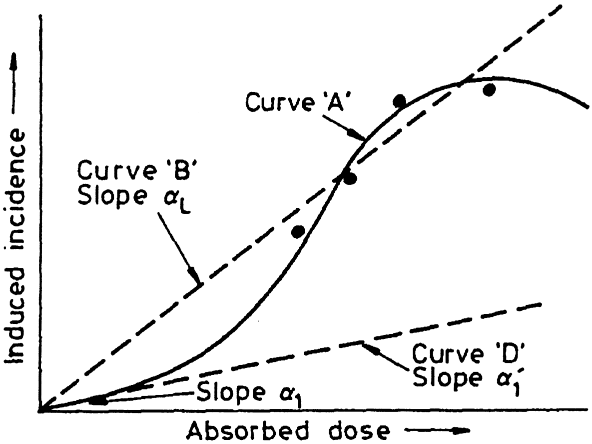

Fig. 3. Schematic curves of cancer incidence versus absorbed dose to illustrate the extrapolation from high to low dose rates (based on [34]).

between exposure doses and cancer cases are available from a still ongoing study with the survivors of the atomic bomb attacks on Hiroshima and Nagasaki. This study includes survivors exposed to high as well as to low doses. A dose and dose-rate effectiveness factor of 2 is recommended by ICRP for the extrapolation to low doses, although the ICRP "recognises that the choice of this value is somewhat arbitrary and may be conservative" [25, p. 19].

The occurrence of cancer is furthermore specified according to the tissue or organ affected. Table 3 shows the lethality fraction and probability of fatal and non-fatal cancer per tissue and organ.

The extrapolation models for dose and dose-rate effectiveness factors, the limited number of available occupational studies, and also the life span studies after the atom bomb attacks on Hiroshima and Nagasaki suggest that the discussion may lead to a range of values rather than the best estimates presented in Table 3. We assume that the probability distribution is lognormal with a squared geometric standard deviation of 3. This includes the probability of confounding factors that are not considered, the uncertainty in the dose and dose-rate effectiveness factor and the extrapolation method. However, a hypothetical threshold behaviour or the hormesis hypothesis are not within the considered range.

Ron and Muirhead [26, p. 170] report the level of association between radiation and various tumour types. They show that for 80% of fatal cancers there is definite or probable evidence of a link to radiation. For nonfatal cancers this share is about 98%. A closer look at their basis and the reasoning reveals that they generated statistics mainly from available epidemiological data. Some tumour types occur in large numbers in the control group

Table 3
Lethality fractions and probabilities of occurrence for the different cancers considered [25] and level of association based on epidemiological studies (atomic bomb survivors and medical radiation) reported by Ron and Muirhead [26, p. 170][a]

Tissue or organ	Lethality fraction (−)	Fatal cancers (10^{-2} cases per man.Sv)	Non-fatal cancers (10^{-2} cases per man.Sv)	Level of association
Bladder	0.5	0.3	0.3	Probable
Bone marrow	0.99	0.5	0.005	Definite
Bone surface	0.7	0.05	0.021	[b]
Breast	0.5	0.2	0.2	Definite[c]
Colon	0.55	0.85	0.695	Probable
Liver	0.95	0.15	0.008	Possible
Lung	0.95	0.85	0.045	Definite
Oesophagus	0.95	0.3	0.016	Possible
Ovary	0.7	0.1	0.043	Probable
Skin	0.002	0.02	9.98	Probable
Stomach	0.9	1.1	0.122	Probable
Thyroid	0.1	0.08	0.72	Definite
Remainder	0.71	0.5	0.204	[b]
Total		5	12.36	

[a] The squared geometric standard deviation (lognormal distribution) is estimated to be a factor of 3 for all tumour types.
[b] No information available in the respective reference.
[c] Female breast.

as well as in the study group. In these cases, the statistical variation is smaller than for less frequent tumour types. The other relevant parameter concerns the relative risk[9]. If the relative risk is high, then it is more probable that the lower end of the probability distribution still shows a positive excess relative risk. These factors are mainly responsible for the finding that the level of association is low for some specific tumour types. However, the level of association of the occurrence of tumors as such, disregarding different tumour types, is "definite" according to ICRP [25].

4.3. Hereditary effects

Severe hereditary effects are estimated to occur in 0.01 cases per man.Sv [25]. This number is very uncertain, because it was derived from animal tests. Kaul and Bennett [27] confirm the high uncertainty but report that UNSCEAR suggests staying with the present estimate. Sankaranarayanan [28] points out that some ICRP assumptions [25] may be overestimates. However, he neglects the arbitrary correction factor for the severity that is needed to convert all kinds of hereditary effects to severe hereditary

[9] The relative risk is defined as the risk of the study group divided by the baseline risk of the control group.

effects and thus ends up with a similar potential effect estimate. UNSCEAR [18] states clearly that there is no doubt about the existence of hereditary effects, but that epidemiological evidence can be produced only with a study including several generations and large numbers of people (due to the relatively small relative risk). To reflect the high uncertainty in this transformation from animal data to humans, we assume a squared geometric standard deviation of 5.

5. Damage analysis

5.1. The DALY concept

The previous section addressed the stochastic effects of low level radiation in terms of fatal and nonfatal cancer and hereditary effects. This section will quantify the damages occurring from these effects.

A fatal cancer case is preceeded by a period of disease or illness, which can be interpreted as damage to human health. The premature death shortens the life expectancy of the subject person; the number of lost years is interpreted again as damage to human health.

Damage to human health can therefore be written as:

$$\text{DALYs} = \text{YLD} + \text{YLL} \tag{1}$$

where DALYs is disability adjusted life years, YLD is years lived disabled, and YLL is years of life lost.

The DALY-concept, developed by Murray and Lopez [12] for the World Bank and the World Health Organisation, is based on great experience in health assessment. The involved assumptions and value-laden choices are discussed in general by Murray and Lopez [12] and are summarised from an LCA perspective by Hofstetter [14].

Two major choices are (a) an eventual discount rate for future health damages, and (b) an eventual age-weighting because society may weight differently one year lived by a child, one year lived by a retired person, or one year lived by a parent in the most productive phase of life. LCA in its current way of application does not contain any information relevant for discounting future damages and implicitly contains a 0% discount rate. Discounting of future impacts is a value judgement. The different opinions on it may be embedded in Cultural Theory (1). However, relevant information on discounting future damages is not readily available in a manner consistent for LCA integration. We therefore do not include any discounting. The age-weighting can be seen as a question of world-view. Here we distinguish two scenarios, one without discounting and without age-weighting (equal weights to each life year), which will be reported as (0,0), and one without discounting **but** age-weighting (0,1).

The calculation of YLD_m, the weighted years lived disabled per tissue

Table 4
Years lived disabled before death or recovery for different cancer sites with and without age weighting[a]

Tissue or organ	Average disability weights D_m (−)	Average age of onset a_m (a)	Average duration of disease L_m (a)	YLD_m (0,0) (a)	YLD_m (0,1) (a)
Bladder	0.087	67.2	4.7	0.41	0.29
Bone marrow[b]	0.060	58.5	3.8	0.23	0.20
Bone surface[c]	0.136	62.6	3.4	0.47	0.38
Breast	0.084	60.3	4.3	0.36	0.31
Colon	0.217	67.5	3.9	0.85	0.61
Liver	0.239	64.3	1.77	0.42	0.34
Lung	0.146	66.7	2.0	0.29	0.22
Oesophagus	0.217	66.2	1.8	0.39	0.30
Ovary	0.095	59.0	3.3	0.31	0.28
Skin	0.045	55.4	4.4	0.20	0.19
Stomach	0.217	66.6	3.0	0.65	0.48
Thyroid[c]	0.136	62.6	3.4	0.47	0.38
Remainder[c]	0.136	62.6	3.4	0.47	0.38

[a] Disability weights from Murray and Lopez [12, p. 414ff], age of onset and duration from Murray and Lopez [13, p. 541ff]; YLD calculated with equation 2 and 3.

[b] Approximated by lymphomas and multiple myleoma.

[c] Due to missing data, the average of all cancer sites is chosen.

or organ m, can be done for the case of no discounting and no age-weighting, case (0,0), by the simple multiplication of the disability weight D_m with the disease duration L_m:

$$\mathrm{YLD}_m(0,0) = D_m L_m \tag{2}$$

Disability weights for different disability types were derived from health expert panels and express the seriousness of harms, where 0 stands for absolute health and 1 for death.

For the age-weighted years lived disabled [$\mathrm{YLD}_m(0,1)$] a somewhat more complicated formula has to be used, according to Hofstetter [14, p. 185]:

$$\mathrm{YLD}_m(0,1) = D_m\left(\frac{Ce^{-\beta a_m}}{\beta^2}\{e^{-\beta L_m}[-\beta(L_m + a_m) - 1] + \beta a_m + 1\}\right) \tag{3}$$

where a_m is age of disease onset, β is a parameter to determine the shape of the age-weighting (here set to 0.04), and $C = 0.1658$, to adjust the average age-weighting to 1.

5.2. Carcinogenic effects

Table 4 summarises the results for calculation of the years lived disabled for tumour incidence. In established market economies, 90% of cancer patients are treated. The weighted average disability weights were taken from Murray and Lopez [12]. Due to lack of more detailed information, it is assumed that the calculated years lived disabled apply to fatal as well as to nonfatal cancer. This assumption is reasonable, because the period lived disabled before death due to cancer may be shorter, but the severity of the disability is higher.

In the case of fatal cancer, the years of life lost must be added to the calculated years lived disabled. As a reference for this calculation, age-specific life expectancies found in the Japanese female population are chosen [13]. Murray and Lopez [12] report the number of cancer death per age-class, sex, and cancer site in established market economies. From this information it is possible to calculate the average years of life lost, with and without age-weighting (Table 5). This procedure is implicitly based on the assumption that the radiation-induced-cancer cases occur at the same age-pattern as for all causes (smoking, radiation, etc.) together.

The derived values in Tables 4 and 5 are assumed to be best guesses, with an uncertainty range that can be described by a lognormal distribution. For the years lived disabled (YLD) there is some uncertainty in the disability weight and considerable uncertainty in the duration of disease. A squared geometric standard deviation of 3 reflects this high uncertainty. The years of life lost (YLL) data suffer from the uncertainty in the estimate for the life expectancy without the premature death. The data were derived from only five age classes, as distinguished by Murray and Lopez [13]. However, a comparison with similar data based on more detailed statistics by Hofstetter [14, p. 253] shows good agreement. Land and Sinclair [29, p. 55] used lifetables and age-specific cancer mortality rates for five countries, differentiated for 18 age classes. The calculated years of expected life lost averaged for the five countries are 10 to 30% lower than the data in Table 5. This can be explained in part by the different standard life expectancy tables used in the two studies. The higher value for bone marrow reflects the more detailed data in Land and Sinclair [29]; it was chosen here for the further calculations without adjustment. An average years of life lost per cancer death of 17 years was confirmed by Upton [30, p. 25]. A squared geometric standard deviation of 2 represents the uncertainty sufficiently. The share of the YLD within total DALY-value is between 1 and 7% only. Therefore, it is assumed that the DALYs are lognormally distributed again with a squared geometric standard deviation of 2.

An additional effect of 0.01 cases of severe hereditary effects per man.Sv was reported in the previous section. The translation of this expected effect into damages is much more complex, because (a) these damages do not occur during the lifetime of the exposed person and (b) the effect factor is based on animal tests, which makes it difficult to predict the resulting human health damages. Dreicer et al. [6] assume that severe hereditary effects result either in immediate death or a severely impaired life. ICRP [25] states that 15% of the potential cases of severe hereditary effects occur in the first generation, 12% in the second, and 73% some time thereafter.

Table 5
Years lived disabled, years life lost, and disability adjusted life years for different cancer sites with and without age weighting per case of fatal cancer

Tissue or organ	YLD_m (0,0) (a/fatal cancer)	YLL_m (0,0) (a/fatal cancer)	$DALYs_m$ (0,0) (a/fatal cancer)	YLD_m (0,1) (a/fatal cancer)	YLL_m (0,1) (a/fatal cancer)	$DALYs_m$ (0,1) (a/fatal cancer)
Bladder	0.41	12.1	12.5	0.29	7.0	7.3
Bone marrow[a]	0.23	16.8/30.9[c]	17.0/31.1[c]	0.20	11.7/25[d]	11.9/25.2[d]
Bone surface[b]	0.47	17.0	17.5	0.38	11.7	12.1
Breast	0.36	20.2	20.6	0.31	14.3	14.6
Colon	0.85	14.5	15.3	0.61	9.1	9.7
Liver	0.42	15.8	16.2	0.34	10.6	10.9
Lung	0.29	15.6	15.9	0.22	10.4	10.6
Oesophagus	0.39	15.5	15.9	0.30	10.3	10.6
Ovary	0.31	18.2	18.5	0.28	12.2	12.5
Skin	0.20	20.0	20.2	0.19	15.2	15.4
Stomach	0.65	18.1	18.8	0.48	12.7	13.2
Thyroid[b]	0.47	17.0	17.5	0.38	11.7	12.1
Remainder[b]	0.47	17.0	17.5	0.38	11.7	12.1

[a] Approximated by lymphomas and multiple myeloma.
[b] Due to missing data the average of all cancer sites is chosen.
[c] More specific data from Land & Sinclair [29, p.55].
[d] Estimated from more specific data in Land & Sinclair [29, p.55].

As we do not consider any discounting within LCA sensu stricto, this distinction does not matter. Sankaranarayanan [28] confirms the high uncertainty and the present impossibility of demonstrating statistically significant differences in hereditary effects between children of A-bomb survivors in Japan and a control group.

Murray and Lopez [12] suggest disability weights of about 0.2 to 0.6 for serious disabilities including genetic defects. In this uncertain environment, it is assumed that half of the cases will result in immediate death and half will live with a disability, weighted as an average at 0.4. Using the standardised Japanese life expectancy tables, this results in 57 DALYs(0,0) per case without age-weighting and in 61 DALYs(0,1) per case with age-weighting. The correct value in any case will be close to a full loss of the life expectancy, because persons with hereditary effects will die earlier than the unaffected population. Therefore, the relative uncertainty is rather small, and the squared geometric standard deviation is estimated to be 1.5. ICRP [25] allocates severe genetic disorders to the gonads and assigns a period of life lost of 20 years. However, both this assignment and the years of life lost, disregard death in childhood and disabilities before death. We therefore did not follow the ICRP allocations.

If the DALYs from carcinogenic and hereditary effects are calculated and compared, it can be seen that the high number of DALYs per case makes severe hereditary effects about equal in important as are all fatal and nonfatal cancer cases together[10]. Therefore, improvements in the effect and damage analyses for hereditary effects will increase the quality of the assessment significantly.

6. Damage assessment scenarios and cultural theory

6.1. Value-laden assumptions

Each part of the assessment described in the preceding sections involves uncertainty in the choice of models, statistical variation in the sample, and variability of the parameters. Some choices are even based on approximations or on opinions that are disputed among experts. The technical uncertainties are addressed by assuming hypothetical probability distributions for each step in the impact pathway. It is assumed that the factors causing the uncertainty are independent. The resulting distributions for the damage factors, shown in Table 6, are still approximately lognormal. The squared

[10] Egalitarian/hierarchical scenario: Cancer: 0.94 DALYs (0/0)/man.Sv; hereditary effects: 0.57 DALYs (0,0)/man.Sv. Individualist scenario: Cancer: 0.66 DALYs (0,1)/man.Sv; hereditary efects: 0.61 DALYs (0,1)/man.Sv. Hence, hereditary effects contribute about 37% and 48% to the total DALYs/man.Sv for the egalitarian/hierarchical and the individualist scenarios, respectively.

geometric standard deviation for the 95% confidence interval is also given in Table 6.

However, there is more than just technical uncertainty. The following assumptions described in the previous sections are value-laden:

(a) the time horizon for the integration of exposure to people,
(b) the area to be considered in the fate and exposure analyses,
(c) the necessary evidence for an association between low level radiation and cancer cases,
(d) the extrapolation model to be used for estimating health effects at very low doses,
(e) the dose and dose-rate effectiveness factor to be applied if linear no-threshold extrapolation methods are used, and
(f) the assumptions in the concept of disability adjusted life years (DALYs).

For (b), the LCA concept considers world-wide effects of product systems [2] by definition. For (c), different DDREF might be applied, because ICRP makes no clear statement whether a DDREF-factor of 2 (as recommended by ICRP and as chosen in this paper) is valid for public protection when extrapolating to very low doses. Nevertheless it is assumed in this paper that the DDREF-factor 2 is also valid for an extrapolation to very low doses[11]. Assumptions (d) and (e) are discussed widely in the international advisory boards for radiation protection. Decades of research have resulted in generally similar opinions. The remaining disagreement therefore may be treated as other technical uncertainties are and is included in the probability distributions mentioned above.

Relevant value-laden assumptions in the DALYs-concept (f) are based on the assumptions made in the general LCA sensu stricto. For instances, the lack of information about time period compels us to refrain from discounting future health states.

Thus remains the age-weighting in the DALY-concept (f), and the time horizon for the exposure integration (a). These two value judgements are dealt with in scenarios, rather than in a sensitivity analysis. Rotmans and Vries [31] and Hofstetter [14] suggest using cultural theory for the generation of value-compatible scenarios in integrated assessment and LCA, respectively.

The choice of emission factors in the inventory table may be value-laden as well. The long-term emission factor for radon-222 from abandoned mill tailings is highly dependent on assumptions about the future treatment of the tailings. In Section 7, results of a sensitivity analysis for the long-term

[11] This missing evidence might be taken into account by chosing different DDREF-factors according to different value-compatible scenarios (see Section 6.2).

Rn-222 emission factor in the mining and milling step of the nuclear fuel cycle are reported.

6.2. Cultural theory for value-compatible scenarios

Cultural theory describes five ways of life that are viable combinations of cultural biases and social relations [32, 33]. Cultural bias refers to shared values and beliefs. The social relations are described by the extent to which people are embedded in a group (group dimension) and the degree to which people are circumscribed by externally imposed prescriptions (grid dimension). Strong group and grid dimensions are labelled as *hierarchy*, strong group–weak grid as *egalitarism*, weak group but strong grid as *fatalism*, and weak on both dimensions as *individualism*. A fifth way of life is labelled *autonomy*, representing the fact that a few people have no social interactions at all.

These five ways of life can be interpreted as perspectives that are taken to view and manage the world. Three cultural perspectives are active in public decision making and interested in the use of LCA: individualists, hierarchists, and egalitarians. Cultural theorists have collected long lists of characteristics of each of these three perspectives (see [14, p. 55f]). This list can now be interpreted as a value backbone for a scenario generator. Its practical use is shown here.

In the light of cultural theory, the appropriate time horizon is a question of intergenerational responsibility and of the valuation of future needs and resources: Egalitarians would argue that exposure in the future (especially distant future) is at least as important as exposure today and would opt that society adjust its needs to limit the exposure of future generations. Individualists would argue that here and now counts, that future exposure is less important, and that in the case of unacceptable future exposure, technical solutions will be developed to limit exposure or effects (e.g., cancer drugs). Hierarchists consider the present and future as equally important and are optimistic concerning the means to limit exposure to effects.

An individualistic world view considers human beings to some extent as a production factor, with its highest market benefit between the age of 20 and 50. Egalitarians and hierarchists pay more attention to the social role of children and elderly people and defend the equal right of all individuals (see also [14, p. 188f]).

From this short discussion we conclude the following scenario assumptions:

- The egalitarian scenario assumes the longest time horizon, here restricted by the data to 100,000 years, and makes no age-weighting for the DALYs.
- The hierarchist scenario is in this case identical to the egalitarian scenario, due to the lack of more differentiated time horizon calculations in the fate and exposure analysis.

- The individualist scenario integrates the exposure for 100 years and applies age-weighting for the DALYs concept.

6.3. Cultural perspective-dependent damage factors

With these assumptions, we now may calculate damage factors by multiplying the exposure, effect, and DALYs factors as developed in Sections 3, 4, and 5 (see Table 6). The DALYs factors are compatible with the damage-oriented impact assessment method Eco-indicator 99 [10]. For impact-oriented characterisation methods like the CML approach [1,8], the figures are converted to uranium-235 air-equivalents by dividing the damage factors with the damage factor of airborne uranium-235.

The following general tendencies are observed:

- Emissions to the ocean lead to the lowest damage factors of a particular radionucleide. Emission to rivers and lakes and airborne emissions, respectively, show the highest damage factors, depending on the radionucleide emitted.
- Large differences in the cultural theory scenarios occur only for long-lived and globally dispersed radionucleides such as carbon-14, krypton-85, and iodine-129.
- For all other radionucleides, including tritium, the difference due to different cultural perspectives is 1.2 and caused entirely by differences in the damage assessment.

The damages per kBq emission occur about half as carcinogenic mortality and morbidity and half as hereditary effects. This is due to the larger damage per case caused by hereditary effects compared with the carcinogenic effects.

7. Example on the relation to other health damages

7.1. Introduction

The derived damage factors for radionucleide emissions that cause low-dose ionising radiation can now be compared with similar evaluations for chemicals causing respiratory and carcinogenic effects reported by Hofstetter [14]. Inventory tables for electricity production with nuclear power plants and with the Swiss and UCPTE power generating facilities are taken from [5]. Data for electricity produced with PFBC[12] hard coal and GCC[13] natural gas power plants are taken from Frischknecht [9]. The comparison is made from the egalitarian/hierarchist scenario showing low (2.5%), mean, and high (97.5%) values and for the individualist scenario (mean values only).

[12] PFBC: pressurised fluidised bed combustion; net efficiency: 47%.

[13] GCC: gas combined cycle; net efficiency: 57%.

Table 6
Damage factors and characterisation factors for two scenarios following three world views combining the data given in Tables 1 to 5[a]

	Damage factor per pollutant				Characterisation factor per pollutant	
	Egalitarian/Hierarchist		Individualist		Egalitarian/Hierarchist	Individualist
	DALYs(0,0)/kBq	σ_g^2	DALYs(0,1)/kBq	σ_g^2	kBq U-235 air-equiv.	kBq U-235 air-equiv.
Emitted to air						
C-14	2.1×10^{-7}	15	1.6×10^{-8}	15	1.00×10^{1}	9.4×10^{-1}
Co-58	4.3×10^{-10}	15	3.6×10^{-10}	15	2.0×10^{-2}	2.1×10^{-2}
Co-60	1.6×10^{-8}	15	1.4×10^{-8}	15	7.6×10^{-1}	8.2×10^{-1}
Cs-134	1.2×10^{-8}	15	1.0×10^{-8}	15	5.7×10^{-1}	5.9×10^{-1}
Cs-137	1.3×10^{-8}	15	1.1×10^{-8}	15	6.2×10^{-1}	6.5×10^{-1}
H-3	1.4×10^{-11}	28	1.2×10^{-11}	28	6.7×10^{-4}	7.1×10^{-4}
I-129	9.4×10^{-7}	65	2.5×10^{-7}	28	4.5×10^{1}	1.47×10^{1}
I-131	1.6×10^{-10}	15	1.3×10^{-10}	15	7.6×10^{-3}	7.6×10^{-3}
I-133	9.4×10^{-12}	15	7.9×10^{-12}	15	4.5×10^{-4}	4.6×10^{-4}
Kr-85	1.4×10^{-13}	28	1.2×10^{-13}	28	6.7×10^{-6}	7.1×10^{-6}
Pb-210	1.5×10^{-9}	15	1.3×10^{-9}	15	7.1×10^{-2}	7.6×10^{-2}
Po-210	1.5×10^{-9}	15	1.3×10^{-9}	15	7.1×10^{-2}	7.6×10^{-2}
Pu alpha	8.3×10^{-8}	15	7.0×10^{-8}	15	4.0	4.1
Pu-238	6.7×10^{-8}	15	5.7×10^{-8}	15	3.2	3.4
Ra-226	9.1×10^{-10}	15	7.6×10^{-10}	15	4.3×10^{-2}	4.5×10^{-2}
Rn-222	2.4×10^{-11}	15	2.0×10^{-11}	15	1.14×10^{-3}	1.18×10^{-3}
Th-230	4.5×10^{-8}	15	3.8×10^{-8}	15	2.1	2.2
U-234	9.7×10^{-8}	15	8.2×10^{-8}	15	4.6	4.8
U-235	2.1×10^{-8}	15	1.7×10^{-8}	15	1.0	1.0
U-238	8.2×10^{-9}	15	6.9×10^{-9}	15	3.9×10^{-1}	4.1×10^{-1}
Xe-133	1.4×10^{-13}	15	1.2×10^{-13}	15	6.7×10^{-6}	7.1×10^{-6}

(*continued*)

Table 6
Continued

	Damage factor per pollutant				Characterisation factor per pollutant	
	Egalitarian/Hierarchist		Individualist		Egalitarian/Hierarchist	Individualist
	DALYs(0,0)/kBq	σ_g^2	DALYs(0,1)/kBq	σ_g^2	kBq U-235 air-equiv.	kBq U-235 air-equiv.
Emitted to rivers and lakes						
Ag-110m	5.1×10^{-10}	15	4.2×10^{-10}	15	2.4×10^{-2}	2.5×10^{-2}
Co-58	4.1×10^{-11}	15	3.4×10^{-11}	15	2.0×10^{-3}	2.0×10^{-3}
Co-60	4.4×10^{-8}	15	3.7×10^{-8}	15	2.1	2.2
Cs-134	1.4×10^{-7}	15	1.2×10^{-7}	15	6.7	7.1
Cs-137	1.7×10^{-7}	15	1.4×10^{-7}	15	8.1	8.2
H-3	4.5×10^{-13}	28	3.8×10^{-13}	28	2.1×10^{-5}	2.2×10^{-5}
I-131	5.0×10^{-10}	15	4.2×10^{-10}	15	2.4×10^{-2}	2.5×10^{-2}
Mn-54	3.1×10^{-10}	15	2.6×10^{-10}	15	1.48×10^{-2}	1.53×10^{-2}
Ra-226	1.3×10^{-10}	15	1.1×10^{-10}	15	6.2×10^{-3}	6.5×10^{-3}
Sb-124	8.2×10^{-10}	15	6.9×10^{-10}	15	3.9×10^{-2}	4.1×10^{-2}
U-234	2.4×10^{-9}	15	2.0×10^{-9}	15	1.14×10^{-1}	1.18×10^{-1}
U-235	2.3×10^{-9}	15	2.0×10^{-9}	15	1.10×10^{-1}	1.18×10^{-1}
U-238	2.3×10^{-9}	15	1.9×10^{-9}	15	1.10×10^{-1}	1.12×10^{-1}

(*continued*)

Table 6
Continued

	Damage factor per pollutant				Characterisation factor per pollutant	
	Egalitarian/Hierarchist		Individualist		Egalitarian/Hierarchist	Individualist
	DALYs(0,0)/kBq	σ_g^2	DALYs(0,1)/kBq	σ_g^2	kBq U-235 air-equiv.	kBq U-235 air-equiv.
Emitted to the ocean						
Am-241	3.1×10^{-8}	15	2.6×10^{-8}	15	1.48	1.53
C-14	1.2×10^{-9}	15	9.9×10^{-10}	15	5.7×10^{-2}	5.8×10^{-2}
Cm alpha	5.7×10^{-8}	15	4.8×10^{-8}	15	2.7	2.8
Co-60	3.9×10^{-10}	15	3.3×10^{-10}	15	1.86×10^{-2}	1.94×10^{-2}
Cs-134	7.9×10^{-11}	15	6.6×10^{-11}	15	3.8×10^{-3}	3.9×10^{-3}
Cs-137	7.9×10^{-11}	15	6.7×10^{-11}	15	3.8×10^{-3}	3.9×10^{-3}
H-3	6.9×10^{-14}	28	5.8×10^{-14}	28	3.3×10^{-6}	3.4×10^{-6}
I-129	1.0×10^{-7}	65	1.9×10^{-8}	28	4.8	1.12
Pu alpha	7.4×10^{-9}	15	6.2×10^{-9}	15	3.5×10^{-1}	3.6×10^{-1}
Ru-106	1.4×10^{-10}	15	1.2×10^{-10}	15	6.7×10^{-3}	7.1×10^{-3}
Sb-125	1.5×10^{-11}	15	1.2×10^{-11}	15	7.1×10^{-4}	7.1×10^{-4}
Sr-90	4.0×10^{-12}	15	3.4×10^{-12}	15	1.90×10^{-4}	2.0×10^{-4}
U-234	2.3×10^{-11}	15	1.9×10^{-11}	15	1.10×10^{-3}	1.12×10^{-3}
U-235	2.5×10^{-11}	15	2.1×10^{-11}	15	1.19×10^{-3}	1.24×10^{-3}
U-238	2.3×10^{-11}	15	2.0×10^{-11}	15	1.10×10^{-3}	1.18×10^{-3}

[a] The σ_g^2 stands for the squared geometric standard deviation. Dividing and multiplying the best estimate by σ_g^2 spans the 95% confidence interval.

Additionally, the results of a sensitivity analysis for long-term Rn-222 emissions show the influence of cultural theory in the inventory analysis[14].

In the sensitivity analysis, it is assumed that abandoned mill tailings will not be covered. The long-term Rn-222 emission factor rises to 2×10^9 kBq per kg uranium instead of 6×10^7 kBq per kg uranium, as assumed for covered mill tailings (cf. [5, VII:48]). The former may be attributed to an egalitarian, the latter to an individualist scenario.

The following evaluation does not yet include damages due to climate change and toxic effects not covered so far. Therefore, the share of the radiation damage will be lower when the assessment becomes more complete. Furthermore, uncertainties in the results of the inventory tables are not considered either. Hence, low and high values are caused only by uncertainty in the impact assessment.

7.2. Egalitarian scenario, uncertainty analysis, and sensitivity analysis

The comparison of the mean values reveals the importance of the carcinogenic and hereditary effects from radionucleide emissions by the production of nuclear electricity (Table 7).

For a nuclear power plant and its nuclear fuel cycle, mean health damages due to ionising radiation are nearly three times the damages from chemical carcinogenesis and respiratory effects together. The major contributors to the total health damages are the long-term emissions of Rn-222 released during 80,000 years from abandoned mill tailings (46% of total health damages) and the exposure to C-14 released during power production (7%). Iodine-129 releases to water and Kr-85 releases to air, both from reprocessing, contribute 0.5% and 0.2%, respectively.

For the average Swiss domestic production, based mainly on nuclear (38%) and hydro (59%, excluding pumping storage) power, the health damages due to radiation have a share of 47% and the respiratory effects account for 34% (mean values).

This picture changes for the average electricity production in the UCPTE countries, where about 37% still stems from nuclear power plants but fossil power plants contribute to more than 45% of the supply. About 4% of the assessed total damage (mean value) stems from radiation and respiratory effects dominate largely the human health damages (77%).

PFBC coal power plants release radionucleides as well. Therefore, Table 7 lists health damages for hard coal—and for reasons of comparability with GCC gas power plants. The share of ionising radiation lies between one

[14] Data on emission factors for long-term radon-222 releases from mill tailings vary considerably. SENES [21] report values between 0 and 2.2×10^7 kBq/kg natural uranium for 8 sites responsible for 67% of the world production in 1997 with different decommissioning plans for their mill tailings. ESU [5, part VII:48f] reports values between 1.3×10^7 and 1.3×10^8 kBq/kg natural uranium for covered and between 1.7×10^8 and 1.9×10^{10} kBq/kg natural uranium for uncovered abandoned mill tailings.

Table 7
Low (2.5%), mean, and high (97.5%) value for human health damages due to respiratory, carcinogenic, and hereditary effects for the egalitarian scenario for three different electricity production systems and two average power plant mixes

Electricity produced by		Human health damages, DALYs(0,0)/TJ				Contributing pollutants with a share >1% (relevance order)
		Due to respiratory effects	Due to carcinogenic effects (chemicals)	Due to carcinogenic and hereditary effects (low dose ionising radiation)	Total[b]	
Average of Swiss nuclear power plants	low	0.00018	6×10^{-6}	0.0008	0.001	Rn-222, C-14, SO_x, particulates, NO_x
	mean	0.00358	0.00068	0.0122	0.0165	Rn-222, C-14, SO_x, particulates, NO_x, Ni, Cr,
	high	0.0581	0.146	0.192	0.396	Rn-222, Ni, Cr, C-14, particulates, SO_x, NO_x, I-129 (water), As, Ni (water)
Average of Swiss nuclear power plants	Sensitivity Rn-222[a]	0.00358	0.00068	0.368	0.372	Rn-222
Average of Swiss power plants	low	0.0002	0.00001	0.00031	0.00048	Rn-222, SO_x, particulates, C-14, NO_x, Ni
	mean	0.0033	0.00186	0.00471	0.00983	Rn-222, Ni, SO_x, NO_x, particulates, C-14, Cr
	high	0.052	0.422	0.074	0.548	Ni, Rn-222, Cr, SO_x, particulates, NO_x, C-14
Average of UCPTE power plants	low	0.00441	0.00016	0.00029	0.00485	SO_x, particulates, NO_x, Rn-222, Ni
	mean	0.0862	0.0219	0.0044	0.112	SO_x, NO_x, particulates, Ni, Cr, Rn-222, As
	high	1.38	4.78	0.0686	6.23	Ni, Cr, SO_x, particulates, NO_x, As, Cd, Ni (water)
PFBC hard coal power plant	low	0.00358	0.00011	1.1×10^{-5}	0.0037	particulates, SO_x, NO_x
	mean	0.0751	0.0113	0.00016	0.0866	particulates, NO_x, SO_x, Ni, Cr, Ni (water)
	high	1.22	2.42	0.0026	3.65	Ni, particulates, Cr, NO_x, Ni (water), SO_x, As,
GCC gas power plant	low	0.000370	8.7×10^{-6}	6.4×10^{-7}	0.000379	NO_x, SO_x, particulates
	mean	0.0106	0.00088	9.8×10^{-6}	0.0115	NO_x, SO_x, particulates, Ni, Cr
	high	0.147	0.186	0.00015	0.334	Ni, NO_x, Cr, SO_x, particulates, Cd, CO

[a] Assuming uncovered mill tailings with an emission factor of $2 \cdot 10^9$ kBq Rn-222/kg Uranium instead of treated (covered) mill tailings with an emission factor of 6×10^7 kBq Rn-222/kg Uranium (cf. ESU [5, part VII, pp.48–50].

[b] The total low and high values do not show exactly the 2.5% and 97.5% percentile. The "true" confidence interval will be smaller according to the level of independence of factors causing the uncertainty. A detailed assessment of the dependence of uncertainty factors was not possible within the scope of this work.

and two per thousand and is about 17 times smaller in absolute terms for gas compared to hard coal power. In addition to Rn-222 the PFBC coal power plant releases larger quantities of the isotope Rn-220 as well, but Rn-220 is not assessed here. The damage caused by PFBC coal power plant releases is therefore higher than calculated in Table 7. The respiratory effects dominate the health damages for hard coal (87%) and gas (92%) power plants.

The share of contribution of individual pollutants varies between the low, mean, and high values. Chromium VI and nickel emissions to air tend to contribute a much larger share in the mean and high value cases, whereas the share of contribution of SO_x and particulates decreases from low to high value cases. This reflects the fact that the uncertainty of damage factors of nickel and chromium VI are larger than those of SO_x and particulates.

The ranking of the three options Swiss nuclear power plants, average Swiss power plant mix, and GCC gas power plant changes between low, mean, and high values, but their total health damage figures are significantly lower, than the damage figures of the PFBC coal power plant and the average UCPTE power plant mix.

In the sensitivity analysis, assuming higher long-term Rn-222 emissions from uncovered abandoned mill tailings, the Rn-222 emissions become even more dominant in the damage assessment of Swiss nuclear power. Nearly 99% stem from low dose ionising radiation in this case. The mean value of total health damages is higher by a factor of 4 compared with the PFBC coal power plant and by a factor of more than 30 compared with the GCC natural gas power plant.

7.3. Individualist scenario

For the individualist scenario, total health damages are smaller by a factor of between 2 and 6 compared with the mean values of the egalitarian/hierarchist scenario (see Table 8). Damages due to long-term Rn-222 emissions are considered only for the first 100 years and are therefore much less important. Health effects due to low dose radiation are far less important and contribute between 13% (Swiss nuclear power plants) and 0.01% (PFBC coal and GCC natural gas power plant) to total health damages. Respiratory effects contribute most to the total health damages, with values between 56% (average of Swiss power plants) and 84% (PFBC coal power plant).

The ranking of mean human health damages of Swiss nuclear power plant, GCC gas power plant, and average Swiss power plant mix is different from the ranking under the egalitarian scenario, but the differences are rather small. The distinct difference between these three supply systems and PFBC coal power plant and the average UCPTE power plant mix remains.

NO_x-emissions to air (except for the GCC gas power plant) and water pollutants (except nickel for the PFBC coal power plant) disappear from

Table 8
Mean values for human health damages due to respiratory, carcinogenic, and hereditary effects for the individualist scenario for three different electricity production systems and two average power plant mixes

Human health damages DALYs(0,1)/TJ					
Electricity produced by	Due to respiratory effects	Due to carcinogenic effects (chemicals)	Due to carcinogenic and hereditary effects (low dose ionising radiation)	Total	Contributing pollutants with a share >1% (relevance order)
Average of Swiss nuclear power plants	0.00191	0.00043	0.00035	0.0027	SO_x, particulates, Ni, Cr, C-14, Rn-222, Kr-85, As
Average of Swiss power plants	0.0017	0.00120	0.00014	0.003	Ni, SO_x, particulates, Cr, C-14, Rn-222
Average of UCPTE power plants	0.047	0.0139	0.0001	0.0610	SO_x, particulates, Ni, Cr, As
PFBC hard coal power plant	0.0389	0.0071	4.0×10^{-6}	0.0460	Particulates, SO_x, Ni, Cr, Ni (water)
GCC gas power plant	0.00193	0.00053	2.4×10^{-7}	0.00246	SO_x, particulates, Ni, Cr, NO_x, NMVOC

the list of significant contributing pollutants. On the other hand, krypton-85 and NMVOC contribute in this scenario more than 1% to the total health damages of the Swiss nuclear and of the GCC gas power plant, respectively. In general, however, the major part of pollutants relevant in the egalitarian scenario is also relevant in the individualist scenario.

8. Discussion of the assessment method and conclusions

In this paper, a new approach for the damage assessment of low dose ionising radiation has been introduced. It enables the inclusion of radiation damages to human health into life cycle assessment in general and into the Eco-indicator 99 [10] in particular. A generic set of exposure factors has been derived from the results of site-specific modelling by considering each site's contribution to the total emissions of the entire nuclear fuel cycle (from mining to reprocessing) and by differentiating exposure factors for liquid releases to the ocean and to rivers and lakes. The concept of disability adjusted life years (DALY) as introduced by Murray and Lopez [12,13] has been used in the damage analysis.

In the impact pathways used for the damage assessment of radionucleides, several model assumptions are necessary that substantially influence the outcome of the results. To model value-laden assumptions, two value-compatible scenarios (egalitarian and individualist) based on cultural theory have been introduced concerning the time horizon for the integration of exposure and the age weighting in the DALY concept. With a sensitivity analysis for airborne Rn-222 emissions from abandoned mill tailings, cultural theory has been applied to inventory analysis.

The following conclusions—both methodological and relevant to LCA—can be drawn.

- The assessment of human health damages due to low dose ionising radiation relies on a combination of site-specific, site-dependent, and generic modelling, which results in damage estimates more accurate than those in present LCA practice.
- With the help of the DALY concept, morbidity is considered more accurately than with to the approaches described in ICRP [25], where lethality factors are used to aggregate nonfatal and fatal cancer and the severity factors of hereditary effects are roughly estimated.
- Cultural theory helps represent value-laden assumptions (in particular those concerning the time horizon relevant for the fate and exposure analyses).
- For long-lived, globally dispersed radionucleides, the egalitarian damage factors are higher by a factor of between 4 and 13 as compared with the individualist damage factors, due mainly to the much longer time horizon considered in the egalitarian scenario (100,000 years, compared with 100 years in the individualist scenario). For all other radionucleides, including tritium, the egalitarian damage factors are 20% higher.
- The new health damage factors for ionising radiation will significantly influence the contribution analysis of LCAs for nuclear energy production, because—depending on the cultural theory scenario—low dose ionising radiation over long periods of time may become its major source of damage to human health.
- In the assessment of nuclear electricity supply systems, long-term airborne Rn-222 releases from mill tailings are the most important, but also the most uncertain, emissions. This is due to the degree of planned covering and protection against erosion of abandoned mill tailings and the long period (several ten thousands of years) of increased emissions.
- In the individualist scenario and in the egalitarian scenario, assuming a low Rn-222 emission factor from covered abandoned mill tailings, additional human health damages due to low dose ionising radiation

will have limited influence on the overall assessment for most national electricity grids.

- In cases where nuclear fuel cycles rely on milling sites with uncovered and unprotected mill tailings, the egalitarian health damage factors for radionucleides will significantly influence the total health damages of national power generating facilities. However, as human health damages caused by other toxic substances and due to climate change will be included, this influence will be diminished.
- Radionucleides other than radon-222 (long-term emissions from mill tailings only) and carbon-14 contribute little to total human health damages of the electricity supply systems analysed, regardless of the cultural theory scenario chosen.
- Uncertainty figures for damages due to radionucleides are based mainly on qualitative information and assumptions. Their uncertainty is generally lower than the uncertainty of damages due to chemicals with carcinogenic effects (e.g., nickel, chromium VI) but comparable to or higher than the uncertainty of damages due to pollutants causing respiratory effects.

The impact pathway and damage factors assigned to the 31 radionucleides considered in this paper correspond to the typical situation for Western European nuclear power supply. It must be emphasised that this assessment does not include human health damages due to ionising radiation released by severe accidents, nor by long-term underground waste storage facilities. Emissions stemming from other sources (e.g., coal power plants) can be evaluated with the same coefficients if the typical situation (population density, meteorology, etc.) is analogous. However, for environmental impact assessments of particular emission sources, site-specific data as applied by Dreicer et al. [6] are preferred. For other cases, new calculations according to the prevalent pathways need to be done. Furthermore, the damage factors given in this paper are expected to change due to future improvements of the models (although we expect the main structure of the proposed approach to be maintainable).

Although several assumptions have been required to bridge data gaps, this paper provides a consistent approach for the assessment of radionucleide releases in life cycle assessments. With this approach, operationalised impact assessment methods become more complete for the assessment of electricity supply systems, which often are major and/or disputed contributors to cumulative emissions of products and services.

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References

[1] Heijungs R, Guinée, Huppes G, Lankreijer RM, Udo de Haes HA, Wegener Sleeswijk A, Ansems AMM, Eggels PG, van Duin R, de Goede HP. Environmental Life Cycle Assessment of Products; Backgrounds, Report 9267. Leiden: CML, 1992.
[2] SETAC. Guidelines for Life-Cycle Assessment, a Code of Practice. Brussels: Society of Environmental Toxicology and Chemistry (SETAC), 1993.
[3] Heijungs R. Life Cycle Impact Assessment—A brief survey with some ideas on radiation. Paper presented at the Technical Committee meeting on Development and Use of Environmental Impact Indicators for Comparative Risk Assessment of Different Energy Sources, IAEA, Vienna, 3–6 May. Vienna: IAEA, 1994.
[4] Frischknecht R, Hofstetter P, Knoepfel I (ETH Zürich), Dones R, Zollinger E (PSI Villigen). Ökoinventare für Energiesysteme, Grundeagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz, 1st ed. Zürich: Gruppe Energie–Stoffe–Umwelt, ETH Zürich, Sektion Ganzheitliche Systemanalysen, PSI Villigen, 1994.
[5] Frischnecht R (final editor), Bollens U, Bosshart S, Ciot M, Ciseri L, Doka G, Hischier R, Martin A (ETH Zürich), Dones R, Gantner U (PSI Villigen). Ökoinventare von Energiesystemen, Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz, 3rd ed. Zürich: Gruppe Energie–Stoffe–Umwelt, ETH Zürich, Sektion Ganzheitliche Systemanalysen, PSI Villigen, 1996.
[6] Dreicer M, Tort V, Manen P. ExternE, Externalities of Energy, Vol. 5. Nuclear, Centre d'étude sur l'Evaluation de la Protection dans le domaine Nucléaire (CEPN), edited by the European Commission DGXII, Science, Research and Development JOULE, Luxembourg, 1995.
[7] Martin A. Radioaktive Emissionen in Ökobilanzen. Diploma thesis. Energy Systems Laboratory, ETH Zürich, 1995.
[8] Heijungs R, Guinée JB, Huppes G, Lankreijer RM, Udo de Haes HA, Wegener Sleeswijk A, Ansems AMM, Eggels PG, van Duin R, de Goede HP. Environmental Life Cycle Assessment of Products; Guide; Report 9266. Leiden: CML, 1992.
[9] Frischknecht R. Life Cycle Inventory Analysis for Decision-Making; Scope-dependent Inventory System Models and Context-specific Joint Product Allocation. ETH Dissertation Nr. 12599. ETH Zürich, 1998.
[10] Goedkoop M, Hofstetter P, Müller-Wenk R, Spriemsma R. The Eco-indicator 98 explained. Int J LCA 1998;3(6):352–60.
[11] Solberg-Johansen B, Clift R, Jeapes A. Irradiating the environment. Int J LCA 1997; 2(1):16–9.
[12] Murray C, Lopez A. The Global Burden of Disease. Boston, MA: WHO, World Bank, and Harvard School of Public Health, 1996.
[13] Murray C, Lopez A. Global Health Statistics. Boston, MA: WHO, World Bank, and Harvard School of Public Health, 1996.
[14] Hofstetter P. Perspectives in Life Cycle Impact Assessment. A Structured Approach to Combine Models of the Technosphere, Ecosphere and Valuesphere. Boston/Dordrecht/London: Kluwer Academic Publishers, 1998.
[15] IAEA editor. The radiological impact of radionucleides dispersed on a regional and global scale: Methods for assessment and their application. Technical Reports Series 250. Vienna: IAEA, 1985.
[16] United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR),

editor. Ionizing Radiation: Sources and Biological Effects. UNSCEAR 1982 Report to the General Assembly, with Annexes. New York: United Nations, 1982.
[17] United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), editor. Sources, Effects and Risks of Ionizing Radiation. UNSCEAR 1988 Report to the General Assembly, with Annexes. New York: United Nations, 1988.
[18] United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), editor. Sources and Effects of Ionizing Radiation. UNSCEAR 1993 Report to the General Assembly, with Scientific Annexes. New York: United Nations, 1993.
[19] Tort V. Assessment of the Radiological Impact of Storing Uranium Ore Tailings; A Sensitivity Study. Fontenay-aux-Roses, France: CEPN, 1998.
[20] Schneider T, Daroussin JL. Comparative Assessment of Radon Impacts Associated with Mill Tailings Storages. Paper presented at Southport '99, Sixth International Symposium. Achievements & Challenges: Advancing Radiation Protection into the 21st century. London: Society for Radiological Protection, 1999.
[21] SENES Consultants Limited. Long term Population Dose due to Radon (Rn-222) Released From Uranium Mill Tailings. The Uranium Institute, London. 1998.
[22] Stanners D, Bourdeau Ph, editors. Europe's Environment: The Dobris Assessment. Copenhagen: European Environment Agency, 1995.
[23] Bundesamt für Gesundheitswesen BAG. Umwelt-radioaktivität und Strahlendosen in der Schweiz 1994, Freiburg (CH): Abteilung Strahlenschutz, 1995.
[24] IAEA, WHO, editors. Low Doses of Ionizing Radiation: Biological Effects and Regulatory Control. Proceedings of a Conference, Seville, Spain, 17–21 November 1997. Vienna: IAEA and WHO, 1998.
[25] International Commission on Radiological Protection. Recommendations of the International Commission on Radiological Protection. Publication 60. Ann ICRP 1990;21(1–3).
[26] Ron E, Muirhead C. The carcinogenic effects of ionizing radiation: epidemiological evidence. In: Low Doses of Ionizing Radiation: Biological Effects and Regulatory Control. Proceedings of a Conference, Seville, Spain, 17–21 November 1997. Vienna: IAEA and WHO, 1998, pp. 165–80.
[27] Kaul A, Bennett BG. Sources, exposures and biological effects of ionizing radiation. In: Low Doses of Ionizing Radiation: Biological Effects and Regulatory Control. Proceedings of a Conference, Seville, Spain, 17–21 November 1997. Vienna: IAEA and WHO, 1998, pp. 23–36.
[28] Sankaranarayanan K. Ionizationg radiation and hereditary effects. In: Low Doses of Ionizing Radiation: Biological Effects and Regulatory Control. Proceedings of a Conference, Seville, Spain, 17–21 November 1997. Vienna: IAEA and WHO, 1998, pp. 145–58.
[29] Land CE, Sinclair WK. The relative contributions of different organ sites to the total cancer mortality associated with low-dose radiation exposure. Ann ICRP 1991;22(1):31–57.
[30] Upton AC. Risk estimates for carcinogenic effects of radiation. Ann ICRP 1991;22(1):1–30.
[31] Rotmans J, de Vries B. Perspectives on Global Change: The TARGETS Approach. Cambridge: Cambridge University Press, 1997.
[32] Douglas M. 1982. Cultural bias. In: Douglas M, editor, In the Active Voice. London: Routledge and Kegan Paul.
[33] Thompson M, Ellis R, Wildavsky A. Cultural theory. Boulder, CO: Westview Press, 1990.
[34] NCRP, editor. Influence of Dose and its Distribution in Time on Dose–Response relationships for Low-LET Radiations. NCRP Report 64. Bethesda, MD: National Council on Radiation Protection and Measurements, 1980.
[35] Goedkoop M. The Eco-Indicator 95. Final report. Amersfoort: PRé consultants, 1995.