nally contaminated. Local decontamination procedures took place, and no deterministic effects were reported in those workers.

Thyroid organ doses exceeding 10 Sv were estimated in two workers. This is a dose level that may result in deterministic effects such as thyroid dysfunction (i.e. hypothyroidism).

Approximately 2 000 workers were given stable iodine (potassium iodide [KI]) during the emergency response phase. Although most of these workers took fewer than 10 tablets, some took up to 87 tablets (self-administration). No allergic reactions were seen in this group. Three workers presented increased levels of thyroid-stimulating hormone (TSH) and decreased levels of thyroxin (T4), one of the two hormones produced by the thyroid. This thyroid dysfunction was transient and parameters returned to normal values once the KI administrations were stopped (see Box 7). The considerations given to the radiation-related risks of thyroid nodules in the general population (see section 5.4.1) are also applicable to workers.

Scenario 4 refers to an effective dose exceeding 500 mSv, but it is assumed to be as primarily due to internal exposure to ^{131}I, implying a high dose to the thyroid (>10 Sv) rather than to any other organ. Scenario 3 refers to an effective dose of 200 mSv and, in contrast to Scenario 4, no internal exposure due to ^{131}I is assumed here. If caesium intake was involved, the value of 200 mSv would be representative of organ doses, as caesium is distributed homogeneously in the body.

New evidence about the dose-response relationship for radiation-induced cataract indicates that the threshold doses can be around 500 mSv. In light of the most recent scientific evidence presented in Chapter 2, the HRA Expert Group concluded that there should be no expectation of cataract. It appears unlikely that the relatively small number

Box 7. Possible adverse effects of iodine thyroid blocking

During nuclear emergencies, public health protective actions may be implemented to prevent radiation exposure and associated health risks. The administration of stable iodine can prevent the uptake of radioactive iodine by the thyroid gland. When potassium iodide is taken before or shortly after exposure it can saturate the thyroid gland, thus reducing the dose and risk of thyroid cancer. Potassium iodide should be taken only when instructed by competent authorities, following dosage recommendations, especially for children.

Side effects may result from the administration of stable iodine for thyroid blocking, especially in iodine-deficient regions and in specific age groups or sub-populations. These side effects are not related to radiation and are therefore beyond the scope of this HRA. However, they are noted here as background information to support the interpretation of possible signs and symptoms that might otherwise be wrongly attributed to effects of radiation.

Reported side effects of iodine thyroid blocking include extra-thyroidal effects (e.g. digestive and skin reactions) and thyroid dysfunctions in connection with pre-existing thyroid disorders, such as auto-immune thyroiditis, Graves disease and nodular goiter. Hypothyroidism has been observed when stable iodine has been taken for longer than 1 week or for 10 days continuously. A large-scale survey in Poland after the Chernobyl accident (*131*) provided solid evidence based on over 10 million doses of stable iodine (potassium iodide [KI]) to children and around 7 million doses to adults. This topic was recently reviewed (*132*) and available studies did not reveal severe adverse reactions to KI in the general public. Persons with known iodine sensitivity as well as newborns and elderly people might be at higher risk.

of workers with doses exceeding 100 mSv[3] will show a statistically significant increase in the incidence of vision-impairing lens opacities at lower doses, although dosimetry data for the lens of the eye are not available.

As discussed in Annex F, section F.3, a threshold of 500 mSv would be compatible with epidemiological data on circulatory disease after radiation exposure, although work is in progress to determine whether such effects may be probabilistic rather than deterministic. It is concluded that there may be an increased risk of long-term circulatory disease, particularly among the workers whose doses exceeded 500 mSv[4]. It must be noted that the risks of circulatory disease among the LSS atomic bomb survivors were about three times lower than the risk for cancer (*56*) so if the cancer risk is small the circulatory disease risk is expected to be even smaller.

The considerations made above for heritable risks in the general population (section 5.4.1) are also applicable to workers.

3. This refers to effective doses or to the organ doses calculated by the HRA Expert Group. Doses to the lens of the eye were not available within the timeframe of the HRA Expert Group work.

4. Note that these are effective doses and are mainly due to thyroid doses.

6. Discussion

6.1 Factors influencing the radiation-related health risks

6.1.1 Age at exposure

Age at the time of irradiation is one of the most important biological variables influencing both short- and long-term effects of ionizing radiation. The influence of age-at-exposure on radiation-induced excess absolute risks (EAR) and excess relative risk (ERR) varies by cancer site (*105,5*). In the context of this HRA, the age-at-exposure becomes particularly relevant to the consideration of possible long-term health effects. Radiation exposure at a young age generally tends to result in higher risks than those resulting from exposure at older ages. Cancer risks in the unborn child are considered to be similar to those in 1-year-old infants (*12*) (see section 6.3.3).

The age pattern in radiation-related leukaemia incidence excess risk is a function of both age-at-exposure and time since exposure (see below). A very strong dependence on age-at-exposure has been found in the excess risk of leukaemia mortality, with exposure at 10 years of age associated with approximately 20-fold ERR/Gy in the period 5–20 years after exposure (*106*). The results of this HRA are consistent with this trend. Indeed, risks are higher in children and infants (LAR ratios 10 years: 20 years = 1.4, and 1 year: 20 years = 2.8).

Of the various cancer types, perhaps the most illustrative example is thyroid cancer, with a robust body of evidence of higher risks at younger ages and with weak evidence for an excess (or substantially lower risks) following radiation exposure in adulthood. The risk of thyroid cancer due to radioiodine exposure from the Chernobyl accident has shown strong age dependence. An ecological study based on a very large number of measurements of individual thyroid doses showed an ERR of 9.1/Gy for 0–4 years at exposure versus 3.4/Gy at 10–18 years (*107*). The HRA Expert Group took into consideration all the existing evidence to support their choice of the risk models to be applied to assess thyroid cancer risk in this HRA. The results reflected the presumptions that higher risks would be estimated in the youngest groups. Indeed, the LAR is much higher for 1-year-old infants and 10-year-old children compared with young adults. It is important to note that, on an absolute scale, the calculated excess risks (LAR) are small. However, in relative terms, the comparison with the low baseline thyroid cancer incidence rates in children translates into a large relative increase (LFR).

As mentioned above, epidemiological studies have indicated that the risk of thyroid cancer decreases with increasing age-at-exposure, with little or negligible risk apparent after the age of 20 years (*21,23*). This age dependence of thyroid cancer risks was also observed in the results of the HRA for workers. Indeed, much lower risks were found for 40-year-old and 60-year-old workers compared with 20-year-old workers.

Figure 21. Influence of the age-at-exposure on the lifetime attributable risk (LAR) of solid cancer

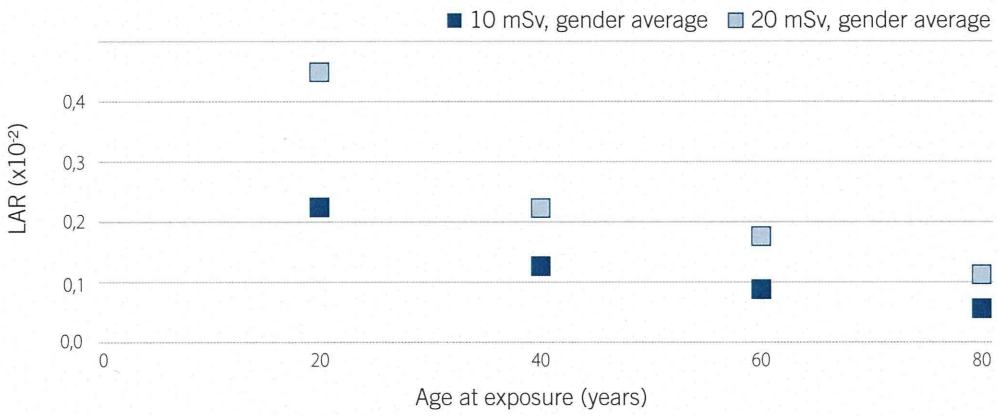
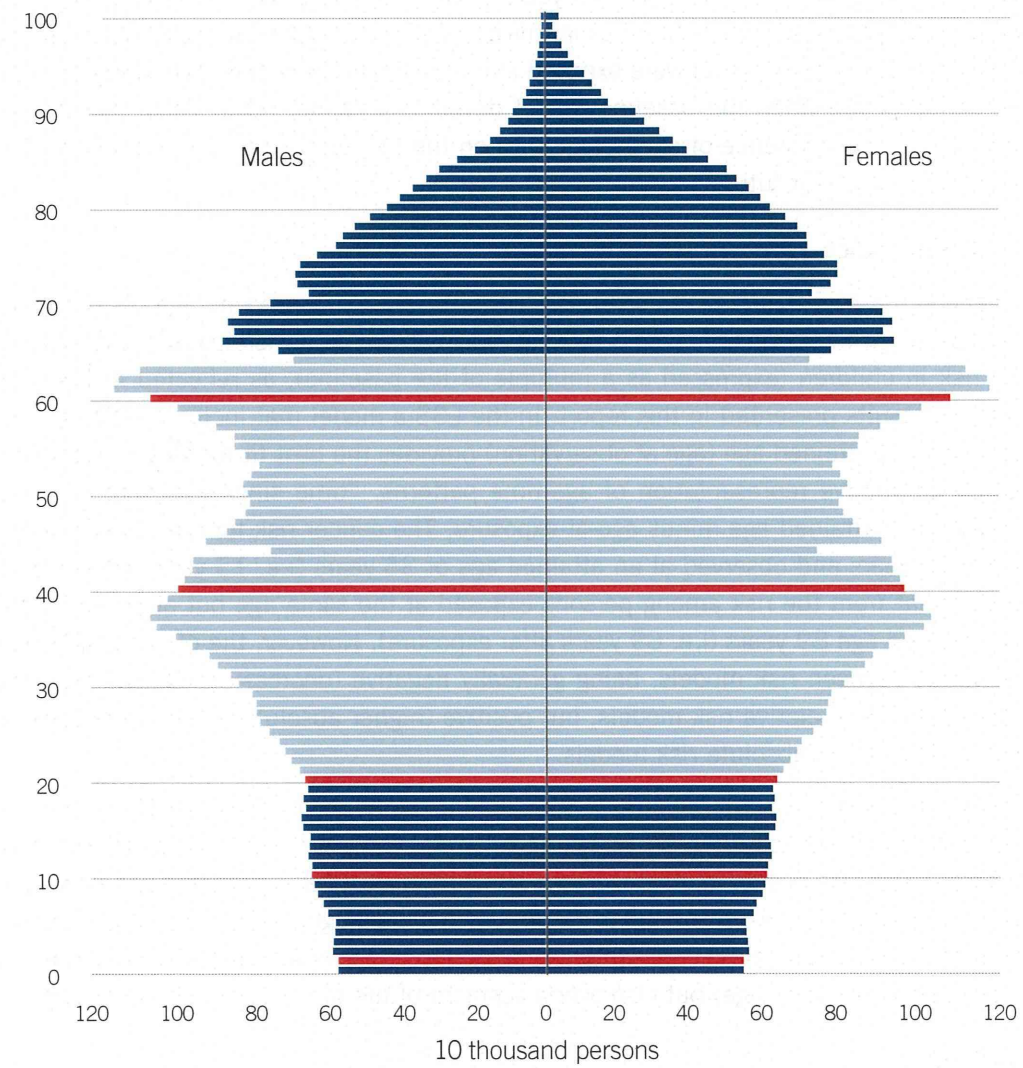


Figure 22. Japan population pyramid for 2010



Source: Ministry of Internal Affairs and Communication, Statistics Bureau, Director-General for Policy Planning (Statistical Standards) and Statistical Research and Training Institute (http://www.stat.go.jp/english/data/nenkan/1431-02.htm)
The red lines indicate the ages-at-exposure that were selected to be representative of the general population (infant, child, adult) and of the emergency workers (20, 40 and 60 years old).

For female breast cancer, a large impact of age-at-exposure on the excess absolute risk (EAR) was shown in epidemiological studies. This was one of the presumptions of the HRA Expert Group, which supported the decision to consider this cancer site separately from the other solid cancers in this assessment. The results of the HRA reflected this trend, though the risks were small in all age groups.

Less variability in risk by age-at-exposure has been shown for all solid cancers among atomic bomb survivors. However, the general trend of higher risks at younger ages was also found for solid cancers in this HRA. To illustrate this, Figure 21 shows an example of the influence of age-at-exposure on the LAR for solid cancers following radiation exposure to 10 and 20 mSv (sex-averaged values). It should be noted that the oldest population group considered for the HRA in the general population is 20-year-old adults. This is a conservative approach because, as seen in Figure 21, this is the most sensitive age group of adults in terms of cancer risk. This is particularly relevant for Japan's population, which has long been among the oldest in the world (Figure 22).

As most of the LSS survivors aged 40 years or older at the time of bombings have already died, information on cancer risk in this population is essentially complete today. In contrast, many survivors who were exposed in childhood are still alive and going through the period of life when the baseline risk of developing cancer rises. Thus, further evidence about the influence of age-at-exposure on the risk of cancer in adulthood can be expected in the near future.

6.1.2 Time since exposure and attained age

In general, baseline cancer incidence and mortality rates increase with age. As the baseline risk is low in childhood: even a modest absolute excess translates into a large relative risk in children (expressed as a multiple of the baseline), as reflected in the LAR and LFR results presented in this report. In the LSS a model that includes both age-at-exposure and attained age (age at observation) provides the best fit for all solid cancers. For an alternative representation of age-time patterns, "time since exposure" can be derived from attained age minus age-at-exposure. The cancer risk for someone exposed at age 20 years and observed at an attained age of 35 years (i.e. 15 years later) would be different from the risk among persons exposed at the same age but observed later– e.g. attained age 89 years (i.e. 69 years after exposure). However, the effect of attained age varies between risk models, being generally negative (decreasing relative risk with older age) in the relative risk models, but positive (higher absolute risk at older ages at observation) in the absolute risk models.

The tables in Annex L show cumulative baseline incidence up to 15 years (BR_{15}) and lifetime baseline incidence up to 89 years of age (LBR). The small absolute increase in risk that translates into a large relative increase in the15 years after exposure may be less evident in a lifetime follow-up (e.g. up to 89 years of attained age). Taking this into account, the HRA Expert Group assessed the cancer risks not only over the whole lifetime (i.e. up to 89 years of attained age) but also over a segment of life of 15 years after exposure.

6.1.3 Sex

Females tend to be at a greater risk of cancer from a given unit dose of radiation than males. The sex difference is largely independent of age-at-exposure. In the LSS study

of atomic bomb survivors, both the ERR and the EAR estimates for solid cancers are about 50% higher for women than for men (the female-male sex excess risk ratio is 1.6 for ERR and 1.4 for EAR). This is one of the key features that can be generalized about the association between radiation exposure and solid cancers observed in the atomic bomb survivors (16). No strong difference in leukaemia risk has been found between sexes (5,106). Nevertheless, no systematic differences between men and women have been consistently observed in studies with low exposures, such as in environmental and occupational settings.

The thyroid cancer risk following the Chernobyl accident has not shown a consistent sex difference as reflected in various findings reported in the literature (*107,108*). In the studies of fallout from the Nevada test site, no obvious sex differences were reported (*109*). In studies of the fallout in the Marshall Islands (*42*), and also in Mayak workers (*110*), there was some evidence for higher risk among women.

In this HRA, the female-to-male ratio was 1.46 for solid cancer LAR, and 4.5 for thyroid cancer LAR. In contrast, leukaemia LAR values are higher for males compared to females, with a female-to-male sex ratio of around 0.67.

6.2 Main sources of uncertainty

6.2.1 Exposure estimates for the general population

First-year dose estimation

The dose estimates used for the risk calculations are conservative. Owing to the preliminary nature of the data available as of September 2011, some assumptions about the implementation of protective measures and food consumption were deliberately based on options that are more likely to overestimate than to underestimate the radiation exposure. For example, assumptions were made that people in the most affected areas outside the 20-km radius continued to live there for 4 months after the accident, whereas a proportion of the population was relocated earlier, the assumption that all monitoring data on food were obtained from food available on the market, and the assumption that all people in Fukushima prefecture consumed only food produced in Fukushima prefecture. Therefore, some possible dose overestimation may have occurred.

This report is based on dose estimates calculated by dosimetric modelling using environmental and food monitoring data, rather than actual human measurements. The experience from the Chernobyl accident indicates that, when human monitoring data (e.g. whole-body counting) were used to determine more precise estimates of human exposure, actual doses were much lower than the hypothetical doses calculated through modelling (93). Data concerning internal and external exposure following the Fukushima Daiichi NPP accident were published during the last year. In some cases the doses reported were substantially lower than those reported in the WHO preliminary dose estimation (*111,112,113,114,115,116,117*). The Fukushima prefectural government and the Fukushima Medical University are now carrying out the estimation of external doses of all residents in Fukushima prefecture, in collaboration with the National Institute of Radiological Sciences (NIRS) (*116*). At the time of the publication of this report, interim data from the Fukushima Health Management Survey indicate external exposure for 99%

of residents at less than 10 mSv, with a highest estimate of 25.1 mSv. Internal exposure from $^{134}Cs/^{137}Cs$ is below 1 mSv in 99.9% of persons surveyed, based on whole-body counting (WBC) performed between June 2011 and July 2012 (*117,118*). It should be noted that short-lived radionuclides such as ^{131}I were no longer detectable when the reported WBC was performed. The methodologies and exposure pathways considered differ from those in the WHO Preliminary dose estimation (*3*). These, together with the main sources of uncertainty in the dose assessment for the first year, are discussed in more detail in section 4.6.

Organ doses

The Dose Expert Panel provided effective doses and equivalent thyroid doses resulting from the exposures during the first year after the Fukushima Daiichi NPP accident. The HRA Expert Group considered it more appropriate to refer to individual organ doses in red bone marrow, thyroid, breast and colon as input data for the cancer risk models for leukaemia, thyroid cancer, breast cancer and all solid cancers, respectively. As explained in Chapter 4, the ratios between the absorbed dose and the effective dose in each of the above organs were applied to the effective dose to obtain organ doses. There is a relationship between organ doses and effective dose, as the effective dose is a weighted sum of organ doses, but this relationship depends on the radionuclide involved. The calculation of the appropriate ratios has therefore an intrinsic uncertainty due to the radionuclide composition of the releases and consequent deposits.

Lifetime doses

For the purpose of calculating the lifetime risk, the HRA Expert Group assumed that the ratio of long-term dose to 1-year dose would be equal to 2[1] on the basis of the Chernobyl experience, and taking into consideration the differences between the Chernobyl and Fukushima Daiichi NPP accidents. The distribution of the lifetime dose on a yearly basis was provided to the risk modellers for the health risk calculations. Implicit in this assumption on the temporal distribution of the lifetime dose are a number of uncertainties associated with the influence of the natural mechanisms mentioned in section 4.1.4, as well as a number of interventions that can reduce radiation exposure – including more stringent standards and remedial actions (e.g. clean-up of buildings, remediation of soils and vegetation, treatment of agricultural fields, waste management). The HRA Expert Group acknowledged these uncertainties and considered it important to perform additional LAR calculations, taking into account first-year exposures. These additional calculations included LAR both up to 15 years after exposure and up to 89 years of age. By comparing the results, the HRA expert group concluded that LAR values do not differ substantially (see one example in Table 17). This is predictable for the locations in Group 1 where it was assumed that relocation of the population took place at 4 months. The factor 2 applied to the first-year dose to get the lifetime dose does not translate into a doubling of the risk. This is because lifetime doses are delivered over many decades, and cancer risks decrease with attained age.

1. With an exception for the locations where people were relocated (i.e. Namie town in Futaba county, Iitate village in Soma county and Katsurao village in Futaba county). In those locations it was assumed that relocation took place at 4 months after the accident. Therefore the dose over the lifetime was calculated as the sum of the doses received during the first 4 months after the accident plus the lifetime dose calculated for the locations within Fukushima prefecture zone 1 (western least contaminated).

Figure 23. Lifetime attributable risk (LAR) for leukaemia in one-year old males (first-year dose 10 mGy) using different transfer weights (100% ERR, 100% EAR and 50% ERR-50% EAR)

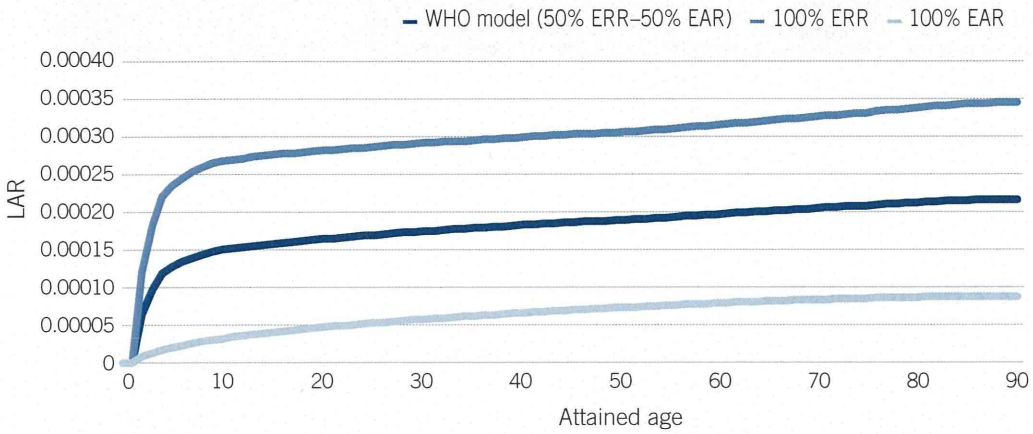


Figure 24. Lifetime attributable risk (LAR) for thyroid cancer in one-year old males (first-year dose 10 mGy) using different transfer weights (100% ERR, 100% EAR and 50% ERR-50% EAR)

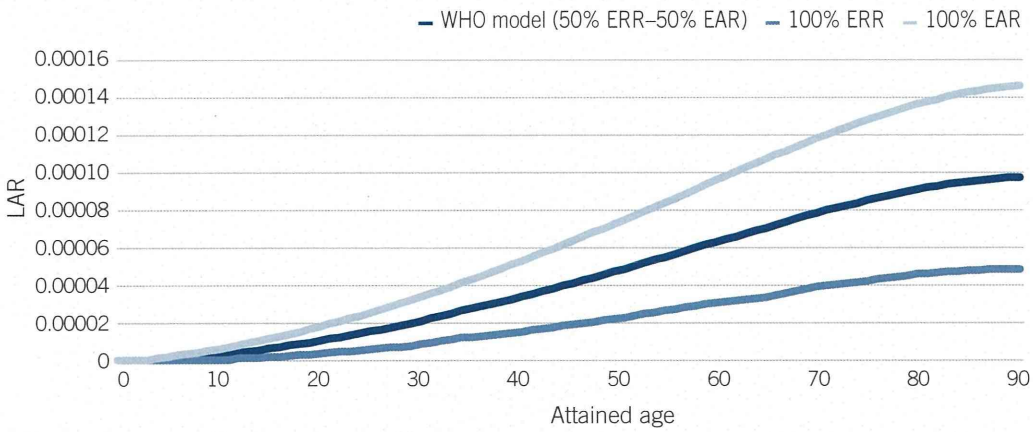


Figure 25. Lifetime attributable risk (LAR) for solid cancer in one-year old males (first-year dose 10 mGy) using different transfer weights (100% ERR, 100% EAR and 50% ERR-50% EAR)

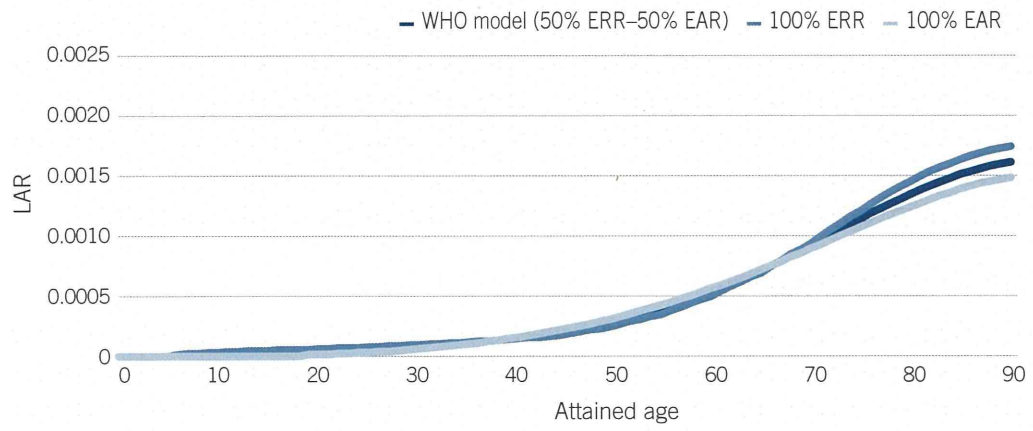


Table 17. Comparison of LAR for all solid cancer calculated using first year doses vs. lifetime doses (20 year old female)

Locations	First year exposure	Lifetime exposure
①	0.567	0.591
②	0.309	0.336
⑤ – ⑭	0.103	0.171

6.2.2 Health statistics

The HRA Expert Group agreed that cancer data from Fukushima were likely to be comparable to those from other parts of Japan. This determination was made on the basis of the similarity of cancer incidence in two neighbouring prefectures for which cancer registries are available (Miyagi and Yamagata) and the other Japanese cancer registries. Also, similarities were found between cancer mortality data in those two neighbouring prefectures compared with cancer mortality data in Fukushima and data from the rest of Japan (although it is noted that thyroid cancer data were insufficient to make a comparison for this organ).

As cancer registries were an important data source for this study, the HRA Expert Group looked for evidence of the robustness of these data sets. First, it was noted that the increasing trend in cancer incidence and declining mortality among both men and women in Japan for the 1984–2005 period was in agreement with international trends. An examination of skin cancer data from registries provided a specific opportunity to probe for registry reliability, as reported disease rates are very low in Japan (on the order of 5% of Caucasian rates). A comparison with the Hawaiian registry (which allows comparison of Japanese and Caucasian populations with similar environmental exposure and the same systems for ascertainment) shows that reported rates of skin cancer in Japan appear to be consistent with the incidence of disease in the Japanese population and confirm that low reporting from Japanese registries is not a result of poor case ascertainment.

Ideally, the ICD codes for the input data (mortality and incidence reference rates) should match those of the data used to fit the incidence risk models for all solid cancers, thyroid cancer and breast cancer (*70*) and the mortality risk models for leukaemia (*83*). In practice, incomplete concordance between some ICD codes was found, but discrepancies were not large, and hence the HRA Expert Group considered that this would not constitute an impediment to using the proposed risk models. The cancer incidence used for this assessment (*104*) corresponds to statistics from 2004, and mortality data used to build the survival curves are more recent (2010). This mismatch was acknowledged by the HRA Expert Group, which considered that it would not substantially affect the results.

6.2.3 Risk models applied

As explained in section 3.3, the HRA Expert Group adopted a hybrid model combining relative and absolute risk approaches for transferring risks estimated from the Japanese atomic bomb survivors (i.e. LSS cohort) to the Japanese population exposed to radiation from the Fukushima Daiichi NPP accident, except for breast cancer for which a pure absolute risk model was used (see Table 3). In addition, calculations were performed for the

pure absolute and relative risk models for leukaemia, thyroid cancer and solid cancers that are also included in the results tables provided in Annexes J and K.

When choosing an approach to transfer radiation-related cancer risks from one population to another, considerations should include comparison of radiation risks from epidemiological studies in populations with different baseline cancer rates. The knowledge about the interactions between radiation and other cancer risk factors (e.g. environmental and genetic cancer risk factors) and their influence on baseline cancer rates is still limited, and this represents one of the sources of uncertainties related to the risk-transfer models applied.

The risk-transfer approach adopted by the HRA Expert Group (i.e. the hybrid model) is considered to be a reasonable compromise that provides an intermediate estimate of the risks, as seen in Figures 23–25.

The differences between the relative and absolute risk models were largest for young age-at-exposure. When risks were predicted over a 15-year observation period, excess risk estimates for all solid cancers and leukaemia obtained using the relative risk model were substantially higher than those predicted by the absolute risk model for the children exposed at 1 year of age, particularly for girls (ERR:EAR ratio of around 5 for 1-year-old males and 6 for 1-year-old females). For thyroid cancer, the relation was reversed. For exposure at 10 years of age, the risk estimates for all solid cancers in boys were comparable and the relative risk model gave slightly higher estimates than the absolute risk model among girls (ERR:EAR ratio of around 1.4). When risks were predicted over a lifetime, differences between the two models were not evident for all solid cancers but persisted for leukaemia and thyroid cancer.

In addition to the uncertainties associated with exposure estimates and other input data, the proposed model structure and parameters are also based on common assumptions derived from uncertain values. Uncertainty attached to the model definition includes estimates of uncertainty on the parameter values of EAR and ERR models (drawn from the literature), latency, weighting of EAR versus ERR, and other modelling assumptions, such as survival curve parameters.

Overall, the model assumptions were either conservative (i.e. they are intended to overestimate rather than underestimate the radiation-related risk) or consistent with the published literature. Preliminary sensitivity analysis of LAR estimates with respect to these model assumptions showed that uncertainty in model parameters was likely to be lower than uncertainty in exposure, so that if other risk models were applied the uncertainties would still be dominated by those related to exposure. It is noted that the chosen model structure becomes highly hypothetical for very low-dose exposures so that the proposed risk estimates are reasonably robust only for those proposed exposure scenarios for which risk quantification was judged to be feasible.

Radiogenic risks in children are generally associated with more uncertainty than risks in the entire population. One reason for this is the fact that the LSS atomic bomb cohort has many survivors who were children at the time of the bombing and who still have years of life to express the risk.

Further research is still needed before achieving definitive conclusions about the optimal choice for risk-transfer between populations (*119*). The risk-transfer approach adopted by