Review of Scientific Literature on the Health Effects of Exposure to Depleted Uranium

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Prepared by:

The Expert Committee to Examine Balkan Veteran Exposure to Depleted Uranium

On behalf of:

The Minister for Veterans' Affairs
Commonwealth of Australia

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Executive Summary

- Uranium is a naturally occurring element found in all soils, rocks, rivers, lakes, oceans, plants and animals. Consequently, all humans are exposed to naturally occurring uranium through ingestion, inhalation and skin contact.

- Depleted uranium arises as a by-product of the uranium enrichment process and may also be obtained by reprocessing spent nuclear fuel. It has the same chemical properties as natural uranium but is 40% less radioactive than natural uranium.

- Uranium is chemically classified as a heavy metal and is weakly radioactive. Any health effects should therefore be considered potentially arising through chemical toxicity or radiation emissions.

- The primary radiation emitted from uranium is in the form of alpha particles. These particles have an extremely short range and are unable to penetrate a sheet of paper or human skin. Inhalation and retention of uranium particles may expose immediately adjacent tissue to ionizing radiation from alpha particles.

- Uranium also emits very small amounts of the more penetrating beta and gamma forms of radiation. Levels of these emissions are very low in comparison to levels of these forms of radiation arising from natural sources such as cosmic rays and other naturally occurring radioactive minerals.

- Depleted uranium sourced from reprocessing spent nuclear fuel contains trace amounts of other radioactive substances that add less than 1% to the low radiation emissions from depleted uranium antitank shells and armour.

- The chemical toxicities of natural uranium and depleted uranium are identical and are dependent on dose, chemical form and route of exposure.

- Depleted uranium has a wide range of everyday, civilian applications. For example, it is used in health care (radiation shielding in equipment used for radiation therapy and containers for radio isotopes), aviation (passenger aircraft counterweights), boat building (counterweights in racing sailboat keels), satellite ballast, and petroleum exploration (drilling equipment). It may also be used as a chemical catalyst eg in the production of synthetic ammonia.

- Due to its high density, high melting point, tensile strength, pyrophoric properties in particle form and availability, depleted uranium metal has military applications in heavy tank armour and munitions including armour piercing, anti-tank shells. On impact with a hard target, a fraction of the depleted uranium in munitions undergoes spontaneous ignition and small, relatively insoluble particles of mainly uranium oxides, as well as fragments of metallic depleted uranium are formed.

- Pathways for exposure to depleted uranium that has been used in military operations are the same as those for natural uranium and are:
  - Inhalation in smoke and dust;
  - Hand to mouth contamination and ingestion of dusts;
  - Contamination of wounds;
  - Skin contact;
  - Agricultural pathways through uptake by crops or grazing animals;
  - Accumulation in drinking water.

- Close proximity to depleted uranium metal, as in storage facilities, carrying shells or driving tanks, even when prolonged, produces negligible internal radiation exposure and levels of external radiation exposure well below the recommended levels for occupational safety.

- Available sound-medical scientific evidence related to the health effects of depleted uranium includes data on human exposure to uranium, studies of miners and workers in the uranium industry, animal experiments, and some studies of Gulf War veterans.

- In animal experiments involving doses of uranium massively above recorded human exposures adverse health effects have been found, including renal failure, respiratory disease and in one species lung cancer.

- The kidney is the organ that is most sensitive to uranium's chemical toxicity. Impairment of kidney function at high levels of uranium exposure has been reported in humans, but is usually reversible. No overall increase of morbidity or mortality from genitourinary disease has been observed in large occupational studies of chronically exposed workers.

- A cohort of US Gulf War veterans who were in combat vehicles at the time the vehicles were struck by depleted uranium munitions have elevated urinary uranium concentrations. No kidney toxicity or other adverse health effects related to depleted uranium have been observed in this group after a decade of follow-up.

- Inhaled insoluble uranium particles less than 10 μm in size may be retained in the lung and continue to emit alpha particles into the lung tissue. However, no excess of respiratory cancer or non-malignant respiratory disease has been found in relation to uranium exposure among combined cohorts of uranium process workers.

- There is no evidence of a significant increase in deaths from any cause, nor from all cancers, nor individual types of cancer nor genitourinary disease in a combined analysis of the cohorts of uranium process workers who have been followed for many years.

- It has been estimated that uranium oxides and metal from spent depleted uranium munitions have made only a small contribution (about 5% at Kosovo target sites) to the occurrence of uranium already naturally present in the soil. The additional contribution of depleted uranium from military use to background radiation dose is within the natural variations found for background levels.

- The Expert Committee considered a range of possible military scenarios for internal and external exposure to depleted uranium ranging at the highest level from being in a vehicle when it is struck by a depleted uranium penetrator down to the lowest level of loading and carrying intact depleted uranium munitions.

- On the basis of the available sound medical-scientific evidence and under realistic assumptions of exposure and dose the Expert Committee concluded that depleted uranium could not produce any adverse health effects in Australian troops serving with NATO forces in the Balkans conflict.

- The sound medical-scientific evidence used in this investigation is extensively referenced in the following text and relevant examples are provided in tables and appendices.

Section One

Background to the Expert Committee's Investigation

Introduction

In January 2001, news media in many parts of the world carried reports suggesting links between North Atlantic Treaty Organisation's (NATOs) use of depleted uranium ammunition in Kosovo and Bosnia and allegedly higher rates of leukemia, other cancers, and other ill-health among NATO (KFOR/SFOR) troops and the local civilian population. These reports followed earlier concerns that ill health amongst veterans of the Persian Gulf War and resident Iraqis and the "Gulf War Syndrome" may be linked to exposure to depleted uranium.

In late January 2001 it was announced that the Australian Defence force would examine the potential for exposure to depleted uranium of approximately 260 Australian Defence personnel who have served with NATO forces in the Balkans. Also, an Expert Committee would be formed to examine the medical scientific evidence for health effects from such exposure to depleted uranium.

The Expert Committee to Examine Balkan Veteran Exposure to Depleted Uranium was accordingly appointed on 19th February 2001 by Mr Neil Johnston, Secretary for the Department of Veterans' Affairs on behalf of the Minister for Veterans' Affairs, the Hon. Bruce Scott MP. The Expert Committee was asked to review the medical-scientific literature on depleted uranium exposure and comment on an appropriate medical follow-up regimen for Australian Defence Force (ADF) members.

Terms of Reference

- To examine and report on whether there is any sound medical-scientific evidence of adverse health effects of exposure to depleted uranium and, if so, what are those effects and what is the nature and strength of the evidence.

- To examine the questionnaires and medical tests conducted by the ADF on current and former members of the ADF who served in the Balkans, to advise on:
  - whether these indicate exposure to depleted uranium by Australian ADF members in the Gulf War and Balkans Conflict and the extent and nature of that exposure; and
  - whether these point to any adverse health effects that may be associated with or caused by exposure to depleted uranium; and
  - an appropriate health monitoring regime for these veterans.

  It is proposed that the Expert Committee report at two stages:

  - by 31 July 2001, on the review of scientific literature on the health effects of exposure to depleted uranium; and
  - by 30 September 2001, on the results of the questionnaires and health screenings of the ADF members who served in the Balkans, and on a follow-up regime.

The term "sound medical-scientific evidence" is defined in Section 5AB2 of the Veterans' Entitlements Act and the definition is at Appendix One.

Membership of the Expert Committee

The committee is made up of the five members of the Repatriation Medical Authority and two experts in the field of heavy metal and radiation toxicity.

Professor Ken Donald MBBS, PhD, FRCPA, MRCPath, FRACMA, FRACS, who is currently the Head of the School of Medicine, University of Queensland and formerly the Professor of Social & Preventive Medicine, and Head of Department of Social & Preventive Medicine, University of Queensland.

Professor Beverley Raphael AM, MBBS, MD, FRANZCP, FACP, FRCPsych, FASSA, who is currently Director, Centre for Mental Health, NSW.

Professor John Duggan AM, MBBS, MD, FRCP, FRACP, FQSA, FRACMA, who was Staff Specialist, Royal Newcastle Hospital 1958-89 and Clinical Associate Professor, University of Newcastle.

Professor John Kearsley MBBS, PhD, FRACR, FRACP, who is currently Director, Division of Cancer Services, Cancer Care Centre, St George Hospital, Sydney and (conjoint) Professor of Radiology Oncology University of New South Wales.

Professor John Kaldor PhD, who is currently the Deputy Director, and Head of the Epidemiology Unit, of the National Centre in HIV Epidemiology and Clinical Research, The University of New South Wales.

Professor Michael Moore BSc, PhD, DSc, MACM, who is currently Director of the Queensland Health Scientific Services and National Research Centre for Environmental Toxicology (University of Queensland); Director of the Australian Centres for Health Risk Assessment and Adjunct Professor of Public Health (Griffith University).

Professor Barry N Noller BSc, MChem, PhD, FRACI, FRSC, CPChem, who is currently the Deputy Director for the National Research Centre for Environmental Toxicology.

Working Procedures

The Expert Committee members were selected for their medical and scientific expertise and broad knowledge and research in the fields of heavy metal toxicity, ionizing radiation and risk assessment. The Australian Expert Committee's Terms of Reference required a consideration of the primary and review literature, and other relevant materials.

The Expert Committee has relied on a number of published reviews for material relating to the chemical effects of uranium exposure and the radiological effects from uranium isotopes. Primary literature relating to the health effects of exposure to depleted uranium was sought using the Medline database and the reference lists of a number of recent reviews of the health effects of depleted uranium. Internet searches were undertaken relating to depleted uranium exposure. Individual Expert Committee members had knowledge of research in the fields of heavy metal toxicity, ionizing radiation and risk assessment. A description of epidemiological terms and

principles, and approaches to the assessment of epidemiological evidence is provided at Appendix Two.

Additionally, a number of organisations have recently reviewed the military use of depleted uranium and exposure in a combat setting.

The Expert Committee held its first meeting on 5th March 2001 and had a total of five meetings during the production of this report. The Expert Committee examined the chemical, toxicological and epidemiological data and assessed the information concerning causality.

Data collection

- The Expert Committee examined published primary and review literature on depleted uranium, and considered the background and substance of the reviews and the primary sources of published literature utilised in their production.
- Additionally Medline searches were undertaken for terms including depleted uranium, uranium, using MESH headings and for epidemiology, aetiology and chemically induced subgroups as well as two year unlimited searches for depleted uranium and uranium.
- Specific searches undertaken for individual factors where information in paragraph two may not be definitive.
- Textword and author searches as per paragraph three.
- Where necessary, referenced texts, other publications, and reference lists were also used to identify primary source material, and to ensure that examination of reported associations was undertaken.
- Search of websites and databases including: Agency for Toxic Substances and Disease Registry, National Institute for Occupational Safety and Health, National Institute of Health, World Health Organisation, United Nations Environmental Program, Gulflink, UK DoD, US DoD, and NATO, as well as general internet searches for sites concerning uranium and depleted uranium, the Gulf War and Balkans War.
- Liaison with and data collection from members of the Australian Defence Force, Canadian Defence Force, World Health Organisation, and data from a presentation by Dr McDiarmid.

Major References and Reviews

In addition to the primary literature, the Expert Committee had the benefit of a number of significant contemporary reviews of the literature concerning uranium and depleted uranium exposure and these included:

- Agency for Toxic Substances and Disease Registry (ATSDR) Toxicological Profile for Uranium prepared by RTI under contract no 205-93-0606 for USDHHS, Sept 1999.
- European Commission: Directorate-General Environment (2001). Opinion of the Group of Experts Established According to Article 31 of the Euratom Treaty - Depleted Uranium. European Commission: Directorate-General Environment, Directorate C - Nuclear safety and civil protection, ENV.C4 - Radiation protection.

- Fulco CE, Liverman CT and Sox HC [Eds] (2000) Gulf War and Health: Volume 1 - Depleted Uranium, Pyridostigmine Bromide, Sarin, Vaccines. Institute of Medicine, National Academy Press, Washington DC.
- Harley N, Foulkes CE, Hilborne L, et al., "A Review of the Scientific Literature As It Pertains to Gulf War Illnesses: Vol.7, Depleted Uranium," MR-1018/7-OSD (Santa Monica, CA: RAND, 1999).
- Office of the Special Assistant for Gulf War Illnesses (OSAGWI) (1998). Environmental Exposure Report: depleted uranium in the Gulf (I). http://www.gulflink.osd.mil/du/
- Office of the Special Assistant for Gulf War Illnesses (OSAGWI) (2000). Environmental Exposure Report: depleted uranium in the Gulf (II). http://www.gulflink.osd.mil/du/
- The Royal Society (2001). The Health Hazards of Depleted Uranium Munitions - Part 1. May 2001.
- United Nations Environmental Program (UNEP) (2001) Depleted Uranium in Kosovo-post-conflict environmental assessment. http://balkans.unep.ch/du/reports/report.html
- United Nations Environmental Program (UNEP) (1999). The Kosovo Conflict. Consequences for the environment & human settlements. UNEP, UNHS
- World Health Organisation (WHO)(2001a). Report of the World Health Organisation : depleted uranium mission to Kosovo. 22 to 31 January 2001.
- World Health Organisation (WHO)(2001b). Depleted Uranium: Sources, Exposure and Health Effects. WHO, Department of Protection of the Human Environment, Geneva, [WHO/SDE/PHE/01.1].

The United Nations Environmental Program published assessments in 1999 and 2001 and the European Union and World Health Organisation also reviewed the risk from depleted uranium exposure in 2001.

Specific assessments by OSAGWI (1998 and 2000) considered depleted uranium exposure and modelled dose estimates. The Royal Society (2001) also undertook mathematical modelling of depleted uranium exposures and resultant dose estimates.

As well, individual NATO countries have sought to re-evaluate the available literature and undertake further research into the potential risks experienced by military personnel and in some cases exposed populations.

Section Two

Definition of Uranium and Depleted Uranium

Uranium

Uranium (U) is a naturally occurring element. It is ubiquitous occurring in rocks, soil, rivers, oceans, plants and animals. The Agency for Toxic Substances and Disease Registry estimates that there are typically four tons of uranium in one square mile of soil, one foot deep (comparable to 1.4 t/km²) (ATSDR, 1999b). It is more abundant than elements such as mercury, silver, or gold; about the same as tin; and slightly less abundant than cobalt, lead or molybdenum (UNEP, 1999). It has also been estimated that each year 180 tons of uranium decay products are added to US agricultural lands due to the presence of trace amounts of uranium in fertilisers (ATSDR, 1999b).

Table 2.1:
Concentrations of uranium in various environmental systems and materials (ATSDR, 1999b)

Physical Entity	Abundance (mg/kg)
Crystal rocks	1.800
Sea water	0.0033
Stream water	0.00004
Human tissue	0.001

Uranium Properties: Physical and Chemical

Uranium is a heavy metal (a metal with a specific gravity of 5.0 or greater) with an atomic number of 92 and an atomic weight of 238.0289 (Lide, 1995). It has a very high density of 18.95 g/cm^3 which is 1.7 times higher than lead (11.35 g/cm^3). Metallic uranium has a high melting point (1132^{o}C), and boiling point (4131 ^{o}C), has a tensile strength similar to most steels and is chemically very reactive (Lide, 1995; Kirk, 1981). In powdered form it has pyrophoric properties, that is, it has the tendency to spontaneously ignite in air when in the form of fine particles.

Naturally occurring uranium, enriched uranium (uranium with elevated concentrations of U^{235}) and depleted uranium (uranium with decreased concentrations of U^{235}) all have the same number of protons and have identical physical and chemical properties (Harley et al, 1999; Fulco et al, 2000).

Appendix Three provides data on the physical and chemical properties of metallic uranium and selected uranium compounds, particularly uranium oxides (UO_2, UO_3, U_3O_8), the major compounds produced after uranium undergoes combustion (ATSDR, 1999b); and the relative solubility of a number of uranium compounds. Different uranium compounds vary with regard to a number of properties including their solubility and these properties affect the potential for chemical and radiation toxicity of any specific uranium compound.

While uranium is ubiquitous in the environment it has no known function in human or animal metabolism. Body fluids can dissolve uranium oxides. Solubilised uranium may react with biological molecules and exert its toxic effects (Hursh and Spoor, 1973). Heavy metals including uranium have a strong affinity for many biological molecules containing phosphate residues such as glucose phosphate, phospholipids, and nucleic acids, or sulphydryl groups including cysteine, glutathione, and many proteins. Because of this high affinity uranium does not exist as a free ion in biological systems, rather it is present in complexes with a variety of molecules. In blood and most body fluids the most important species controlling uranium mobility are the carbonate, bicarbonate species and the citrate complexes of U(VI) which are all soluble (Hursh and Spoor, 1973; Cooper 1982). In blood, at near neutral pH the compound is stable and in this form does not significantly react further with biological molecules. It readily decomposes, however, at more acid pH in urine, with liberation of the reactive uranyl ion. Once the uranium is solubilised in the blood, the kidney will efficiently excrete about 90% of it in urine over approximately three days. Renal excretion of uranium, like that of other heavy metals, is determined by such factors as the ability of the circulating complexes to be filtered out and on the ability of filtered complexes or their decomposition products to be reabsorbed or secreted in the tubule (Harley et al, 1999).

Uranium Properties: Radiation

Natural uranium is weakly radioactive and consists of three isotopes U^{238} (99.276%); U^{235} (0.718%) and U^{234} (0.0056%) (ATSDR, 1999b). All isotopes of uranium have the same number of protons however the number of neutrons in the nucleus varies, giving the isotopes different radiological properties. Radioactive nuclei transform spontaneously into nuclei of another element. Usually this process is accompanied by the emission of radiation.

U^{238}, which constitutes more than 99% of the mass of natural uranium, is the least radioactive of the three isotopes. In contrast, U^{235} exhibits approximately seven times and U^{234} approximately 18,000 times the radioactivity of U^{238} per unit weight. U^{234} contributes to more than half of the radioactivity of natural uranium even though by weight its percentage is extremely small (ATSDR, 1999b; Fulco et al, 2000).

Different types of atomic decay can be distinguished. In an alpha-decay, the nucleus emits a particle consisting of two protons and two neutrons (a helium nucleus). The range (the distance the charged particle travels from the point of origin to its resting point) of alpha particles is approximately 3-10 cm; in biological tissue the range decreases dramatically to 25-80 μm. Alpha particles released by uranium are unable to penetrate a sheet of paper or human skin and are not considered to be an external hazard. Alpha emitters, however, can be a health hazard if inhaled or ingested in sufficient quantities. Once the alpha particle's energy is expended it combines with two electrons to become a helium atom, which is not chemically reactive with biological material. In a beta-decay, the nucleus emits an electron. Beta particles are able to penetrate skin a few millimetres and can pose both an internal and external health risk. Alpha, as well as beta-decay, can be accompanied by gamma radiation, a high energy electromagnetic radiation. Since a gamma ray is a photon of energy with no mass and no charge, it is extremely penetrating, and can be both an internal and external health hazard (ATSDR, 1999a). See figure 2.1 for the relative penetration of these emissions.

The three isotopes of naturally occurring uranium decay via alpha particle emission. These isotopes possess very long half-lives, the shortest being 250,000 years for U^{234}. U^{238} decays via emission of alpha particles into two short-lived "progeny", thorium-234 (Th^{234}, half-life of 24.1 days) and protactinium-234m (Pa^{234m}, half-life of 1.17 minutes) which are beta and weak gamma emitters. Because of this constant nuclear decay process, very small amounts of these progeny are always present in both natural and depleted uranium. U^{235} decays into protactinium-231 (Pa^{231}, half-life of 32,500 years), which is an alpha, beta, and gamma ray emitter. The U^{238} and U^{235} chains continue

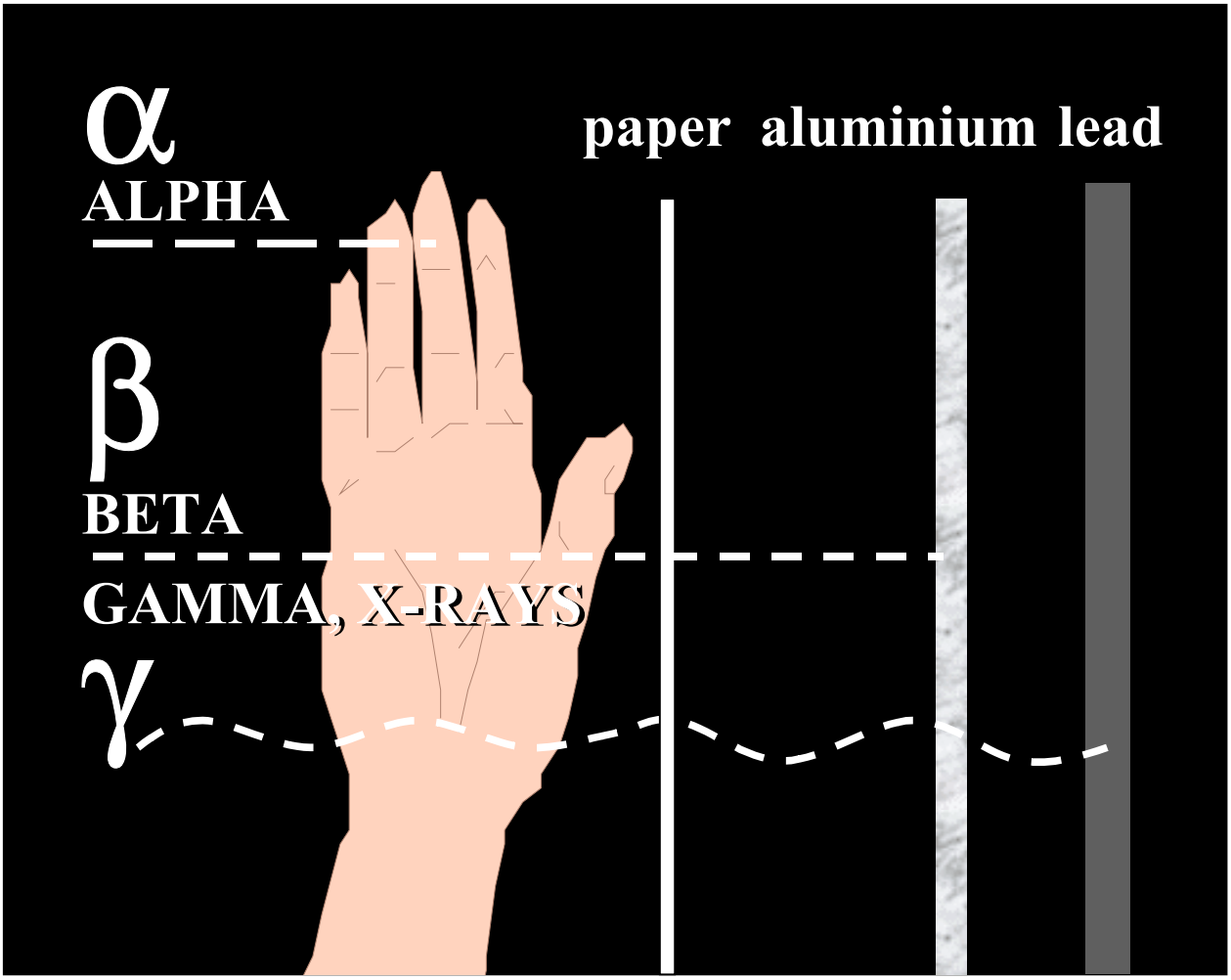

Figure 2.1
Relative penetration of alpha, beta and gamma emissions.

through a series of long-lived isotopes before finally decaying to the stable, non-radioactive lead isotopes (ATSDR, 1999b; Harley et al, 1999). Appendix Four details the decay chains for U^{238} and U^{235}.

In nature, natural uranium, together with its progeny yields 7-8 times as many alpha decays per unit time as pure uranium. When uranium is separated from ore, the decay products, being different elements, are removed. After separation, the uranium isotopes continue to decay. The first members of the decay chains have short half-lives, and soon return to their equilibrium values. However, Th^{230} (formed from decayed U^{234}), and Pa^{231} (formed from decay of Th^{231}) have long half-lives, and so "grow" in very slowly. Hence in processed uranium (natural or depleted) the activities of Th^{230} and Pa^{231} and subsequent members of the chains are negligible (Harley et al, 1999).

The unit of measurement for radioactivity is the Becquerel (Bq). An activity of one Bq means that one decay takes place per second. U^{238}, U^{235} and U^{234} predominantly emit alpha particles (>95%). The alpha-activity of natural uranium amounts to about 25 kBq per gram. The progeny from the alpha decay of uranium continue themselves to decay, mostly by emitting beta-particles. The activity of these progeny is added to that of uranium. The beta-radiation of the progeny of natural uranium and depleted uranium have practically the same intensity, amounting to about 25 kBq/g. Uranium, together with its progeny has an activity of 50 kBq/gm. This means that 50,000 decays take place per gram per second.

The very long half-life of U^{238} (4.5 billion years) yields a very low decay rate per unit mass of uranium. Naturally occurring uranium, which mostly consists of U^{238}, is one of the least radioactive substances containing unstable isotopes on the planet. It is classified by the International Atomic Energy Agency in the lowest hazard class for radioactive materials (ATSDR, 1999b; Fulco et al, 2000).

Exposure to the radiation emitted from uranium can occur if it is outside the body or if it is taken in by ingestion, inhalation or other means. It is useful to consider the exposure pathways with regard to average radiation exposure in the normal environment. The Sievert (Sv) is the SI measure of radiation expressed as a dose equivalent (1000 mSv equals 1 Sv). In the general population ingestion of uranium and its decay series in food and drink gives a committed effective dose of 0.11 mSv per year for adults as compared to 0.0058 mSv through inhalation, excluding radon (1.2 mSv). This dose corresponds to 5% of the average annual dose due to internal and external exposure to natural sources of radiation (2.4 mSv). It relates essentially to progeny, Pb^{210} and Po^{210}. U^{238} accounts only for 0.00025 mSv total dose and 0.000021 mSv for inhalation. External exposure from all natural U^{238} in soil is also negligible. U^{238} series together with other primordial radionuclides, Th^{232} and K^{40}, cause a world-average annual external exposure of about 1 mSv per year. (UNSCEAR, 2000).

Uranium, is but one of many sources of ionizing radiation. The world is bathed with low levels of radiation all the time. The sources of radiation dose include radon (55% of the total), cosmic rays (8%), and man-made sources, such as x-rays, nuclear medicinal exposures, consumer products such as smoke detectors and colour television, and other sources. A further discussion of ionizing radiation and glossary of relevant terms are contained in Appendix Five.

Depleted Uranium

Depleted uranium is a by-product of the uranium enrichment process. Enriched uranium is used for nuclear power and nuclear weapons, which require the greater concentration of U^{235} ranging from 2–90% by weight, rather than the 0.7% found in natural uranium (ATSDR, 1999b). To achieve this increased concentration, naturally occurring uranium is subjected to an enrichment process. One product is enriched uranium which contains an increased percentage of the fissionable isotope U^{235} and is used for nuclear reactor fuel and nuclear weapons. The other product is depleted uranium. Depleted uranium may also be obtained from uranium extracted from spent nuclear fuel rods by reprocessing.

Depleted uranium is chemically identical to, and 40% less radioactive than, natural uranium. Two-thirds of the U^{235} and a proportion of the U^{234} is removed from natural uranium to produce depleted uranium, thus, depleted uranium is almost entirely U^{238}. The Agency for Toxic Substances and Disease Registry definition for depleted uranium is uranium having the weight percentage of U^{235} smaller than the 0.7% found in natural uranium (ATSDR, 1999b). Military specifications mandated by the US Department of Defense (DoD) require that the percentage of U^{235} be less than 0.3%, and the US DoD uses depleted uranium with a U^{235} content of approximately 0.2% (OSAGWI 1998; OSAGWI, 2000).

Depleted Uranium: Physical and Radiological Properties

The physical and chemical properties of depleted uranium are the same as those of natural uranium. As with naturally occurring uranium (see Appendix Three) the physical and chemical properties of depleted uranium are dependent on its oxidation state and physical properties such as its pyrophoric capacity depend on whether it is a continuous solid mass or in fine particles.

Depleted uranium is almost entirely U^{238} in isotope composition and is 40% less radioactive than natural uranium. The alpha activity of depleted uranium amounts to about 15 kBq/gram. Together with its progeny, depleted uranium has an activity of approximately 40 kBq/gram. This means that about 40,000 decays take place per gram per second. Depleted uranium has a low radioactivity per unit mass of material (specific activity) and is considered as only weakly radioactive (UNEP/UNCHS Balkans Task Force 1999; Snihs & Åkerblom 2000).

Comparison with selected radioactive materials in Table 2.2 below demonstrates the low specific activity of uranium and depleted uranium.

Table 2.2:
A comparison of the activities of selected radioactive materials
(European Commission Report, 2001)

Radioactive Material	Specific Activity
Iodine ^{131}	4,598,000,000,000 kBq/g
Cesium^{137}	3,206,000,000 kBq/g
Plutonium^{239}	2,298,000 kBq/g
Natural uranium together with its progeny	50 kBq/g
Depleted uranium together with its progeny	40 kBq/g
Natural uranium	25 kBq/g
Depleted uranium	15 kBq/g

Implications of Depleted Uranium Sourced from Reprocessing

Depleted uranium sourced from reprocessing of spent nuclear fuel rods has been identified in some munitions used in the Balkans conflict (UNEP, 2001a; UNEP, 2001b) and in the Gulf War (US AEPI, 1995). In depleted uranium made in this way, U^{236}, a synthetic isotope not found in natural uranium but with similar specific activity to U^{238}, will be present. U^{236} is produced by neutron bombardment of U^{235}. Similarly transuranics (Np^{237}, Pu^{239}, Pu^{240}, Pu^{241} and Am^{241}) are produced via neutron bombardment of U^{238}. See Appendix Four, figure A4.1 for further detail. Trace amounts of these radionuclides and fission products (Tc^{99}) may also be present (OSAGWI, 2000).

An analysis of 60 samples reported to be representative of the stockpile of depleted uranium used to manufacture all ammunition and armour since the Gulf War undertaken by US Department of Defense concluded that depleted uranium "may contain trace levels (a few parts per billion) of transuranics. These tests on samples of depleted uranium have shown that transuranic contamination added 0.8% to the radiation dose from depleted uranium." (OSAGWI, 2000)

Based on the evidence available depleted uranium sourced from reprocessing of spent nuclear fuel rods would not have any significant impact on the overall radioactivity of depleted uranium used in ammunition and armour (UNEP, 2001a; WHO, 2001a; WHO, 2001b; OSAGWI, 2001). On the basis of measured concentrations of the radionuclides in samples of depleted uranium, it has been calculated that in the event of an intake of depleted uranium, traces of transuranics and fission products would increase the radiation dose resulting from the uranium isotopes by less than one percent (WHO, 2001b).

Availability of Depleted Uranium

There is a plentiful supply of depleted uranium and considerable stockpiles exist. In June 1998, the US Department of Energy stored 734,000 metric tons of uranium-hexafluoride. Two-thirds of this - about 560,000 metric tons - is depleted uranium (ATSDR, 1999b). Figures for the "stockpiles" in other countries with enrichment facilities are not published. Together it is anticipated that they store at least again as much depleted uranium.

Section Three

Uses of Depleted Uranium

Nonmilitary Uses for Depleted Uranium

Depleted uranium has a wide range of non-military applications. The two main properties that have made it attractive are those that it shares with other heavy metals such as lead namely its high density and impenetrability to radiation. Additionally metallic uranium's high melting point (1132^{o}C), high tensile strength, chemical reactivity and pyrophoric nature make it useful in a number of arenas.

Depleted uranium compounds have a range of commercial applications which include (OSAGWI, 1998):

- ballast and counterweights;
- balancing control services on aircraft (civilian and military);
- balancing and vibration damping on aircraft;
- machinery ballast and counterweights;
- gyrorotors and other electromechanical counterweights;
- neutron detectors;
- radiation detection and shielding for medicine and industry;
- shielding for shipping containers for radiopharmaceuticals, radioisotopes, and spent nuclear fuel rods;
- chemical catalyst (uranium carbide may be used in the production of synthetic ammonia (Lewis, 1993);
- shielding in X-ray tubes; and
- in glass and ceramics manufacture, for brilliant colours.

Military Uses for Depleted Uranium

Depleted uranium has physical properties which make it useful for both defensive and offensive military purposes. Its high density, tensile strength and the autoignition of fragments at high temperature are the major features which have attracted munitions designers to this element. Its high density offers improved defence when used as armour shielding and both its density and its pyrophoric properties increase the penetration of armour piercing munitions. Depleted uranium munitions are regarded as conventional weapons and as such is not subject to the requirements of treaties relating to nuclear weapons. The United States, United Kingdom, Russia, Turkey, Saudi Arabia, Pakistan, Thailand, Israel, and France are developing or already possess weapon systems that contain depleted uranium in their inventories (OSAGWI, 2000).

Depleted uranium may be applied defensively as in the armour system of some tanks such as the Heavy-Armour variant of the Abrams M1A1, which has sheets of depleted uranium enclosed in steel plate. The layered steel and depleted uranium protects against penetration by projectiles made of less dense metals, such as tungsten-carbide (OSAGWI, 2000).

Depleted uranium may be used offensively in cartridges and munitions rounds (penetrators) to pierce armoured targets. For this purpose the depleted uranium is alloyed with 2% molybdenum or 0.75% titanium. The depleted uranium alloy is then used to form a long rod that is held within a sabot (a lightweight carrier designed to centre a smaller calibre projectile). The sabot is discarded after the round is fired. These munitions carry no explosive charge but do have a considerable energy of motion (kinetic energy). Large calibre (120-mm), tank-fired depleted uranium rounds have a muzzle velocity of 1.5 kilometres per second; at this velocity, the kinetic energy of a 5 kilogram depleted uranium penetrator is equivalent to 1.4 kg of TNT. The kinetic energy of a 30-mm Gatling-gun penetrator (with about 275 grams of depleted uranium) is equivalent to about 50 grams of TNT (Fetter and von Hippel, 1999). The kinetic energy of the high speed, very dense depleted uranium penetrator is sufficient to punch a hole in the complex armour of a modern tank.

Upon impact with a hard target such as an armoured tank depleted uranium munitions maintain their form better than those of tungsten or steel; additionally the penetrator "sharpens" itself on impact, in contrast to tungsten projectiles, which tend to blunt (OSAGWI, 2000). The impact of a depleted uranium penetrator on its target causes a proportion of its kinetic energy to be quickly (< 1 millisecond) converted into heat. This rapid release of energy can convert much of the depleted uranium into small, hot fragments and particles. The particles and smaller fragments burn; increasing its destructive effects and generating depleted uranium oxide aerosol. When a penetrator strikes a soft target, such as a personnel carrier, truck, or soil, much less aerosol is generated and much of the metal depleted uranium penetrator may remain intact. When such a projectile intercepts a tank this may lead to ignition of its fuel tank or detonation of the stored ammunition (OSAGWI, 2000). Additionally, conventional explosive munitions may also be fired with the depleted uranium munitions to take advantage of the penetrance offered by the depleted uranium rounds.

Military Assessments of Depleted Uranium Munitions and Armour

Depleted uranium munitions had been developed and trialed across several decades. The history of the development of depleted uranium munitions and a range of those currently in use is at Appendix Six. The first operational use of depleted uranium munitions occurred in 1991 during Operation Desert Storm (the Gulf War). Depleted uranium was used in the Gulf War as a layer in the armour system on heavy armour tanks and in a number of offensive weapons. Before the Gulf War it had been determined that these weapons would be superior to the alternative tungsten penetrators, particularly with respect to the destruction of Soviet-built T72 tanks used by the Iraqi forces. During the Gulf War US and UK forces are known to have had depleted uranium munitions, Iraqi forces however did not have access to such weapons.

The US military has published most of the available comment on use and effects of depleted uranium in the military context. The military use of depleted uranium as a protective layer in the armour system of some tanks and in munitions in the past decade appears to have contributed to tactical superiority in active combat situations against forces without access to such weapons. No conflict to date has seen depleted uranium weapons systems available to all combatants though this must be an anticipated occurrence.

No US tank protected by a depleted uranium defensive system was lost to or penetrated by enemy fire during the whole of the Gulf conflict (OSAGWI, 2000). The unit histories from the Gulf War contain many anecdotes attesting to the effectiveness of depleted uranium "silver bullets." "One Armour Brigade Commander described looking on in "amazement" as his soldiers (who in training had never fired at targets beyond 2,400 meters) routinely scored first-shot kills on targets out to 3,000 meters and beyond." (OSAGWI, 2000 Tab F p1)

Depleted uranium armour gained an equally impressive reputation. A story illustrating depleted uranium's offensive and defensive capacity involves an M1A1 tank, that had become trapped in mud, but was protected by a depleted uranium defensive system ((OSAGWI, 2000 Tab F p1).

> The unit (part of the 24th Infantry Division) had gone on, leaving this tank to wait for a recovery vehicle. Three (Iraqi) T-72's appeared and attacked. The first fired from under 1,000 meters, scoring a hit with a shaped-charge (high explosive) round on the M1A1's frontal armour. The hit did no damage. The M1A1 fired a 120mm armour-piercing round that penetrated the T-72 turret, causing an explosion that blew the turret into the air. The second T-72 fired another shaped-charge round, hit the frontal armour, and did no damage. The T-72 turned to run, and took a 120mm round in the engine compartment and blew the engine into the air. The last T-72 fired a solid shot (sabot) round from 400 meters. This left a groove in the M1A1's frontal armour and bounced off. The T-72 then backed up behind a sand dune and was completely concealed from view. The M1A1 depressed its gun and put a sabot round through the dune, into the T-72, causing an explosion.

Depleted Uranium: Use in Australia

There is no record of the use of depleted uranium munitions on Australian soil. U^{238} (depleted uranium) metal was used in a number of trials of individual components or sub-assemblies of nuclear weapons associated with British Nuclear testing in Australia in the 1950s and 1960s. The report of the Royal Commission into British Nuclear Tests in Australia indicates that over eight tonnes of deleted uranium was used in part of the "minor trials" at Emu and Maralinga. Most of the depleted uranium was used at the Kuli location with 6.6 tonnes used in the TM 4 Tims trial between 1956-1960 and 0.73 tonnes in the TM 16 Tims trial between 1960 – 1963 (Royal Commission into British Nuclear Tests in Australia, 1985).

Depleted Uranium: In the Gulf War

Large and small calibre depleted uranium munitions were first used in combat during the Gulf War. Large calibre depleted uranium munitions were fitted in battle tanks such as the British Challenger II and the American M1A1 Abrams. These 105 and 120-mm rounds have penetrator rods with depleted uranium masses of four to five kilograms. The US Air Force used small calibre 30-mm depleted uranium armour piercing munitions for the Gatling gun mounted on the A-10 "Warthog" aircraft. These small calibre rounds each contain about 275 grams of depleted uranium. At that time other offensive weapons containing depleted uranium were available to the US Marine Corps (AV-8B Harrier aircraft and tank munitions) and US Navy (Phalanx Close-in weapon system) (OSAGWI, 1998; Fulco et al, 2000).

Overall, approximately 320 tons of depleted uranium munitions were expended during the Gulf War over a total area in excess of 10,000 km^2 (Fetter and von Hippel, 1999). The US Army used about 9,500 large calibre (120-mm) anti-tank rounds containing depleted uranium during the Gulf War, many as training and practice rounds. About 3,000 large calibre rounds were destroyed, many in a fire at Camp Dohoa, Kuwait (Harley et al, 1999, tables G.1 and G.2). An estimated 783,500 small calibre (30-mm) depleted uranium rounds were fired by A-10 aircraft in this conflict. US Marine aviators used eleven tons of 25-mm depleted uranium munitions. Use of depleted uranium munitions by the US Navy and British forces was relatively minor (OSAGWI, 1998; Fulco et al, 2000).

During the Gulf War some Allied military personnel are known to have received exposure to depleted uranium-oxides in aerosols, dusts and retained fragments. US Abrams tanks mistakenly fired depleted uranium penetrators into US combat vehicles, destroying or damaging them. US DoD has reported that US depleted uranium munitions struck 15 Bradley Fighting Vehicles and 14 Abrams Tanks (Livengood, 1998). In addition, personnel recovering, repairing, or

decommissioning vehicles damaged by depleted uranium penetrators may have inhaled or ingested residual depleted uranium fragments and particles. There were also several accidental tank fires and an ammunition explosion and fire at Camp Doha, Kuwait, which burned, oxidized and fragmented many depleted uranium rounds. Other personnel entered Iraqi vehicles destroyed or damaged by depleted uranium. No relevant record of the experience of surviving Iraqi personnel was identified.

These exposure scenarios are further considered in Section Four and are the subject of further US DoD research to delineate any potential risk which may be associated at the dose levels attained (OSAGWI, 2000). A subset of heavily exposed US military personnel with retained depleted uranium fragments are also the subjects of ongoing assessment (McDiarmid et al, 1999; McDiarmid et al, 2000; McDiarmid, 2001)

Depleted Uranium: In the Balkans War

United Nations (UN) intervention occurred in the Balkans in 1991 and the first peace keepers were deployed to the region at this time. Continued regional and ethnic fighting has seen repeated UN and NATO military response.

NATO military intervention has included airborne bombing on a number of occasions. No large calibre depleted uranium munitions have been utilised in this conflict, there is evidence that small calibre (20-mm - 30-mm) depleted uranium munitions have been used over two periods of time by airborne NATO forces.

Bosnia

Only small calibre depleted uranium munitions are known to have been utilised. In Bosnia in 1994-1995, NATO forces fired about 10,000 30-mm rounds (2750 kilograms) from Gatling guns mounted on A-10 aircraft. A list of target sites is included in Appendix Seven (a).

Kosovo

Military intervention by NATO forces in Kosovo began in March 1999. Air strikes were launched against Serbian military targets, and continued for 78 days. Operation Allied Force utilised a completely airborne force and inflicted massive damage while suffering no hostile fire casualties.

In Kosovo again, the only depleted uranium ammunition used was 30-mm rounds fired from Gatling guns mounted on A-10 aircraft. These were fired in conjunction with high explosive rounds in ratios of around 4:1 (WHO, 2000a).

One hundred and twelve attacks involving 31,000 depleted uranium rounds were made on 84 geographically different sites in Kosovo and environs in the period from 6 April to 11 June 1999. The number of rounds (both depleted uranium and high explosive) per location varied from 50 to 1300 (NATO, 2001). NATO data concerning the A-10 attack sites is at Appendix Seven (b). At the time of this bombing no NATO forces were known to be present in Kosovo. Since the bombing no systematic clean up of these sites has been carried out, although one NATO brigade collected 36 depleted uranium penetrators at one location.

On June 20, 1999, Operation Allied Force was officially terminated. The province was subsequently divided into peacekeeping zones where NATO troops were deployed to assist in enforcing law and in the restoration of the area's infrastructure. Most of the depleted uranium rounds were fired in what became the German (MNB South) and Italian (MNB West) sectors, predominantly in the border areas with Albania. Fewer attack sites are located in the American

(MNB East) and British (MNB Centre) sectors and only three locations (all about four to eight kilometres south of Mitrovicë/Mitrovica city) were attacked in the French (MNB North) sector (WHO, 2001a). According to the United Kingdom Ministry of Defence, very little of the depleted uranium munitions were expended in what is now the British sector of Kosovo.

Australian Defence Force Personnel in the Balkans

Approximately 260 Australian Defence Force personnel are recorded to have served with NATO forces during the Balkans conflict. Most Australian Defence Force personnel in the Balkans served with British units. These troops have served in a range of roles and for varying times.

Australian Defence Force sources state that "of the 254 ADF personnel identified, approximately 180 were members of the various Australian Contingents of Operation OSIER or Operation AGRICOLA and served 6 months in the Balkans. The remainder of these personnel served on Third Country Deployments to the Balkans with either German, Canadian, US, and UK forces for periods ranging from several days to several months." (Personal communication, 27/7/2001)

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Section Four

Human Uranium Exposure

As outlined in Section Two, uranium is a naturally-occurring radioactive element that is present in rocks, soil, rivers, oceans, plants and animals. All humans are exposed naturally to uranium, its isotopes and decay products.

Uranium compounds have different solubilities and this physical property determines the absorption and retention of the specific compounds. The solubility of the compounds varies from "soluble" to "insoluble" (see Appendix Three) and this affects the rate of dissolution, uptake and excretion by the body.

In this Section only exposure is considered and refers to the potential for external or internal exposure and not to dose. Dose cannot occur without exposure however other variables interplay to determine dose. An exposure may occur but if the agent is not absorbed no dose may ensue. It is the dose to the target organs and not merely exposure, which contributes to the attending risk for adverse outcomes.

Exposure Pathways

Uranium is present in the natural environment and in materials made from earth's natural substances. Found in air, water, and food, small amounts are consumed and inhaled by all people on the planet (Harley et al 1999). While ingestion in food and water is the usual pathway for uranium into the human body for the general population, inhalation is the most common route of intake in occupational settings. Occupational exposure may occur via hand mouth contamination and thus ingestion. Skin contact is not an important route of intake of uranium oxides into the body as insoluble uranium oxides have not been shown to pass through intact human skin (ATSDR, 1999b).

Ingestion

The uranium content of food and water has been extensively studied. Uranium is adsorbed onto the roots of plants and root vegetables are a primary source of uranium in the diet. Cereals and table salt also act as sources of dietary uranium. On average, every day each of us takes in between 0.9-1.5 μg of uranium from food and water and a very small fraction via inhalation (ATSDR, 1999b). Most daily intakes within a country span an order of magnitude (UNSCEAR, 2000). The radiation dose from ingestion of natural uranium is small and the world average for uranium ingestion is 0.00025 mSv per year (European Commission, 2001). This forms a small part of the average individual dose of ionizing radiation from other natural sources of 2.4 mSv per year as outlined in Appendix Five of this report. (UNSCEAR, 2000).

Higher rates of uranium intake have been reported for some populations, and studies in US, Canada and Finland have demonstrated areas in which uranium intakes of hundreds of micrograms per day occur. The potential for increased uranium intake is greater in those who eat foods grown in areas with elevated soil uranium or who have elevated uranium in drinking water (ATSDR, 1999b) particularly when taken in conjunction with elevated water carbonate levels. There is a wide range of concentrations from below 0.01 μg/l to in excess of 1500 μg/l water. Ingested uranium is poorly absorbed from the digestive tract even in its soluble forms. Only 0.2 to 2% of uranium ingested in

food or water is absorbed via the gastrointestinal tract and absorption depends on the biosolubility of the particular uranium compound, previous food consumption and concomitant exposure to oxidising agents (Wrenn et al, 1985).

Inhalation

Inhalation of environmental uranium provides a small component of most people's exposure to uranium but this is the primary route of exposure for a number of occupational groups. In workplaces where workers handle ores, concentrates or powders of uranium oxides these uranium ore and oxide dusts may be suspended in air and inhaled. The larger particles will settle by gravitation and yield a dust layer which may give rise to hand contamination, then hand to mouth contact and ingestion. The settled dust and deposited aerosol particles may later be resuspended when the surface is disturbed, together with other dust or soil.

Workers in occupations involved in the extraction and processing of uranium such as uranium mining and milling, uranium conversion and enrichment, uranium fuel fabrication, and uranium weapons production have recorded increased exposures to uranium (ATSDR, 1999b). The internal accumulation of uranium dusts with such occupational exposure is supported by data from post mortem examination studies (Kathren et al, 1989).

The amount of dust deposited in lung tissue after inhalation depends on factors such as particle size and shape, and breathing rates. Models have been developed by the International Commission on Radiological Protection (ICRP) which describe the behaviour of inhaled (and ingested) uranium. These models enable organ doses to be calculated from intake or excretion of uranium. The most recent models for the respiratory tract and for uranium absorbed into blood were published in 1994 and 1995 (ICRP, 1994; ICRP, 1995).

Skin Absorption

Skin absorption of uranium compounds has not been characterised in humans (ATSDR, 1999b). While available occupational studies provide little to support this as a route for absorption (ATSDR, 1999b), animal studies have demonstrated that topical application of soluble uranium compounds such as uranyl nitrate hexahydrate and ammonium uranyl tricarbonate to shaved, bare skin may lead to systemic absorption (de Rey et al, 1983). Rabbits have been shown to be more sensitive than rats or guinea pigs to the toxic systemic effects of dermal application of soluble uranium compounds (Orcutt, 1949a; Voegtlin and Hodge, 1953; ATSDR, 1999b; WHO 2001b). The relevance of these findings in humans is unclear as the concentrations of uranium which were applied to these animals were extremely high.

Pathways of Exposure to Depleted Uranium

Exposure to depleted uranium may occur via the same pathways as those for natural uranium, that is, by inhalation, ingestion or by skin exposure. Skin exposure reflects external exposure while ingestion and inhalation reflect internal exposure to depleted uranium. Additionally, other pathways for internal incorporation of depleted uranium occur, for example, during the Gulf War a number of US military personnel were injured in “friendly fire” incidents and have retained depleted uranium fragments in their bodies.

Pathways for exposure to depleted uranium that has been used in military operations are the same as those for natural uranium and are:

- Inhalation in smoke and dust;
- Hand to mouth contamination and ingestion of dusts;
- Contamination of wounds;
- Skin contact;
- Agricultural pathways through uptake by crops or grazing animals;
- Accumulation in drinking water.

These pathways may be more or less relevant for different populations. Acute and high level internal exposures may be experienced by some of those present when a depleted uranium penetrator hits a hard target. While the intake of food and drinking water may contribute to long term, low level exposure and may be of particular relevance to the population residing in the affected region.

No literature is available on occupational exposure to depleted uranium in non-combat or non-military situations. One assessment of accidental exposure following an aeroplane crash (depleted uranium counterweights combusted after crash) in Bijlmer (Amsterdam) in 1992 resulted in a small calculated dose to bystanders of 0.001 mSv (best estimate) with an upper value of 0.7 mSv (Uijt de Haag et al, 2000). This compares to an average individual dose of ionizing radiation from natural sources of 2.4 mSv per year, and the allowable dose for radiation workers of 20 mSv/year averaged over 5 years.

Depleted Uranium in the Field of Battle

Personnel may handle depleted uranium penetrators and fragments and in the absence of protective gloves this may contribute to external skin exposure. Additionally, crews in tanks with depleted uranium armour systems may also experience external exposure. Depleted uranium is not absorbed into the body during such exposure.

When depleted uranium munitions are fired they may miss their target, hit a soft target or hit a hard target. Penetrators that miss the target may penetrate two to three metres into the soil (depending on soil type) (WHO, 2001a). Penetrators may hit and pass through a soft target such as a lorry, and these enter the earth and remain in a solid mass at variable depths in the soil. When depleted uranium munitions hit a hard target such as an armoured tank or a rocky outcrop, a significant proportion of the depleted uranium forms into an aerosol. Sufficient heat is generated by the impact to ignite the aerosolized uranium metal. The penetrator fragments then burn vigorously at a high temperature and, in effect, melt their way through steel plating into the interior of an armoured vehicle. Once inside the confined space of a tank, the heat from the burning penetrator fragments ignites flammable vapours and munitions within the vehicle. The ignition of flammable components within a tank is reinforced by the use of high explosive rounds in conjunction with depleted uranium ones (WHO, 2001b).

On impact, from 10 to 35 percent (with a maximum of 70%) of the original depleted uranium metal forms an aerosol and undergoes spontaneous ignition and small, relatively insoluble particles of mainly uranium oxides, as well as fragments of metallic depleted uranium are formed. The percentage varies according to a number of factors, such as the hardness of the target, velocity and angle of impact, pathway through the target (ie, what it impacts - engine, depleted uranium armour, etc.) (Harley et al, 1999; OSAGWI, 1998). If the depleted uranium round easily penetrates a target, as it does conventional armour, less of it will aerosolise than it does when hitting a hard target, such

as laminated steel (Harley et al, 1999). After impact with a hard target, residual fragments of the penetrator and depleted uranium dust are deposited close to the target site (OSAGWI, 2000).

The main uranium compounds formed after a depleted uranium penetrator hits a hard target are uranium oxides. It is recognised that there is currently only limited data on the combustion and thermal oxidation of depleted uranium. Extensive studies on melt down products of uranium with concrete, from simulated nuclear reactor studies, show that the main product even at such high temperatures remains uranium oxide. Studies of high temperature chemical reactions show that uranium and metals such as aluminium, used as a casing for shells, can form vapour phase chemical species when combined with halides such as chlorine. This possibility of mixed compound formation during uranium combustion requires the presence of a halogen source which may exist if polyvinyl chloride is present and combusted. The formation and toxicity of such compounds is unknown.

Accidental dispersion of depleted uranium has occurred in military settings. During the Gulf War a large fire occurred in a depleted uranium store at Camp Doha, Kuwait (OSAGWI, 2000). While no literature is available on military plane crashes this could be a potential source of exposure in combat. One civilian disaster, the 1992 Bijlmer plane crash (Uijt de Haag et al, 2000), describes the combustion of depleted uranium ballast. However, in these fires large pieces of depleted uranium oxidise slowly and normally only a very small fraction is dispersed as respirable particles (Harley et al, 1999).

Both the impact of a depleted uranium penetrator on a target and the burning of depleted uranium produce depleted uranium oxide dusts or aerosol particles. The main oxidation product is depleted triuranium octaoxide (U_3O_8), but also small amounts of depleted uranium dioxide (UO_2) and depleted uranium trioxide (UO_3) will be formed (OSAGWI, 1998; OSAGWI, 2000). Most of the suspended aerosols will rapidly settle to the ground. Most of the depleted uranium dust will be deposited within a distance of 100m from the target (US Army Corps of Engineers 1997 in WHO, 2001b). Upon weathering, the non-oxidised small particles and surfaces of remaining uranium metal will also slowly oxidise to those three depleted uranium oxides over time (OSAGWI, 1998). It is estimated that a penetrator buried in soil would corrode to uranium oxide over about 500 years (WHO, 2001a)

Activity or surface winds may disturb the settled particles on the ground and resuspend and redistribute a fraction of them. The World Health Organisation report highlights that deposited uranium dust might slowly be transformed through environmental weathering processes into more mobile and soluble forms and dispersed in the environment by air currents. Uranium oxide dust in soil or in contact with carbonate rich water may enter ground water including well water (WHO, 2001a).

Levels of Exposure to Depleted Uranium

Precise measurements for depleted uranium exposures in the Gulf or Balkans War are not currently available. Various attempts have been made to model doses from exposure scenarios developed from US experience in the Gulf War. Three broad levels of exposure have been modelled (OSAGWI, 2000) and these have been assessed to a greater or lesser extent by other bodies (Royal Society, 2001). These levels were originally developed by US researchers for consideration of exposures of US personnel to depleted uranium during the Gulf War.

Table 4.1
Gulf War Exposure Scenarios to Depleted Uranium by Assessed Level
(OSAGWI, 2000)

"Level I includes veterans in or near combat vehicles at the time these vehicles were struck by depleted uranium munitions, or veterans who entered vehicles immediately after they were struck by depleted uranium munitions. These veterans could have been struck by depleted uranium fragments, inhaled DU aerosols, ingested depleted uranium residues, or had depleted uranium particles land on open wounds, burns, or other breaks in their skin—or any combination of these possibilities.

Level II includes veterans and a small number of DoD civilian employees who worked in and around vehicles (mostly friendly-fire wrecks) containing depleted uranium fragments and particles. These individuals may have inhaled depleted uranium residues stirred up (resuspended) during their activities on or inside the vehicles, ingested depleted uranium after transferring it from hand to mouth, or spread contamination on their clothing. Soldiers who were involved in cleaning up depleted uranium residues from Camp Doha's North Compound after the July 11, 1991, explosion and fires are also included in this group.

Level III is an "all others" group whose exposures were largely incidental or fleeting. This group includes individuals who entered depleted uranium –contaminated Iraqi equipment, soldiers downwind from burning Iraqi or US equipment struck by depleted uranium rounds, or soldiers downwind from burning DU ammunition (e.g., soldiers at Doha during the July 11 fire). While these individuals could have inhaled airborne depleted uranium particles, the possibility of receiving an intake high enough to cause health effects is extremely remote."

The approach has been broadened most recently in the UK with an analysis by the Royal Society which considers more generally the exposures in conflicts where depleted uranium has been or may be deployed (Royal Society, 2001). They delineate battlefield exposures to include:

Table 4.2
Military Exposure Scenarios to Depleted Uranium by Assessed Level
(Royal Society, 2001)

Level One exposure, being those to personnel in a vehicle when it is struck by a depleted uranium penetrator, or to those who entered the vehicle immediately in rescue operations. The exposure is predominantly through inhalation of depleted uranium and actual injury and retained depleted uranium fragments.

Level Two exposure, occurring post combat, for personnel working for hours on or in contaminated vehicles to carry out cleaning and repairs. The exposure is predominantly through inhalation of resuspended depleted uranium and by hand to mouth transfer and ingestion.

Level Three exposures include all other exposures such as being downwind of impacts or fires or brief (minutes) entry into contaminated vehicles

The US Gulf War scenarios have provided the core of the exposure levels outlined above. While many Iraqi tanks were destroyed at this time using depleted uranium penetrators, information on Iraqi survivors is lacking.

In Bosnia (1994-1995) and Kosovo (1999) the NATO forces used aerial bombing assaults and the personnel close to an impact would have been predominantly Serbian and Yugoslavian soldiers. Some of these troops could therefore have been exposed to depleted uranium aerosols and dusts by inhalation. Those personnel involved in immediate rescue of, for example, targeted tank crew could also have received depleted uranium exposure. The acute exposure of NATO forces to high ambient levels of depleted uranium dusts and smoke at the time of an attack is much less likely, particularly during the 1999 Kosovo conflict as this was an air-borne offensive by NATO forces.

Uptake and Excretion of Uranium

Most uranium entering the body by ingestion is not absorbed, but is eliminated via the faeces. After inhalation of uranium-containing dusts or aerosols, soluble uranium compounds are absorbed from the lungs within days while insoluble forms may take months to years to be absorbed into the systemic circulation.

Experimental evidence shows that about 90% of the absorbed uranium will be excreted via the kidneys and passed from the body in urine within a few days. The exact proportion depends on its chemical speciation in the blood. Normal excretion of uranium usually lies in the range between 0.04-0.5 µg /L of urine. The uranium-carbonate complex passes through the kidney and is filtered rapidly at the glomerulus where 60-80% of absorbed uranium is excreted in the first 24 hours after acute exposure. The uranium not excreted is reabsorbed by the proximal tubules and retention of uranium in the kidney has been attributed to the creation of uranium protein and uranium phospholipid complexes within the proximal tubules (Wedeen, 1992 in WHO, 2001b). Less than one percent of the excretion of absorbed uranium occurs through faecal route (ICRP-69, 1995). The time for half of the absorbed uranium to be excreted in the urine has been estimated to be in the range from 180 to 360 days.

Sites of Accumulation of Uranium in the Body

On average the human body contains about 100 µg of uranium. This is distributed in the body with about two-thirds in the skeleton, 16% in the liver, 8% in the kidneys and 10% in other body tissues, but can vary depending on the pattern and nature of exposure (ATSDR, 1999b).

Once in bone, uranium competes with calcium to form complexes with phosphate ions, thus becoming part of the bone matrix. This bone matrix then serves as a storage site from which uranium is slowly released back into circulation. Clearance from the bone is slow and half- lives of 300 and 5000 days have been estimated (Kathren et al 1989; ATSDR 1999b; WHO 1998b).

Animal studies indicate that at low doses uranium does not readily distribute to the central nervous system (CNS) but with higher doses (8mg/kg/day orally for 4 weeks) brain uranium levels are comparable to those in liver and in bone. Pellmar et al (1999a) have studied surgically implanted depleted uranium in rats and found that at one and six months post implantation, uranium levels were significantly increased in brain tissue compared to control rats.

Section Five

Toxicology of Uranium and Depleted Uranium

The toxicology of an agent may be defined as the scientifically derived information concerning its adverse effects in animals and humans. Such effects may vary by dose, route of administration, the chemical form of the specific agent, and across species.

There are relatively few studies specifically of the health effects of depleted uranium in animals or humans. There is however a considerable research base available on the health effects of natural and enriched uranium. Both have identical chemical properties to depleted uranium and both are more radioactive than depleted uranium. Indeed, enriched uranium has a specific activity about two orders of magnitude greater than either uranium or depleted uranium.

The toxicology of uranium has been comprehensively reviewed by a number of authoritative agencies and individuals. Studies of its toxicity have assessed both radiological and chemical effects including the consequences of inhalation, ingestion and parenteral administration of a variety of uranium forms in animal models and in humans.

One of the most comprehensive assessments of uranium's basic toxicology was assembled from work undertaken in conjunction with the Manhattan Project during the Second World War. This work, *"Pharmacology and Toxicology of Uranium Compounds"* in four volumes was edited by Voegtlin and Hodge and published in 1949 -1953. The literature on uranium toxicology was considered in detail in the 1999 revision of the Agency for Toxic Substances and Disease Registry publication *"Toxicological Profile for Uranium"* (ATSDR, 1999b). The human studies of exposure to uranium have been recently reviewed by several groups (Fulco et al, 2000; Royal Society, 2001; Harley et al, 1999; Durakovic 1999). These and other toxicological assessments have been used by this Expert Committee. The Expert Committee has not sought to reproduce the work already undertaken by such groups as the Agency for Toxic Substances and Disease Registry but rather summarise, highlight and update relevant findings.

Uranium has been studied with regard to both its chemical and radiation toxicity. The chemical toxicity of any given compound is related to the interaction of the compound with the biochemical processes of the human body. The chemical toxicity of uranium varies according to dose, chemical form and route of exposure.

Radiation toxicity is related to the effect radioactive substances exert by transferring energy (as ionizing radiation) to exposed tissue. Uranium is predominantly an alpha emitter, which means that it could only effect cells in immediately adjacent tissue. For example, by inhalation and retention in the lungs or deposition through metabolic pathways in bone. Through close range exposure, alpha emitters may damage cellular DNA, thereby leading to mutation and cancer. In contrast to chemical toxicity, radiation effects are theoretically more likely to be associated with the insoluble compounds, such as uranium dioxide particles, because they are likely to remain at the site of deposition, for extended periods of time. Although radiation exposure has been assumed in some models to be carcinogenic at all dose levels this has not been established at low doses (ATSDR,

1999b). There is little specific information on uranium (or depleted uranium) and reproductive or developmental effects (ATSDR, 1999b).

Animal Studies

Animal experiments have been considered in three broad groups. The first two groups, nonmalignant effects and carcinogenesis deal with exposure to a variety of uranium compounds via inhalation, ingestion or skin contact. The third group covers studies dealing with the effects of implanted depleted uranium metal.

Nonmalignant Effects

The 1999 ATSDR review summarised the findings of 119 inhalation, 84 ingestion and 37 skin exposure assessments in a range of animal species. They examine minimal effect levels for a variety of systemic effects including death. Animal experiments use exposures which produce doses orders of magnitude greater than that of most documented human exposures and the relevance of such studies to human risk may thereby be limited. It has been calculated, for example, that a man would have to inhale nearly 2,000 mg of uranium per day to achieve the same blood levels as the lowest exposure (0.15 mg) animals (mouse) in feeding studies (Harley et al, 1999).

Studies in a number of species confirm that the kidney is the major target organ associated with exposure to uranium compounds. The effects, where present, are of acute onset occurring within days of exposure and range from microscopic lesions to severe necrosis in the proximal renal tubules and death (Voegtlin and Hodge 1953, ATSDR, 1999b).

The more water soluble uranium compounds (uranyl nitrate hexahydrate, uranium hexafluoride, uranyl fluoride, uranium tetrachloride, and uranium pentachloride), have been shown to enter the systemic circulation, and are the most potent renal toxins. The less soluble compounds (sodium diuranate, ammonium diuranate) are of moderate to low renal toxicity, and the insoluble compounds (uranium tetrafluoride, uranium peroxide, and uranium oxides: uranium trioxide, uranium dioxide and triuranium octaoxide) have little potential to cause renal toxicity (ATSDR, 1999b).

Animal experiments of inhalation, ingestion and skin contact have demonstrated that some uranium compounds at high dose levels can cause a range of systemic effects. Acute respiratory effects have been described after inhalation studies using soluble uranium compounds (Leach et al, 1984). Insoluble compounds including uranium oxides have not shown this acute toxicity (ATSDR, 1999b). Several long term, high dose studies have demonstrated pulmonary, and pulmonary lymph node fibrosis in some but not all species exposed to inhaled uranium dioxide (Leach et al, 1970, 1973). No respiratory effects have been reported after oral or skin administration (ATSDR, 1999b).

Non-malignant neurological effects including muscle weakness, gait instability and acute cholinergic symptoms have been reported after inhalation and oral ingestion of high doses of uranium (Dygert et al, 1949; Domingo et al, 1987). Other studies have not found these effects (Gilman et al, 1998a). Two early studies of toxic sub-lethal and lethal doses of soluble uranium given by injection to dogs and rabbits showed degeneration in the cerebral and cerebellar cortices (Purjesz et al, 1930; Verne, 1931). This occurred after evidence of other systemic toxicity and usually just before the animal's death (Fulco et al, 2000). The effects of implanted depleted uranium pellets are being considered in a range of animal studies detailed in the following section.

A small number of studies have examined reproductive or developmental effects after ingestion of soluble uranium compounds. No studies concerning reproductive effects after inhalation of uranium

oxides were identified. Intermediate studies of ingestion of uranium as uranyl nitrate in rats and rabbits have shown no changes in reproductive effects or reproductive organ weights (Gilman et al, 1998a; Gilman et al, 1998b; Gilman et al, 1998c). Llobet et al (1991) found male mice treated with high doses of uranyl acetate dihydrate had normal testicular size and spermatogenesis, but a non dose related pregnancy rate after mating. Chronic duration studies on rats have shown testicular degeneration after two years of high oral doses of uranyl nitrate hexahydrate (Maynard et al, 1953).

Dose related developmental effects have been observed with high dose oral uranyl acetate dihydrate administered during gestation. Maternal toxicity was evident in all 20 pregnant mice. A no-effect-level of 5 mg uranyl acetate dihydrate/kg/day during pregnancy was suggested (Domingo et al., 1989). No studies were identified which examined developmental effects associated with uranium oxide ingestion. No studies were located that reported genotoxic effects in animals following oral exposure to uranium for any duration.

Irritation of the skin has been noted after high dose dermal, but not oral or inhaled uranium exposure. Haematological effects have been recorded after some but not all, high dose studies of exposure. Rats inhaling uranium have exhibited decreased red cells and increased myeloblasts and lymphoid cells however in other experiments injected and oral uranium has been shown to have no haematological effects (Fulco et al, 2000). Cardiovascular, gastrointestinal and hepatotoxic effects are not prominent in animal experiments by any route of exposure.

These animal experiments have found uranium to have a low order of metallotoxicity in mammals. It is less toxic than other heavy metals, such as lead, arsenic and mercury at the same dose level (ATSDR, 1999b). Dose levels of uranium are estimated for no-observed-adverse-effect levels (NOAELs) and "less serious" and "serious" lowest-observed-adverse-effect levels (LOAELs) (ATSDR, 1999b). Appendix Eight contains summary information concerning animal studies involving inhalation, ingestion and dermal exposure to a variety of uranium compounds and the ATSDR (1999b), NOAELs and LOAELs for acute and chronic animal studies involving inhalation, ingestion exposure to a variety of uranium compounds.

Carcinogenic and Mutagenic Effects

A small number of experimental studies have specifically considered the carcinogenic or mutagenic potential of uranium or depleted uranium.

Chronic inhalation studies on beagle dogs (32 exposed, 13 with five year exposure and six control) and monkeys (eight exposed, six with five year exposure and five control) maintained for up to 6.5 years after exposures of 1-5 years to UO_2 aerosol at a concentration of 5.8 mg/m^3 and approximately one micrometer median particle diameter were undertaken by Leach et al, 1973. The magnitude of the uranium concentration at the end of the five year exposure was approximately 2,000 µg U/gm (dogs), and 4,000 µg U/gm (monkeys) in the lungs. Leach estimated the lung burden in dogs to be 151 mg of uranium at the end of the exposure period. The estimated radiation dose from the experiment (alpha radiation accumulated over the five year exposure plus 2 - 6.5 years post exposure) to the dog lungs was 600-700 rads (6-7 Gray). This study conveyed a considerable loading of particulate matter and internal radiation to the lungs of the exposed animals. Pulmonary neoplasms (2 adenoma and 2 adenocarcinoma) and atypical epithelial proliferation were noted in nine of 13 dogs, while only one of five controls had bronchiolar proliferation. This finding in dogs has not been replicated in the same or other species of animals and may represent an idiosyncratic effect in dogs. Extensive lung fibrosis but no pulmonary neoplasms were seen in the seven maximally exposed monkeys (Leach et al, 1973).

The magnitude of the uranium concentration in the tracheobronchial lymph nodes at the end of the five year exposure was approximately 60,000 μg U/gm (dogs), and 70,000 μg U/gm (monkeys). Most dogs and all monkeys in the maximally exposed group demonstrated fibrosis of the tracheobronchial lymph nodes. Monkeys displayed replacement of lymphoid tissue with pigment and fibrosis. Lymphoma was noted in one of seven exposed monkeys while no abnormalities were noted in the tracheobronchial nodes of five control monkeys. No lymphoma was noted in any of the exposed dogs. The estimated cumulative radiation dose to the nodes of maximally exposed monkeys was at least 7000 rads (70 Gray) (Leach et al, 1970).

Golden Syrian hamsters (four study groups each with 102 animals) inhaling 19 mg U/m^3 as uranium ore dust for 16 months showed no apparent increase in tumours of the liver, kidney, spleen, trachea, lungs or heart compared to unexposed controls (Cross et al 1981a). In the hamsters, uranium ore dust provoked inflammatory and proliferative responses and alveolar hyperplasia but these changes did not progress with time. Long term, high dose feeding studies in rats, rabbits and dogs did not find evidence of cancer occurrence (Maynard and Hodge, 1949).

In vitro studies of depleted uranium have suggested that depleted uranium may cause human cell transformation to a neoplastic phenotype, with an effect level comparable to other biologically reactive carcinogenic heavy metal compounds such as nickel (Miller et al, 1998a). To assess potential mutagenic effects of implanted depleted uranium, Sprague-Dawley rats were allotted to one of five experimental groups (Miller et al, 1998b and for description of groups see Pellmar et al, 1998 above). Mutagenic potential of uranium was assessed at 0 days, 6, 12 and 18 months after implantation. Urine from depleted uranium implanted animals enhanced mutagenic activity in *Salmonella typhimurium* strain TA98 and Ames II mixed strains in a dose and time dependent manner. The sera from animals implanted with depleted uranium or tantalum did not enhance mutagenicity in any bacterial strain (Miller et al, 1998b).

The majority of animal studies considering the potential for carcinogenic effects have not found an association with uranium exposure. One series of experiments conveying protracted, high level exposure to inhaled uranium dusts found evidence that internal, high dose alpha radiation may contribute to tumor development.

Implanted Depleted Uranium Pellets

Prior to the Gulf War the effects of the implantation of depleted uranium fragments had not been studied in animal models because it had been assumed the fatality rate would approach 100% for personnel inside vehicles hit by depleted uranium munitions (Livengood, 1998). Experience has now shown that this is not the case and since the Gulf War several animal studies have been undertaken to consider the redistribution of uranium and toxicity of embedded depleted uranium fragments.

The US Armed Forces Radiobiology Research Institute (AFRRI) research program began to assess the retention of depleted uranium in the body in 1993. Under this program, depleted uranium pellets have been implanted into rats in an attempt to develop an animal model which could assist in the management of individuals with retained depleted uranium fragments. The current status of the program and findings to date have recently been reviewed (McClain et al, 2001).

In the AFRRI animal experiments, male rats had depleted uranium pellets surgically implanted in their legs. Surgical controls were implanted with inert tantalum pellets. The number of uranium pellets implanted controlled the dose of depleted uranium. Six, twelve and eighteen month uranium levels were high and dose-dependent in the kidney, urine, and bone in depleted uranium implanted rats, but no physiological or histological evidence of kidney damage was evident (Pellmar et al,

1998; Pellmar et al, 1999a). The concentration in the most highly exposed rats was above the level of uranium known to be acutely toxic (Pellmar et al, 1998). Bone and kidney were the primary reservoir for uranium, with deposition also in the brain, lymph nodes, testes and spleen (Pellmar et al 1999a). Uranium concentrations rose in the brain in a dose and time dependent manner. Uranium was found in the hippocampus of the brain of depleted uranium implanted rats, but no neurotoxicity was evident (Pellmar et al, 1998; Pellmar et al, 1999a; Pellmar et al, 1999b). The authors concluded that "these data suggest that renal toxicity may be less of a hazard than originally anticipated. However, cognitive deficits need to be considered." (Pellmar et al, 1998 p9)

The elevation of uranium levels in the brains of these rats has been the focus of further research. Pellmar et al (1999b) studied electrophysiological changes in the hippocampus of depleted uranium implanted rats. At twelve months, significant differences in electrophysiological measures were evident between depleted uranium implanted and control (tantalum implanted) rats. However there was no difference found in locomotor, discrimination learning or results of general functional measures between these two groups (Pellmar, 1999a). At eighteen months the difference in electrophysiology between the depleted uranium implanted and control rats, which had been noted at twelve months, was no longer present. These animal studies suggest that while uranium does accumulate in the brain of depleted uranium implanted rats, there is no clear pathophysiological significance of this accumulation.

At six months, urinary uranium concentrations in the three dose groups of exposed ranged from 46 to 674 µg/l. The highest concentrations of uranium measured in the urine of Gulf War veterans has been approximately 30 µg/l and in uranium miners near 200 µg/l (Pellmar et al, 1998). Harley et al (1999), note that “comparisons of exposure levels between rats and humans are difficult, but … the rats with the smallest number of depleted uranium pellets had a higher urinary uranium level than Gulf War veterans with embedded fragments, who were among the most highly exposed Gulf War veterans.”

AFRRI has also examined the effects of implanted depleted uranium pellets on reproduction in female rats, measuring various maternal and litter parameters, including pup weight and litter size. Results from mother rats with high levels of uranium indicate the pups are normal (Harley et al. 1999).

Human Studies

ATSDR (1999b) has analysed the epidemiological information on uranium exposure in humans by specific route and duration of exposure (acute < 14 days; and chronic 365+ days). No human data were available on exposure duration 15-364 days. (ATSDR, 1999b). The human epidemiological studies relate primarily to occupational settings (uranium miners, millers and processors) where inhalation, ingestion (via hand to mouth contamination) and skin exposures may occur. There is little information regarding the health effects of uranium in humans from ingestion or from skin exposure alone.

Acute Human Exposure to Uranium

Epidemiological studies demonstrate that routine exposure to airborne uranium is not associated with increased mortality in human populations (ATSDR, 1999b; Fulco et al, 2000; Royal Society, 2001). Severe, acute-onset respiratory impairment and deaths from acute exposure to high concentrations of airborne uranium hexafluoride have been reported in humans. However the deaths were attributed to the impact of the explosions at a uranium processing plant which released

uranium hexafluoride and consequent production of highly toxic hydrofluoric acid (Moore and Kathren, 1985; Kathren and Moore, 1986).

Acute renal toxicity has been documented following poisoning incidents (ATSDR, 1999b). One was an inhalation exposure to powdered uranium tetrafluoride (Zhao and Zhao, 1990), and the other an intentional ingestion of approximately 131 mg U/kg as uranyl acetate (Pavlakis et al, 1996). A volunteer who ingested one gram of uranyl nitrate (equivalent to 470 mg uranium) experienced vomiting, diarrhoea and slight albuminuria. Fulco et al, 2000 report that in one study (Morrow et al, 1980), ingestion of uranyl nitrate at doses of 925 mg three times daily did not cause abnormalities on urinalysis. Four patients given 10.8 mg of uranyl nitrate had no alteration in kidney function (Stopps and Todd, 1982).

Bernard (1958) administered terminally ill patients between 4-50 mg of uranium compounds intravenously. Transient proteinuria occurred in patients receiving doses greater than 0.071 mg/kg. At autopsy there was no evidence of acute damage to the renal tubules. Hodge et al (1973) described six patients who were given intravenous uranyl nitrate (0.0063-0.109 mg /kg) over a 1-2 day period. Transient proteinuria occurred 4-6 days after injection in the two patients given the highest dose (greater than 0.042 mg/kg). Human case studies from accidental and chronic exposure demonstrate renal function returns to normal after exposure has ceased (Kathren and Moore, 1986; Eisenbud and Quigley, 1956). This is consistent with animal studies which demonstrate regeneration of damaged tubular epithelium when exposure to uranium is discontinued and sometimes despite continued exposure (Dygert, 1949; Rothstein, 1949; Stokinger et al, 1953).

No studies are available describing the effects of intermediate duration exposure to uranium in humans for any route.

Chronic Human Exposure to Uranium

Chronic exposure to uranium has been examined in a number of occupational cohort studies. Inhalation is the predominant route of exposure in these studies. While oral and dermal exposures occur in concert with inhalational exposures the poor absorption from these routes limit their contribution to total dose (ATSDR, 1999b).

A number of reviews have preceded this current examination of the data. Several comprehensive assessments have already been made of the literature relating to the health of uranium miners (ATSDR, 1999b; Fulco et al, 2000); and workers at plants processing uranium ore for use in nuclear reactors and nuclear weapons (Fulco et al, 2000; Darby and Beral, 2001).

Uranium Miners

Uranium exposure in mining arises largely from uraninite which is essentially uranium oxide (UO_2). These uranium oxides are similar to those formed after hard target impact of depleted uranium penetrators. Any effect of uranium in underground miners is however confounded by concurrent exposure to radon gas and progeny, which are known to be carcinogenic to the lung (IARC, 2000).

The mortality experience of several mining populations has been assessed in a series of studies. Underground uranium miners in the US states of Colorado, Utah, New Mexico and Arizona (the Colorado Plateau states) have been identified as a research cohort for several decades (Lundin et al, 1969; Saccomanno et al, 1971; Saccomanno et al, 1976; Saccomanno et al, 1986; Archer et al, 1973b; Archer et al, 1976; Auerbach et al, 1978; Band et al, 1980; Whittemore and McMillan 1983; Hornung and Meinhardt 1987; Roscoe, 1997).

In the first of these studies Lundin et al (1969), described a 50% increase in mortality in 3,414 miners who had worked underground in the period 1950-1967; with the excess due to violent deaths and respiratory cancer. The risk of respiratory cancer increased with cumulative estimated exposure to radon progeny. The study also described a statistically significant decrease in cardiovascular-renal disease mortality. A number of subsequent studies have continued to examine respiratory cancer in these workers. Saccomanno et al, (1986) found that exposure to radon gas increased the risk of lung cancer. Cigarette smoking has been shown to further increase this risk (Band et al, 1980).

In the most recent assessment of mortality, (Roscoe et al, 1997) lung cancer mortality was elevated with a standardised mortality ratio (SMR) of 580 (95% CI=520-640). Again the risk of respiratory cancer increased with cumulative exposure to radon.

Several studies have assessed lung cancer incidence or mortality in Navajo men involved in underground uranium mining (Gottlieb and Husen, 1982; Samet et al, 1984; Roscoe et al, 1995). Case control (Samet et al, 1984) and cohort (Roscoe et al, 1995) studies have demonstrated significant associations between lung cancer and uranium mining in those men.

Woodward et al (1991), reported similar findings for 2,574 workers from the Radium Hill uranium mine, (South Australia), identified from mine records. Exposures to radon progeny were estimated from historical records of radon gas concentrations in the mine and from individual job histories. Lung cancer mortality was increased relative to the general population of the period (SMR = 194, 95% CI=142-245). Compared with surface workers, lung cancer mortality was markedly increased in the underground workers with radon exposures greater than 40 Working Level Months WLM (relative risk = 5.2, 95% CI = 1.8-15.1).

Studies of Canadian underground uranium miners (Muller et al, 1985; Nair et al, 1985; Howe et al, 1986; Kusiak et al, 1993) again demonstrate an increase in lung cancer and in one a highly significant linear relationship between estimated radon exposure and risk of lung cancer was found (Howe et al, 1986).

Among 4,320 Czechoslovakian uranium miners (Tomasek et al, 1993; Tomasek et al, 1994) lung cancer mortality was significantly elevated (SMR=508; 95%CI=471-547) but mortality from all other cancers combined was not (SMR= 111; 95%CI=98-124) (Tomasek et al, 1993). Mortality was greater than expected for accidents, homicide, mental disorders, cirrhosis and non-rheumatic circulatory disease. Despite high levels of exposure to radon, arsenic and dust, mortality from chronic respiratory disease showed only a small and non-significant increase (SMR=121). Deaths from genitourinary disease were fewer than expected (SMR=77). (Tomasek et al, 1994)

The studies on uranium miners are of very limited relevance to the health effects of exposure to depleted uranium because the miners were exposed for prolonged periods to high concentrations of radon gas as a consequence of working underground in confined spaces.

ATSDR (1999b) and others have concluded that the consistent increases in lung cancer incidence and mortality were due to radon gas and its progeny, rather than uranium metallotoxicity or uranium radioactive emissions. In addition, miners were known to have been exposed to other possibly toxic dusts, diesel gas, and cigarette smoke. Some studies of underground miners have described increased risks for other types of cancer and chronic respiratory disease but results are inconsistent and multiple potential confounders exist.

Uranium Process Workers

Since the 1940s workers have been engaged in the extraction, milling and machining of uranium for the production of nuclear power and nuclear weapons. Workers in the nuclear industry are generally not exposed to radon gas and its progeny and so studies of these workers are not confounded in the same way as the studies of miners. However, uranium workers may be exposed to enriched uranium, other radionuclides and other toxic chemicals which may themselves convey adverse health effects.

There are some similarities between exposures to uranium in industrial settings and exposures that may occur in the battlefield. Both carry the possibility of inhalation and ingestion of uranium dust and aerosols. On the other hand, some authors have highlighted differences between occupational uranium exposures and battlefield depleted uranium exposures, such as the high temperature aerosol formation of depleted uranium compounds (Durakovic, 1999; Royal Society, 2001).

The recent US National Academy of Sciences Institute of Medicine assessment of depleted uranium and uranium in "*Gulf War and Health*", (Fulco et al, 2000) provided a detailed review of studies of process workers whose occupations would be likely to have involved exposure to uranium. Beral and Darby (2001) considered those cohort studies analysed by Fulco et al (2000) and those published since that report. After attempting to eliminate overlap between studies, they undertook a meta-analysis of 14 major studies of the health of uranium industry workers. Eleven were undertaken in the US and three in the UK (Beral et al, 1988; McGeoghegan and Binks, 2000a; McGeoghegan and Binks, 2000b). All 14 are summarised in Table A7.1 of Appendix Nine.

Of the 14 studies 12 involved workers in the nuclear industry. Additionally, Stayner et al (1985), report on workers employed in the processing of phosphate ore which involves release of uranium compounds. Teta and Ott (1988) considered workers at a plant that had participated in the Manhattan project, where uranium and other radionuclides were detectable.

It has been estimated that the 14 studies considered in the meta-analysis include the majority of the US and UK workers exposed to uranium in the nuclear industry during the last 50 years. In total approximately 120,000 workers were study subjects and 33,000 deaths have been reported from this total cohort. Causes of death were analysed against the expected death rates in the general population.

There are some acknowledged problems in interpreting the data from such retrospective, occupational cohort studies. These include the lack of reliable data on exposure for workers particularly in the studies dealing with the earliest years of the nuclear industry, when occupational radiation exposure was likely to be highest. Workers may be exposed to a number of radiation sources including enriched uranium and other radionuclides. There are only limited data concerning potential confounders including cigarette smoking and other chemical exposures. Despite large sample size, such studies have only limited ability to detect very small risks, and an unquantified healthy worker effect. Nevertheless the meta-analytic method used by Beral and Darby (2001) greatly improves the power to detect an increase in risk. The summary table for the meta-analysis is at Table 5.1.

Information available concerning the extent of external and internal radiation exposure to workers from radionuclides has improved over time. Six of the earliest studies had no information on the level of uranium exposure received by individuals (Hadjimichael et al, 1983; Waxweiler et al, 1983; Stayner et al, 1985; Brown and Bloom, 1987; Teta and Ott, 1988; Cragle et al, 1988). Three studies monitored external radiation doses (McGeoghegan and Binks, 2000a; McGeoghegan and Binks,

2000b; Dupre-Ellis, 2000). Frome et al (1997), and Beral et al (1988) had information on external doses and on whether or not an individual had been monitored for internal exposure to uranium or other radionuclides. A further two studies had estimated internal lung doses from alpha radiation produced by radionuclides (Dupree et al, 1987; Ritz et al, 2000). Ritz (1999a) had monitored external doses and estimated internal lung doses (Beral and Darby, 2001).

In the studies where monitoring data are available authors provided some form of dose response analysis for ionizing radiation exposure for a number of different cancer types. In these analyses a number of positive, if inconsistent, findings have emerged. A significant increase in prostate cancer was seen in UK workers exposed to uranium and other radionuclides (Beral et al, 1988). A significant dose-response relationship was described between external monitored radiation dose (with 10 year lag period) and 'all cancer' mortality (Frome, 1997 and Ritz, 1999); Hodgkins disease and bladder cancer (McGeoghegan and Binks, 2000a); renal cancer (Dupree-Ellis, 2000); and lung cancer (Frome, 1997; Ritz, 1999). A significant dose response between internal lung dose and upper aerodigestive tract cancers and haematopoietic and lymphatic cancers was described (Ritz, 2000). No study has reported a significant dose-response relationship between internal lung dose and mortality from lung cancer (Beral and Darby, 2001).

Table: 5.1
Summary table of mortality in uranium workers compared with the mortality of the general population (Figure 17, Beral and Darby, 2001)

Cause of Death	Total Number of Deaths	O/E (95% Confidence Interval)
All causes	33502	0.86 (0.79-0.93)
All cancer	7442	0.91 (0.85-0.97)
Stomach cancer	365	0.76 (0.62-0.89)
Colorectal cancer	728	0.91 (0.78-1.04)
Liver cancer	123	0.84 (0.62-1.07)
Lung cancer	2846	0.94 (0.83-1.05)
Bone cancer	30	0.93 (0.53-1.33)
Prostate cancer	490	0.98 (0.89-1.07)
Bladder cancer	196	0.83 (0.71-0.96)
Kidney cancer	151	0.78 (0.59-0.96)
Brain cancer	223	0.91 (0.65-1.17)
Thyroid cancer	7	0.38 (0.00-0.81)
NonHodgkins lymphoma	266	0.82 (0.71-0.92)
Hodgkins disease	68	0.83 (0.61-1.06)
Leukaemia	295	0.90 (0.67-1.14)
All genitourinary disorders	318	0.70 (0.54-0.87)

In the Beral and Darby (2001) meta-analysis, for each study the total deaths observed in the population (or subgroup) and the expected deaths (age, sex and calendar period standardised death rates) were extracted. Where available, data on observed and expected deaths from all cancer, 13 individual cancer types and genito-urinary disease were extracted. The ratio of observed to expected deaths (O/E) was calculated for each cause of death as well as the corresponding 95% confidence interval (95% CI). All studies were combined and summary values calculated. Testing indicated heterogeneity of study results which was considered in summary estimate calculation (Beral and Darby, 2001).

The summary table of mortality (Table 5.1) demonstrates that there is no evidence of a significant increase in deaths from any cause; or from all cancers; or individual types of cancer or genito-urinary disease in large, combined cohorts of uranium process workers who have been followed for many years.

As stated earlier, the studies lack good data on levels of uranium exposure, particularly through inhalation. However, if uranium conveyed a substantial risk for mortality this should have been evident in the studies of uranium process workers. The findings from the meta-analysis of occupational cohorts do not support exposure to uranium or depleted uranium as a substantial risk with respect to death, mortality from cancer or genitourinary disease in humans.

Lung Cancer

When inhaled, insoluble uranium particles may remain within the pulmonary tissues for several years so that inhalation of uranium may result in exposure to radiation of immediately adjacent tissue, especially if such exposure is protracted. For this reason data on mortality from lung cancer has been of interest when examining the possible long-term health effects of uranium exposed workers. Lung cancer is a common malignancy and most mortality data also reflect patterns of disease incidence. Occupational cohort studies can provide adequate numbers for statistical analysis.

In epidemiological studies of uranium miners (see uranium miners section above) and those mining other ores such as tin and iron, an increased risk of lung cancer has been described but attributed to radon progeny.

Uranium mill workers have not shown excess lung cancer despite their increased exposure to uranium and radon progeny. A population of workers was exposed to insoluble uranium dust at levels of 0.5-2.5 mg U/m^3 with some exposed up to an estimated 10 mg U/m^3 for about five years. None of these workers were exposed to other potential irritants. They did not exhibit respiratory disease during the study period (Eisenbud and Quigley, 1956).

Lung cancer mortality has been assessed in a number of cohort studies of uranium industry workers. A number of these studies use similar or overlapping cohorts and as far as possible replication of data have been avoided. The summary data used by the Royal Society (Beral and Darby, 2001) updates that of Fulco et al (2000) and have been preferred in this review.

Each of the 14 cohorts referred to in Table A9.1 of Appendix Nine provides assessment of lung cancer mortality. The summary value (O/E) for these study results combined is 0.94 (95% CI=0.83-1.05) based on a total of 2,846 deaths from lung cancer in these studies.

A number of these studies were large and had prolonged follow-up periods. Of the available studies those which featured large numbers, long follow-up periods, internal controls, some assessment of confounders and multivariate analysis, and measurement of cumulative radiation exposure and subanalysis of dose response data were considered to be the most useful for detailed consideration.

Mortality from lung cancer was not elevated in most cohorts but a small elevation was evident in the large Oak Ridge cohort (Frome et al, 1997) with 1,849 lung cancer deaths and an O/E=1.18 (95% CI=1.13-1.21). In this study dose-response analyses for external penetrating (gamma) radiation were assessed in a sub-cohort of 28,347 men. Variables included in the analyses were age, birth cohort, a measure of socioeconomic status (SES), length of employment, internal radiation exposure potential and facility. The only specific cancer with a positive association with external radiation was lung cancer with an excess relative risk of 1.68 per Sv (95% CI= 0.03-4.94). However data revealed strong SES effects and baseline differences between factories evaluated; as well, smoking histories for workers were not available.

Table 5.2:
Ratio of observed number of deaths from lung cancer in uranium workers compared to that expected in the general population
(from Figure 6. Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	360	0.85 (0.77-0.95)
Dupree-Ellis et al, 2000	98	1.02 (0.83-1.24)
Ritz et al, 2000	46	0.81 (0.59-1.08)
McGeoghegan and Binks, 2000b	67	0.90 (0.69-1.14)
Ritz 1999	112	1.01 (0.83-1.21)
Frome et al, 1997	1849	1.18 (1.13-1.24)
Teta and Ott, 1988	97	1.07 (0.87-1.31)
Cragle et al, 1988	83	0.83 (0.66-1.03)
Beral et al, 1988	11	0.65 (0.32-1.16)
Dupree et al, 1987	21	0.97 (0.60-1.48)
Brown and Bloom, 1987	48	0.88 (0.65-1.17)
Stayner et al, 1985	10	1.13 (0.54-2.08)
Waxweiler et al, 1983	26	0.83 (0.54-1.21)
Hadjimichael et al, 1983	18	0.92 (0.54-1.45)
Summary value	2846	0.94 (0.83-1.05)
Test for heterogeneity: $\chi^2_{13} = 58.70$; $P<0.001$		

Ritz (1999) found no overall increase in lung cancer deaths, with 112 observed and 111 expected deaths from lung cancer (O/E=101, 95% CI=83-121) in a cohort of 4,014 uranium-processing workers. However, a significant dose-response relationship was described between external monitored radiation dose (with a 10-year lag period) and lung cancer. Workers exposed to ionizing radiation experienced an increase in mortality from lung cancer (RR = 2.77; 95% CI = 1.29-5.95). Effects were strongest when exposure had occurred at older ages (>40 years). In addition, an increase in lung-cancer mortality was observed for workers exposed to greater than or equal to 200 mSv of internal (alpha) radiation (RR = 1.92; 95% CI = 0.53-6.96) compared to the control category of <10 mSv exposure. When internal (alpha) and external (gamma) radiation doses were combined lung cancer mortality increased with internal doses greater than 200 mSv and the external dose greater than 50 mSv. Interpretation of the study is somewhat limited by very small numbers at higher dose levels (no cells with more than two in any combined assessment with internal doses greater than 200 mSv) because few workers received high levels of internal or external radiation exposure. Also, there was no control for smoking status. No increase in risk of respiratory cancer was found with increasing internal radiation dose (with or without a lag period).

Ritz et al (2000) examined the effects of chronic exposure to radionuclides, primarily uranium and mixed-fission products, on cancer mortality in a retrospective cohort study of 2,297 workers monitored for internal radiation exposures. 441 deaths were recorded, 112 from lung cancer. Internal lung-dose estimates were calculated based on urinalysis and whole-body and lung counts

reported for individual workers. A significant dose response between internal lung dose and upper aerodigestive tract cancers was described but no dose response was found for respiratory cancer.

Dupree-Ellis et al (2000), undertook dose response analyses in their study of 1,013 deaths from a cohort of 2,514 male uranium industry workers. No excess of lung cancer was described (SMR=1.03, 95% CI=0.84-1.24; based on 103 cases) and no dose response relationship for radiation exposure and lung cancer was found.

Such exposures have not been shown to be associated with respiratory cancer. In the studies of uranium process workers reviewed by ATSDR 1999b and Fulco et al, 2000 no excess of respiratory cancer or non-malignant respiratory disease was established in relation to uranium exposure. No study of uranium industry workers has reported a significant dose-response relationship between internal lung dose and mortality from lung cancer (Beral and Darby, 2001). Fulco et al (2000) concluded that there was limited/suggestive evidence of no association between exposure to uranium and lung cancer at cumulative internal dose levels lower than 200 mSv.

The summary epidemiological data do not support exposure to uranium or depleted uranium as a substantial risk with respect to mortality from lung cancer. Consideration of the summary data and analyses of dose response data do not provide support for an association between exposure to uranium and lung cancer at cumulative internal dose levels lower than 200 mSv. However there is insufficient evidence to determine whether an association does or does not exist between exposure to uranium and lung cancer at higher levels of cumulative exposure.

Bone Cancer (Osteosarcoma)

The bone is an important site for the potential effects of uranium because uranium may be stored in bone, where it replaces calcium in bone matrix. Once there it may remain for several years. Because bone retains uranium, which emits alpha particles, the potential for a radiation related increased risk of bone cancer has been explored in a number of occupational cohorts.

Osteosarcoma is the most common of a number of different types of bone cancer. Its incidence is highest in the younger and adolescent age range. It is difficult to study as it is a rare malignancy in adults and only a few cases would be expected in any study population. Occupational cohort studies using mortality end-points would not be sensitive to establish a small elevation in risk. Eight cohort studies have been considered in the meta-analysis. In most studies only 0-2 deaths from bone cancer were seen and low case numbers lead to wide confidence estimates. Because of the small number of cases no dose response data are available.

Cancer registration data (1971-1991) has been assessed in two studies of uranium process workers (McGeoghegan and Binks, 2000a; McGeoghegan and Binks, 2000b). No cases of incident osteosarcoma were identified in either cohort.

Osteosarcoma has not been observed in humans at a skeletal radiation dose of less than about 10 Gray and has not been observed in populations exposed to any form of uranium, including enriched uranium (Harley et al, 1999). Fulco et al (2000) found there was inadequate/insufficient evidence to determine whether an association does or does not exist between exposure to uranium and bone cancer.

Table 5.3:
Ratio of observed number of deaths from bone cancer in uranium workers compared to that expected in the general population
(from Figure 13. Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	2	0.67 (0.08-2.42)
Dupree-Ellis et al, 2000	1	1.20 (0.07-5.26)
McGeoghegan and Binks, 2000b	0	0.00 (0.00-8.02)
Ritz 1999	0	0.00 (0.00-3.70)
Frome et al, 1997	25	1.19 (0.77-1.76)
Teta and Ott, 1988	0	0.00 (0.00-2.63)
Cragle et al, 1988	1	0.63 (0.02-8.65)
Hadjimichael et al, 1983	1	1.55 (0.02-8.65)
Summary value	30	0.93 (0.53-1.33)
Test for heterogeneity: $\chi^2_7 = 4.58$; P>0.1; NS		

While the available studies are of insufficient quality, and statistical power to permit an absolute conclusion regarding the presence or absence of any association they do exclude a large excess risk. The eight cohort studies have been considered in the meta-analysis of deaths from osteosarcoma and O/E = 0.93 (95% CI = 0.53-1.33) based on a total of 30 deaths from the disease. No association was found in the meta-analysis between uranium exposure and osteosarcoma.

Lymphatic Cancer

The lymphatic system is an important potential site for effects of inhaled insoluble uranium dusts because animal and autopsy studies demonstrate that insoluble uranium may remain for several years in the pulmonary lymph nodes. Lymphatic cancer or lymphoma results from the malignant transformation of cells in the lymphatic nodes and other lymphatic tissues and is broadly classified into non-Hodgkins lymphoma and Hodgkins lymphoma.

Lymphatic cancer is much less common than lung cancer and because of the smaller case numbers is more difficult to study. The small number of cases contributes to wide confidence intervals in individual studies, insensitivity to small alterations in risk and an inability to assess dose response associations. Also occupational cohort studies using mortality endpoints for lymphatic cancers may not accurately reflect disease occurrence, because mortality does not equate to overall incidence.

Lymphatic cancer, assessed as non-Hodgkins lymphoma and Hodgkins lymphoma, has been examined in human studies and no statistically significant association between these forms of lymphatic cancer and uranium exposure has been reported in individual studies of mortality. One assessment of cancer registration (McGeoghegan and Binks, 2000a) noted a trend for increased registration of non-Hodgkins lymphoma and Hodgkins lymphoma with increased cumulative external dose of radiation. These trend estimates were based on small cell numbers (0-2) and the authors considered chance the most likely explanation for these findings.

Twelve cohort studies have been considered in the meta-analysis of deaths from non-Hodgkins lymphoma. For non-Hodgkins lymphoma a statistically significant decrease in the mortality from the disease was described O/E = 0.82 (95% CI = 0.71-0.92) based on a total of 266 deaths from the disease.

Table 5.4:
Ratio of observed number of deaths from non-Hodgkins lymphoma in uranium workers compared to that expected in the general population
(from Figure 7. Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	15	0.63 (0.35-1.04)
Dupree-Ellis et al, 2000	10	0.77 (0.37-1.42)
Ritz et al, 2000	4	0.44 (0.12-1.14)
McGeoghegan and Binks, 2000b	5	1.09 (0.35-2.55)
Ritz 1999	8	1.67 (0.72-3.29)
Frome et al, 1997	187	0.87 (0.75-1.00)
Teta and Ott, 1988	8	0.94 (0.41-1.85)
Cragle et al, 1988	14	0.65 (0.36-1.10)
Beral et al, 1988	1	1.44 (0.04-8.07)
Brown and Bloom, 1987	8	1.57 (0.68-3.11)
Waxweiler et al, 1983	4	0.91 (0.25-2.33)
Hadjimichael et al, 1983	2	0.48 (0.05-1.73)
Summary value	266	0.82 (0.71-0.92)
Test for heterogeneity: $\chi^2_{11} = 8.99$; P>0.1; NS		

Eight cohort studies have been considered in the meta-analysis of deaths from Hodgkins lymphoma. For Hodgkins lymphoma O/E = 0.83 (95% CI = 0.61-1.06), based on a total of 68 deaths from the disease, see Table 5.5. Several studies have reported non-statistically significant elevations in mortality from Hodgkins lymphoma. The authors of these studies suggest a number of explanations for their findings including variations consistent with random variation, exposure to thorium and vanadium as potential confounders. They note internal inconsistency in that deaths due to Hodgkins lymphoma occurred in workers with the shortest periods of employment and with an inverse dose response (Fulco et al, 2000).

The available studies are of insufficient quality, consistency, or statistical power to permit a conclusion regarding the presence or absence of any association however no association between uranium exposure and Hodgkins lymphoma was found in the meta-analysis.

Table 5.5:
Ratio of observed number of deaths from Hodgkins lymphoma in uranium workers compared to that expected in the general population
(from Figure 14. Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	9	1.24 (0.57-2.36)
Dupree-Ellis et al, 2000	2	0.92 (0.15-2.83)
McGeoghegan and Binks, 2000b	2	1.77 (0.21-6.39)
Ritz 1999	6	2.04 (0.74-4.43)
Frome et al, 1997	40	0.77 (0.55-1.05)
Teta and Ott, 1988	2	0.51 (0.06-1.85)
Brown and Bloom, 1987	4	1.54 (0.43-4.01)
Waxweiler et al, 1983	3	2.31 (0.48-6.75)
Summary value	68	0.83 (0.61-1.06)
Test for heterogeneity: $\chi^2_7 = 5.00$; P>0.1; NS		

Other Cancer

Appendix 9 contains the summary tables from the meta-analysis (Beral and Darby, 2001). No elevations in mortality from all cancer or any specific cancer were found. In particular, given the origins for this investigation, mortality from leukaemia was not found to be elevated in these workers O/E = 0.90 (95% CI = 0.67-1.14) based on a total of 295 deaths from the disease in 13 cohorts. In the two studies that have examined both mortality data and cancer registration data from radiation exposed uranium process workers no increase in incidence of or mortality from leukemia was found (McGeoghegan and Binks, 2000a; McGeoghegan and Binks, 2000b).

Alpha-particle induced cancers have been studied in US radium dial painters (Rowland et al, 1978; Fry, 1998), miners exposed to radon gas (ATSDR, 1999b) and patients who received Thorotrast (Andersson et al, 1993). The neoplasms identified in these cohorts have primarily been osteosarcomas and head sinus cancer in the radium dial painters; lung cancer in the miners and liver cancer and leukemia in those administered Thorotrast. Data on lung cancer and osteosarcoma have been considered previously. In the two studies that have examined both mortality data and cancer registration data from radiation exposed uranium process workers no increase in incidence of or mortality from nasal sinus cancer or liver cancer cancers was found ((McGeoghegan and Binks, 2000a; McGeoghegan and Binks, 2000b).

Genitourinary Disease

Uranium has been identified as a nephrotoxic metal, although less so than cadmium, lead, and mercury (ATSDR, 1999b). In the nephron, uranium exerts its toxic effect mostly in the proximal tubules but also in the glomerulus and other areas of the tubules (Voegtlin and Hodge, 1949).

Uranium mill workers occupationally exposed to elevated levels of "yellowcake" have been studied. Yellowcake can be either ammonium or magnesium diuranate. The study reported findings that indicate reduced proximal renal tubular reabsorption of amino acids and low molecular weight

proteins. Concentration data were not obtained. This finding, including mild proteinuria, aminoaciduria, and a dose-related increase in clearance of beta-2 microglobulin relative to that of creatinine, correlated with the duration of uranium exposure (Thun, 1985). In another study, renal injury was not observed in workers exposed to 0.5 to 2.5 mg/m^3 of insoluble uranium dust, including UO_2, for about five years (Eisenbud and Quigley, 1956). The authors noted, "the negative findings relative to renal injury among workers exposed to insoluble compounds are particularly significant in view of the high levels of exposure reported."

It has been noted elsewhere (WHO, 2001b) that the negative findings regarding renal injury among workers exposed for medium to long periods to insoluble uranium compounds (Eisenbud and Quigley 1956; Fulco et al, 2000); and shorter periods of exposure to relatively soluble uranium compounds (Kathren and Moore, 1956) are significant because of the high levels of exposure reported in these studies. The comprehensive review of the human toxicity of uranium undertaken by Fulco et al (2000), covering both epidemiological and experimental studies concluded that "although uranium is a heavy metal that causes transient renal dysfunction, the preponderance of evidence indicates little or no clinically important renal effects of exposure to uranium."

Mortality from genitourinary and renal disease has not usually been found to be elevated in uranium workers. The meta-analysis gives an O/E = 0.70 (95% CI = 0.54-0.87) based on seven cohorts with 318 deaths from genitourinary disease. This statistically significant negative finding may be contributed to by a strong healthy worker effect and by the difficulty of studying this condition in mortality data, however it does not support any material contribution to disease by uranium exposure in workers.

Dupree-Ellis et al (2000), (not included in this arm of the Beral and Darby meta-analysis) found no increase in genitourinary disease. They did identify a non significant increase in chronic nephritis in uranium process workers based on six deaths giving an SMR = 1.88 (95% CI = 0.75-3.81) in their cohort of 2,514 uranium process workers. The authors do highlight the potential inaccuracies in the coding of this data which may lead to uncertainty in interpretation. Their analysis of diseases of the genitourinary system in general was based on 14 deaths giving an SMR of 0.95 (95% CI = 0.53-1.54). Addition of this study to the meta-analysis would not materially alter the summary results given the small number of additional deaths in the cohort and essentially similar results for the category of genitourinary disease.

Table 5.6:
Ratio of observed number of deaths from genitourinary diseases in uranium workers compared to that expected in the general population
(from Figure 16. Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	28	0.57 (0.38-0.83)
Ritz et al, 2000	5	0.78 (0.25-1.81)
McGeoghegan and Binks, 2000b	7	0.98 (0.39-2.02)
Ritz 1999	3	0.21 (0.04-1.29)
Frome et al, 1997	270	0.83 (0.73-0.94)
Brown and Bloom, 1987	3	0.54 (0.11-1.56)
Stayner et al, 1985	2	0.89 (0.11-3.18)
Summary value	318	0.70 (0.54-0.87)
Test for heterogeneity: $\chi^2_6 = 8.04$; P>0.1; NS		

Two studies that show changes in renal function (Lu and Zhao, 1990; Zamora et al, 1998) have been questioned (Harley et al, 1999; Fulco et al, 2000) because use of their doubtful clinical relevance, the potential for reversibility of the renal changes and for confounding factors affecting study outcomes.

The comprehensive review of the human toxicity of uranium undertaken by Fulco et al (2000) covering both epidemiological and experimental studies concluded that " although uranium is a heavy metal that causes transient renal dysfunction, the preponderance of evidence indicates little or no clinically important renal effects of exposure to uranium." Additionally, the meta-analysis has shown a statistically significant decrease in deaths from genitourinary disease O/E = 0.70 (95% CI = 0.54-0.87).

Nonmalignant Respiratory Disease

Nonmalignant respiratory effects from any inhaled dust or aerosol depend on particle size and solubility and particle clearance in the upper and lower airways. Uranium workers are exposed to a range of uranium compounds. Uranium miners are exposed predominantly to insoluble uranium oxide dusts, as well, they are exposed to a number of agents such as silica which are known to cause lung disease. Nuclear industry workers are exposed to both insoluble uranium oxides and more soluble uranium compounds such as uranyl fluoride. They too have other exposures which may affect lung tissue.

A review of eight available studies of US uranium process workers (Poledak and Frome, 1981; Hadjimichael et al, 1983; Stayner et al, 1985; Brown and Bloom, 1987; Dupree et al, 1987; Checkoway et al, 1988; Frome et al, 1990 and Ritz, 1999) by Fulco et al (2000) found lack of control for potential confounders and inconsistent results. The review concluded that there was inadequate/insufficient evidence to determine whether an association does or does not exist between exposure to uranium and nonmalignant respiratory disease.

Three studies published since Fulco et al (2000) analysis were also considered by this Expert Committee. Dupree-Ellis et al (2000) found an SMR of 0.80 (95% CI = 0.62-1.01) for diseases of the respiratory system, in their retrospective cohort of 2,514 uranium process workers. Pneumonia, asthma and emphysema were considered in separate analyses and all were below expected (non-statistically significant decreases). No control was made for smoking status and no assessment by radiation dose was published for these conditions. Two large UK retrospective cohort studies (McGeoghegan and Binks, 2000a; McGeoghegan and Binks, 2000b); have recently reported no increase in death from nonmalignant respiratory system disease in workers from the Springfields or Capenhurst sites of British Nuclear Fuels. Analyses were undertaken for non-radiation exposed and radiation exposed workers and comparison with general and local area population mortality rates were made. For the Springfields workers SMR (non radiation exposed) = 100, and SMR (exposed to radiation) = 79 ($p<0.001$). For the Capenhurst workers SMR (non radiation exposed) = 90, and SMR (exposed to radiation) = 70 ($p<0.001$). Analysis by cumulative external radiation dose group and with varied lag periods did not reveal any positive associations in either cohort. No control for smoking was made but authors suggested smoking was not likely to be a confounder in either cohort.

Studies of mortality from nonmalignant respiratory disease have been undertaken in occupational cohorts, but are difficult to interpret as none control for the effect of smoking habit, which is an important cause of this group of conditions. Also, few control for potential occupational confounders such as asbestos. Additionally mortality data may not be sensitive to assess this endpoint.

Nonmalignant Central Nervous System Disease

Animal studies have found uranium at high doses can cross the blood brain barrier. This finding, and the interest in the health of those with retained depleted uranium pellets and persistently elevated uranium excretion has set some focus onto this group of conditions.

Fulco et al (2000) reviewed five studies which had considered mortality from nonmalignant central nervous system disease in cohorts of uranium exposed workers. Results from these cohorts were inconsistent. One study, Hadjimichael et al (1983) reported a significant excess SMR = 346 (95% CI=126-753) based on only six cases, two studies (Poledak and Frome, 1981; and Frome et al, 1990) reported a non statistically significant decrease; and two studies (Brown and Bloom, 1987; and Stayner et al, 1985) reported a statistically significant decrease in mortality with SMR = 40 (95% CI = 21-68) and SMR = 34 (90% CI = 9-89) respectively.

Three studies published since Fulco et al (2000) analysis were also considered. Dupree-Ellis et al (2000) found an SMR of 0.82 (95% CI = 0.43-1.41) for diseases of the nervous system and sense organs, in their cohort of 2,514 uranium process workers. Two UK studies (McGeoghegan and Binks, 2000a; McGeoghegan and Binks, 2000b); have recently reported a significant decrease in mortality from this group of conditions in workers from the Springfields and Capenhurst sites of British Nuclear Fuels; SMR = 62 ($p<0.001$) at Springfields (based on 49 deaths) and 71 ($p<0.05$) at Capenhurst (based on 42 deaths) when compared to the general population.

Case studies of individuals who were accidentally heavily exposed to uranium have not described clinically detectable, persistent neurological dysfunction (Moore and Kathren, 1985). While cohort studies of mortality may not be sensitive to these conditions, overall there is little epidemiological evidence to support an association between occupational uranium exposure and central nervous system disease.

Immunological Dysfunction

Subtle immune dysfunction has been raised as a possible consequence of depleted uranium exposure (Durakovic, 1999). Epidemiological studies in humans have not specifically addressed the potential for adverse immunological effects from uranium. ATSDR, (1999b) cites a number of cohort studies which were not designed as immunotoxicity studies (Archer et al, 1973b, Brown and Bloom, 1987, Checkoway et al, 1988 Polednak and Frome, 1981) but which histologically evaluated immune system structures for the effects of uranium exposure. These studies found no significant immunological change in human subjects.

One study of German uranium miners reported an increased risk of scleroderma (Baur, 1996); and two studies of quartz dust-exposed uranium miners suggest at least the potential for an increase in systemic autoimmune disease (Conrad et al, 1996; 1998). These miners all had exposure to silica and there is a body of literature suggesting an association between silica and these conditions. Additionally these workers would have been exposed to radon gas and progeny. Studies in mice have shown immunological alterations including in the numbers and percentages of lymphocytes and macrophages, which the authors suggest, may contribute to hypersensitivity and cancer formation (Nagarkatti et al, 1996). No association may be drawn between uranium exposure and alteration in immune function from the limited available information.

Depleted Uranium Exposure

Retained Depleted Uranium Fragments

This form of exposure has drawn interest following "friendly fire" incidents during the Gulf War, when depleted uranium munitions struck about 30 US Bradley Fighting Vehicles and Abrams Tanks (OSAGWI, 1998; Livengood, 1998). More than ten percent of the men in these tank crews were killed at impact however a significant proportion survived, many with considerable physical injuries and some with retained depleted uranium shrapnel (Fetter and von Hippel, 1999). While the potential risk associated with depleted uranium fragments is very small in comparison to the incidents which produced them, there is a desire to assess and quantify this risk. Animal studies of implanted depleted uranium pellets have been discussed earlier in this section.

Urinary analyses were undertaken in 1993-94, of 33 veterans exposed to depleted uranium many via retained munitions fragments. Increased urinary uranium was detected in those with retained fragments with mean urinary uranium excretion of 4.47 vs 0.03 μg/g creatinine in those without confirmed retention of fragments.

In 1997, 29 of the initially assessed group and 38 non-exposed Gulf Veterans were evaluated extensively (McDiarmid et al, 2000). A large proportion (76%) of those with retained depleted uranium fragments continue to have active medical problems due to the injuries received during the Gulf conflict compared with 14% of the veterans in the control group (McDiarmid et al, 2000). How these injuries and resultant medical conditions impact on the study of depleted uranium itself remains to be determined. Currently there are no clinical conditions (other than injury) which have been found to be more prevalent in the exposed cohort. In 1999, 21 original participants and 29 newly identified participants who had been exposed to "friendly fire" incidents were assessed in the exposed cohort (McDiarmid presentation January 2001 GWI conference, Washington DC).

The published analysis of the 1997 assessment compared urinary uranium determinations, clinical laboratory values, and psychiatric and neurocognitive test results for 29 Gulf War veterans with retained fragments of depleted uranium shrapnel, and 38 non-exposed Gulf War veterans. History and follow-up medical examination were performed (McDiarmid et al, 2000).

Urinary uranium concentrations in Gulf War veterans with retained depleted uranium shrapnel have been shown to be greater than those from Gulf War veterans without such exposure (controls) when measured at two, four and seven years after the Gulf War (Hooper et al, 1999; McDiarmid et al, 2000). The level of increase in urinary uranium has been as much as 150 times that of the controls. Veterans with retained depleted uranium shrapnel had urinary uranium concentrations of 0.01-30.74 μg/g of creatinine while the control veterans urinary uranium ranged from 0.01-0.047 μg/g of creatinine. Hooper et al (1999) report excretion levels between 10-20 μg/litre. Despite higher levels of urinary uranium in those with retained depleted uranium shrapnel, renal injury has not been described and renal function as measured by serum creatinine, BMG, retinol-binding and urine proteins is the same for the exposed and control veterans. These findings suggest that this chronic exposure has not caused renal damage (McDiarmid et al, 2000).

McDiarmid et al (2000) reported that results of standard neurocognitive function tests were similar for those with depleted uranium shrapnel and controls. However, results from a battery of computer-based neurocognitive tests performed in 1997 showed an association between the level of urinary uranium and “problematic performance on automated tests assessing performance efficiency and accuracy”. This association was not evident at the latest reassessment in 2000 (McDiarmid presentation, January 2001, GWI Conference Washington DC).

Reproductive function has been assessed by a number of means. Semen has been shown to contain uranium, but the semen volume, and sperm concentration, morphology, and motility are the same in both those with high and low uranium excretion. Additionally measures of the related hormones FSH, LH, testosterone and prolactin were similar for those with high and low urinary uranium excretion. As of January 2001, 38 children had been born to the exposed cohort and no birth defects have been reported in these offspring (McDiarmid presentation, January 2001, GWI Conference, Washington DC).

One post hoc, subgroup analysis from the 1997 assessment revealed higher urinary uranium excretion in men with prolactin levels greater than the median, compared to those with prolactin below the median (McDiarmid et al, 2000). This finding was assessed by the National Academy of Sciences and they considered this an unconventional analysis of questionable validity in a small sample; as well, no measures of cortisol (a mediator of prolactin plasma levels) or account of moment to moment daily variation in prolactin were available and it was suggested that these findings are hypothesis generating (Fulco et al, 2000). The association with prolactin levels had disappeared by the 2000 reassessment (McDiarmid presentation, January 2001, GWI Conference, Washington DC).

Haematologic measures were reported to be similar in those with retained depleted uranium fragments and the control group of Gulf War veterans. As well, no differences existed in sub analyses between those with shrapnel who had high or low urinary uranium excretion. Retained depleted uranium was found to have no association with on haematocrit, haemoglobin, platelets, lymphocytes, neutrophils, basophils, eosinophils, or monocytes. Additionally these groups demonstrated the same background frequency of chromosomal aberrations and sister chromatid exchanges in peripheral blood lymphocytes (McDiarmid et al, 2000).

No dermal, ocular or musculoskeletal effects of depleted uranium have been reported in human literature.

Other Studies of Soldiers Potentially Exposed to Depleted Uranium

The total number of US military personnel present at one time or another during the interval of Operation Desert Shield/Storm (the Gulf War) was about 700,000 (IOM, 1997). Despite the arduous conditions, morbidity rates among US troops were lower than in previous wars (Hyams et al, 1995). Mortality was also much lower than expected. Altogether 372 deployed US troops died in 1990-1: 40% from combat, 52% from accidents (primarily related to training and motor vehicles), and 8% from illness (Writer et al, 1996). About 53,000 UK troops saw service in the Gulf and Australia's involvement in the Gulf war was numerically small, with some 1,800, predominantly Naval personnel, seeing service across the period of this conflict. No Australian deaths were recorded during this service.

Studies of the health of Australian, US and UK veterans of the Gulf War are in progress and depleted uranium exposure is one of many exposures being considered. US and UK studies demonstrate a small excess of death among veterans of the Gulf War from external causes in the years after the war (Kang and Bullman, 1996; Macfarlane et al, 2000). This is a finding which has been described in several cohorts of returned soldiers across the Twentieth century (Freed and Stringer, 1965; CDC Vietnam Experience Study, 1987). No study of veterans of the Gulf or Balkans War has so far described any excess of deaths which could be attributed to the chemical or radiobiological effects of depleted uranium.

McDiarmid et al (2001b) have examined exposure scenarios and 24 hour urinary uranium concentrations in Gulf War veterans, outside those assessed in the “heavy exposure” studies

outlined in the previous section. In 1998-1999, 169 US Gulf War veterans submitted 24-hour urine samples for determination of urinary uranium concentration and questionnaires describing their potential exposures to depleted uranium while in the Gulf War theatre. Depleted uranium exposure was assessed by the questionnaire for 19 exposure scenarios. Results of urinary uranium analysis were stratified into high and low uranium groups with 0.05 μg uranium/gram creatinine being the cut point and approximate upper limit of the normal population distribution. Twelve individuals (7.1%) exhibited urinary uranium values above this level, while the remaining 157 had urine uranium values in the low range. A repeat test of urine for six of these 12 produced uranium results in the low range for three of these individuals. The presence of retained shrapnel was the only scenario predictive of a high urinary uranium value. The authors suggest that elevated urinary uranium is unlikely, as are any uranium-related health effects, in the absence of retained depleted uranium metal fragments.

Overview

Available sound-medical scientific evidence related to the health effects of depleted uranium includes animal experiments, studies of miners and workers in the uranium industry and studies of Gulf War veterans.

Uranium is chemically classified as a heavy metal and is weakly radioactive. Any health effects should therefore be considered potentially arising through chemical toxicity or radiation emissions.

Uranium has relatively low order toxicity compared to other heavy metals. Animal experiments have identified the kidney as the organ that is most sensitive to uranium's chemical toxicity. Acute respiratory, neurological and haematological effects have been reported after high doses of uranium. Cardiovascular, gastrointestinal and hepatic effects are not prominent in animal experiments by any route of exposure.

An increase in lung cancer has been reported in beagles in one long term, high dose inhalation study, but the majority of animal studies considering the potential for carcinogenic effects have not found an association with uranium exposure.

In studies of rats implanted with depleted uranium pellets, elevated urine uranium levels have been found, as well as uranium deposition in the bone, kidney, brain, lymph nodes, testes and spleen. There was nevertheless no physiological or histological evidence of organ damage.

Most animal studies have employed very high dose regimens to explore toxicity, and so raise the question of the applicability to human exposures at levels which are orders of magnitude below the experimental exposures.

There have been a few human reports of high level, acute exposure to uranium. Renal impairment has been reported in some of these cases and that was generally reversible.

Long term exposure to uranium, largely through inhaled and ingested dust, has been examined in occupational cohort studies involving underground uranium miners and uranium industry workers. Uranium dusts may be deposited in the lungs and associated lymphatic tissues and after systemic absorption a proportion is stored in the bone, kidneys and liver.

Studies on underground miners are of very limited value in assessing the lung carcinogenicity and other health effects of depleted uranium as these miners were also exposed to high concentrations of radon gas, a recognised lung carcinogen.

Studies of uranium industry workers, who are involved in the processing of uranium and may inhale and ingest uranium dust and aerosols are not confounded by exposure to radon gas. There may nevertheless have been exposure to other radionuclides in these workers, either internally or externally.

A combined analysis of 14 long term follow up studies of uranium industry workers found no evidence of any increase in deaths from any cause, nor from all cancers, nor individual types of cancer nor genitourinary disease. Specifically, mortality from lung cancer, osteosarcoma, lymphatic cancer and leukaemia were not found to be increased (Beral and Darby, 2001).

While no overall increase in lung cancer was observed, several studies described a dose response for external radiation dose and mortality from lung cancer (Frome et al, 1997; Ritz, 1999). This form of exposure would have arisen from exposure to radionuclides other than uranium, because alpha particles do not penetrate tissue. Fulco et al (2000), after a detailed assessment of the available literature, concluded that there was "limited/suggestive evidence of no association" between exposure to uranium and lung cancer for cumulative internal doses below 200 mSv. They also found that there was inadequate/insufficient evidence to determine if any association existed at higher doses.

There has been no established increase in mortality or morbidity from respiratory disease, neurological disease or immunological disease in workers exposed to uranium in uranium processing.

The available epidemiological data are however limited by uncertainty in assessment of exposure, low statistical power for studying rare events and a lack of data on exposures to other potentially toxic workplace and environmental exposures.

Studies have also been carried out of Gulf War veterans. In those who have retained fragments of depleted uranium following combat-related injury, it has been possible to detect elevated urinary uranium concentrations but no kidney toxicity or other adverse health effects related to depleted uranium after a decade of follow-up.

Section Six

Risk

The concept of risk implies the potential for increased liability to harm, or to some adverse outcome. It is used technically in terms of hazards which may impact through exposure to cause increased likelihood of injury or illness, or death. The judgement about the likelihood or otherwise of such outcomes is made on the basis of population data comparing exposed and non-exposed groups, ie through epidemiological studies.

An individual's likelihood of being affected is derived from his/her characteristics and exposure in relation to population outcomes from similar exposures. The individual perceives and may experience risk in terms of a range of factors. These include:

- The understanding (or lack of understanding) of the scientific basis of "risk" and how it may or may not apply to him or her.
- Past experience of threats, similar or dissimilar and their consequences.
- Levels of arousal and/or anxiety about potential threats, general and/or specific.
- Trust for sources of information about the threat.
- Degree of uncertainty or perceived uncertainty about effects, particularly if these are not immediate.
- Personal coping or cognitive style including personality characteristics which may mean the individual sees their world as more potentially threatening, out of his or her control, or more negatively.
- Myths or beliefs held by the individual and his or her social group about the threat/hazard and risks associated with it, including potential health or other outcomes.

Thus while there may be scientific information about population risks each individual's perception and experience of it, in addition to objective factors such as exposure, will need to be taken into account. Furthermore, it is possible that when perception of risk and threat is heightened, or uncertain, these perceptions may in and of themselves impact on the individual's well-being. Ultimately the harm to health may not be from the actual risk but from the anxiety about the risk.

Risk Assessment

Risk assessment is the characterisation of potential adverse effects of human exposures to hazardous agents or activities. Hazard has no time base, exposure and risk have a time base. Risk results from the combination of hazard and exposure to the hazard with resultant dose to sensitive or target tissues. When there is no current or potential exposure there can be no risk. Where exposure has occurred it is the amount or dose which determines the size of the risk.

People make informal judgements about risks every day. Some risks are familiar, even comfortable; others are unfamiliar and can be sources of considerable fear. Different people have different perceptions of the same risks.

Assessment of risk can be controversial, reflecting the important role that both empirical scientific observation and judgement play in drawing conclusions about the likelihood of effects on human health and the environment. Often, the controversy arises from what we don't know and from what risk assessments can't tell us, because our knowledge of human vulnerability and of environmental impacts is incomplete. This is particularly relevant when considering the risk from relatively low levels of chemical or radiation exposure.

The potential pathways of exposure for environmental chemicals are limited and are the same for depleted uranium as for any other chemical ie ingestion, inhalation and skin contact (including wound contamination). These are pathways for external or internal exposure and do not imply dose. An exposure may occur but if the agent is not absorbed no dose may ensue. It is the dose to the target organs and not merely exposure, which contributes to the attending risk for adverse outcomes.

Risk Assessment for Exposure to Depleted Uranium

External contact route

Manufacture and Storage of Depleted Uranium

Based on extensive study of the health of uranium process workers (see Section Five) the risk from depleted uranium manufacture and storage is negligible.

External Exposure to Depleted Uranium Containing Munitions

There is no evidence that skin contact to natural or depleted uranium by humans can lead to chemical toxicity. A number of reviews have considered the external radiation doses from depleted uranium. Some have used theoretical calculations (Fetter and von Hippel, 1999) others have attempted direct measurement (OSAGWI, 2000). This exposure could occur in unprotected handling of the depleted uranium munitions such as loading and working with depleted uranium munitions and in handling fragments of these munitions. Usually gloves would be worn. Skin exposure could also occur in rescue or clean up operations after the use of depleted uranium munitions or fires as in the Camp Doha incident during the Gulf War (OSAGWI, 2000).

As outlined in Section Two, depleted uranium is predominantly an alpha emitter however small amounts of beta emissions and photons (x-rays and gamma rays) also are produced. The inert outermost layers of the skin stop alpha particles. Beta particles can travel about a centimetre into the body. Photons (x-rays and gamma rays) are more penetrating and can pass straight through the body. Depleted uranium external to the body is a potential source of small amounts of gamma and beta radiation. See Figure 2.1 for a diagrammatic representation of relative penetration of these emissions. Intact depleted uranium anti-tank shells are partially enclosed in metal casing. The 30-mm Gatling gun munitions have a 0.8mm aluminium jacket. The metal casing would absorb alpha and beta emissions from the enclosed depleted uranium.

Fetter and von Hippel (1999) provide the only comprehensive theoretical assessment of the hazards from depleted uranium munitions which has been published in a peer-reviewed journal (Royal Society, 2001). They estimate that the theoretical maximum whole-body gamma-ray dose-rate (the whole-body dose within an infinite slab of radioactive material) from external exposure to depleted uranium is 0.025 mSv per hour. They suggest dose rates in this range may be experienced by a person completely surrounded by depleted uranium munitions in a uranium store or in a vehicle reinforced by depleted uranium containing amour and armed with its complement of depleted uranium weapons. Measurements by the US Army have demonstrated that the highest exposures

from gamma radiation are likely to arise in a depleted uranium-armoured vehicle carrying depleted uranium ammunition. According to the US Army, the whole-body dose rate in a tank fully loaded with depleted uranium munitions is less than 0.002 mSv per hour (Harley et al 1999 Tables G1 and G2). Thus, driving a fully loaded tank for 2000 hours would result in a dose roughly equal to the average annual dose from natural background radiation (3.0 mSv per year (ATSDR, 1999a)) (Fetter and von Hippel, 1999; Harley et al, 1999). This is equivalent to driving a tank every working day of the year for the whole of an eight hour working day and results in a radiation dose that is 15% of the allowed occupational exposure for ionizing radiation.

Fetter and von Hippel (1999) have calculated the dose rate to a person standing on flat ground uniformly contaminated with 1 ton of depleted uranium per square kilometre at about 0.01 mSv per year, and for comparison, the dose rate from natural uranium in soil is ten times greater at about 0.1 mSv per year. Even in the area immediately surrounding a vehicle destroyed by depleted uranium munitions, the depleted uranium generated dose rate from external radiation is unlikely to exceed 0.3 mSv per year—ten times less than the natural background dose rate for the US (ATSDR, 1999b, Fetter and von Hippel, 1999). US regulatory limits for public exposure to other than background sources of ionizing radiation is 1 mSv per year (ATSDR, 1999a).

WHO (2001a) calculated the amount and fate of depleted uranium deposited at an 'average' attack site in Kosovo. Calculations are at Appendix Eleven and demonstrate that even if all the depleted uranium munitions expended during an attack remained within one kilometre of the target site the increase of uranium in the soil would be five percent. The additional contribution of depleted uranium from military use to background radiation dose in Kosovo is within the natural variations found for background levels.

The WHO (2001a) report suggests that “picking up a penetrator and keeping it in a pocket is the only realistic way of a long period of exposure to external (beta) radiation from depleted uranium. Snihs & Åkerblom (2000) considered that "by keeping it in the same position for several weeks, it might be possible that the dose administered to that immediate area of skin would exceed the skin dose limit for the general population, though not that of radiation workers. The effect of such exposure would be localised and the delivered dose would not be sufficient to cause any deterministic effect”.

The available evidence indicates that external contact with depleted uranium in intact penetrators or as uranium oxide dusts does not convey an appreciable excess risk for adverse health effects.

Internal Exposure to Depleted Uranium

Inhalation

As outlined earlier in Section Four, when a hard target such as an armoured tank or a rocky outcrop is hit by depleted uranium munitions, 10 to 35 % with a maximum of 70% of the original depleted uranium metal forms into an aerosol of the metal and its combustion products, which are predominantly uranium oxides. Aerosols and dusts containing depleted uranium and its oxides may be inhaled into the lungs. It has been estimated that approximately 60 to 69 percent of the aerosolized fraction of this depleted uranium is respirable based on particle size (OSAGWI, 2000; Harley et al, 1999).

The inhalation and fate of these particles depends on their size. Some particles will be exhaled, some will deposit in the upper airways (the nose, mouth and bronchial tree), and some will deposit deep in the lungs. 95% of the larger (greater than 10 μm aerodynamic equivalent diameter) particles

are deposited in the upper respiratory tract (bronchioles, bronchi, trachea). The majority of these particles are cleared by the normal bronchial mucociliary clearance mechanism and swallowed or blown out of the nose. "As such, it is only the smaller respirable particles which represent a potential health hazard from inhaled (natural) uranium. As particle size decreases below 10 μm, deposition decreases in the extrathoracic and bronchial regions but increases in the bronchioles and alveoli (pulmonary regions) such that, at particle sizes below 0.5 μm, the alveoli represent the major site of deposition. Retention of particles in various lung compartments depends on the efficiency of the mucociliary clearance mechanism which decreases in the deeper portions of the lung, or by macrophage action and solubilisation." (ICRP, 1994).

The acute hazard from inhaled uranium aerosols is related to the extent and rate of transfer of inhaled uranium to the blood and the presumed amounts reaching the primary targets in the kidney. “Two factors will influence the degree of hazard: the site of deposition in the respiratory tract, dependent on the aerodynamic equivalent diameter, and the fate in the lung, dependent on the physical and chemical characteristics of the particles, such as the solubility, exposed surface areas, and intercrystalline forces” (OSAGWI, 2000). A radiation dose predominantly from alpha decay is delivered to the airways and lung while the uranium remains in the respiratory system. Absorption of depleted uranium in the body following inhalation is very limited. Mean absorption following inhalation exposure is about 0.8 to 0.9%, with less soluble compounds as uranium oxides remaining in the lungs (UNEP/UNCHS Balkans Task Force, 1999).

UNEP (2001) has estimated that the inhalation and ingestion of depleted uranium contaminated dust, even under extreme conditions, and shortly after the impact of projectiles, as determined by the amount of dust that can be inhaled, would be less than about 10 mSv. This represents about half the annual dose limit for radiation workers. For people in open areas near destroyed tanks or near burning depleted uranium, the aerosol dose is considerably less. For people who entered the tanks or the vicinity of the former fire sites after the aerosol had settled, the internal contamination is also much smaller. At the most, a smaller portion of the aerosol is resuspended and could be inhaled. Overtime this exposure is further reduced by climatic influences, such as rain or snow.

A number of descriptive assessments have been made for depleted uranium exposure and resultant dose. These are generally theoretical assessments as measurement for some scenarios is lacking. Appendix Twelve contains assessments used by UNEP (1999) and European Commission (2001).

Ingestion

As outlined in Section Four ingestion may occur via hand mouth transfer of depleted uranium dusts, or consumption of soil, water or foodstuffs containing depleted uranium. Typical world-wide dietary intake is estimated between 0.9-4.5 μg/day with an average of 1.5 μg/day (ATSDR, 1999b). Only about 2% of uranium in easily soluble form is taken up into the bloodstream from the digestive tract, the rest is eliminated rapidly through the intestines. Uranium in the form of uranium oxide, which is poorly soluble in the body, is taken up by the digestive tract in negligible amounts, only about 0.2%.

The absorbed fraction is taken into the blood and is rapidly cleared (in a few minutes), with approximately 90% leaving the body in urine within the first week, and the remainder being distributed to tissues. The evidence available indicates that this route would provide a negligible systemic dose of insoluble uranium oxides for personnel in combat situations.

Agricultural route

Food may become a vehicle for uptake of depleted uranium. As outlined in Section Four plants may take up uranium from the soil and soil may adhere on to the root surface. Calculations used by

the European Union (2001) indicate that consumption of 100kg/year of vegetables grown in soil with 70 mg of depleted uranium/kg of soil would yield a radiation dose of 0.0026 mSv/year. Consumption of unwashed root vegetables may increase exposure due to ingestion of soil. Washing reduces soil intake by 99%. UNEP (1999) estimate that the effective dose by ingestion (all forms of food) would be at most 0.007 mSv/year. This is a small proportion of the average individual dose of ionizing radiation from ingestion of other radionuclides (0.3 mSv/ year).

It is recognised that aquatic macrophytes accumulate uranium and there are species of plants in the Alligator River region of Australia, which show preferential concentration of uranium. The 2001 WHO mission to Kosovo explored the theory that uranium dust might become incorporated in vegetables and crops. "The mission was advised by the Food and Agricultural Organisation of the United Nations (FAO) that in the published literature there are no known (cultivated) plants that preferentially accumulate uranium and the normal amounts of uranium taken up in plants would not be expected to be dangerous to humans, birds or other animals" (WHO, 2001a).

Drinking water route

As outlined in Section Four drinking water can contain naturally occurring uranium over a wide range of concentrations. After environmental incorporation of depleted uranium a number of physical, chemical and geological factors affect the potential for exposure to additional uranium.

The European Union (2001) estimated that a radiation dose of about 0.0001 mSv/yr (worst case 0.024 mSv/yr) could result from consumption of drinking water. This calculation was based on the assumptions of 10 kg of depleted uranium spread over 1000 m^3 of soil surface eventually dissolving; a proportion of this then enters the groundwater per year, and this water is then used for drinking.

The WHO mission to Kosovo considered exposure of the population via drinking water contaminated by migration of soluble depleted uranium compounds in ground or surface water. "In particular, possible contamination of wells or spring protection tanks close to an attack site from pieces of depleted uranium might be an isolated occurrence and its relevance should be considered further." (WHO, 2001a). The very small amounts of depleted uranium in comparison with the natural uranium present in soils and rocks is not likely to significantly increase the amount of uranium already present in drinking water.

Exposure Standards for Uranium

Risk assessment for uranium has been undertaken in relation to both heavy metal toxicity and radioactivity. Exposure standards have been defined for the general population and for workers in the radiation industry. The World Health Organisation has been involved in the setting of guidelines on uranium exposure which apply to depleted uranium (Repacholi, 2001). Currently these are:

- Permissible exposure limit time weighted average of 0.05 mg/m^3 for soluble uranium compounds and 0.25 mg/m^3 for insoluble uranium compounds (ATSDR, 1999b)
- The tolerable intake for soluble uranium compounds is 0.5 µg/kg/day and is 5.0 µg/kg of body weight/day for insoluble compounds (WHO, 2001b)
- Limits of ionizing radiation exposure of 1 mSv/year for the general public and 20 mSv/year averaged over 5 years for radiation workers (ATSDR, 1999b)

Combat Related Exposure Assessments

Three levels of "battle related" exposure have been considered for internal exposure to depleted uranium and are provided in Section Four. These levels do not incorporate external exposure to intact munitions or depleted uranium armour as no depleted uranium is taken into the body from these exposures.

The two available models (OSAGWI, 2000; Royal Society, 2001) differ slightly in description however the framework is as follows:

> Level One exposure refers to the most highly exposed personnel ie personnel in a vehicle at the time it is struck by depleted uranium munitions, or to those who enter the vehicle immediately after such an occurrence. The exposure is through inhalation and ingestion of depleted uranium, wound contamination and retention of depleted uranium fragments.
>
> Level Two exposures refer to less highly exposed personnel ie personnel working for hours on or in contaminated vehicles to carry out cleaning and repairs. The exposure is through inhalation of resuspended depleted uranium and by hand to mouth transfer and ingestion.
>
> Level Three exposures include all lesser internal exposures such as being downwind of impacts or fires or brief entries into contaminated vehicles.

Mathematical modelling has been used to estimate doses for these levels. Such modelling requires many assumptions. No direct measurements of exposure were undertaken at the time of these events. Differences in modelling contributes to the differing assessments by US an UK scientists (OSAGWI, 2000; Royal Society, 2001). The US Army has been tasked to more fully characterise the actual exposure to depleted uranium in combat vehicles struck by depleted uranium munitions and this testing has an expected completion date in 2002 (Royal Society Annexe C page 3).

Table 6.2 summarises the US military upper limit estimates for all levels (OSAGWI, 2000). The estimates are based on the upper extremes of exposure for a given scenario, for example, Level II estimates assume each individual was exposed to all 31 US depleted uranium contaminated vehicles, and Level III estimates assume exposures of 100 vehicle-hours, such as from one-hour exposures in 100 vehicles. As such, they serve as upper limit assessments of modelled exposure and secondarily modelled dose.

Detailed analyses of the potential for exposure and uptake of depleted uranium into the body have been undertaken and are relevant for the exposure scenarios before the Expert Committee. In performing its analysis, OSAGWI (2000) considered the following sources of depleted uranium exposure; and these are similar in most respects to those considered in the Royal Society report (2001):

- airborne depleted uranium from 120mm munitions penetrating Abrams tanks and Bradley Fighting Vehicles;
- residues inside those depleted uranium-contaminated vehicles, including both depleted uranium-penetrated and burned-out vehicles with onboard munitions, resuspended (stirred up) by personnel entering and working;
- depleted uranium on the ground resuspended by vehicle and personnel traffic;
- depleted uranium in smoke from burning vehicles and munitions.; and
- depleted uranium residues inside and outside contaminated vehicles that personnel could ingest or transfer to wounds.

Table 6.1:
Summary of estimated intakes (mg) and effective doses (mSv) for the different battlefield scenarios.
(Royal Society, 2001)

Scenario	Central Estimate		Worst case	
	Intake mg	Dose MSv	Intake mg	Dose MSv
Level I inhalation of impact aerosol	250	22	5000	1100
Level II inhalation of resuspension aerosol within contaminated vehicle	10	0.5	2000	440
Level II ingestion within contaminated vehicle	5	0.0005	500	0.3
Level III inhalation of resuspension aerosol within contaminated vehicle	1	0.05	200	44
Level III ingestion within contaminated vehicle	0.5	0.00005	50	0.03
Level III inhalation of plume from impacts	0.07	0.004	5	2.8
Level III inhalation of plume from fires	0.05	0.004	1	1.2
Level III inhalation of resuspension from ground	0.8	0.03	80	18

As will be clear from a comparison of Tables 6.1 (Royal Society, 2001) and 6.2 (OSAGWI, 2000) estimates for intake and effective radiation dose differ considerably in several key aspects. Most obviously the Royal Society central estimates approximate the upper limit intakes estimated by the OSAGWI. The Royal Society estimates for "worst case" scenarios are up to two hundred times that for their central estimates. Even in the “worst case” scenarios Level III exposure is not associated with theoretical exposures which could affect health.

The potential for exposure is obviously greatest inside a struck vehicle at the time of impact by a depleted uranium penetrator. These exposures result from inhalation, ingestion, wound contamination, and retained uranium fragments. The mathematical modelling of these high level exposures has been the subject of greatest variability. Models of dose have then been extrapolated using these exposure estimates and range of critical assumptions. Validated data are however limited.

Table 6.2. OSAGWI Estimated upper limit for intakes, kidney concentration, and radiation dose (CEDE)

	Intake (mg)	Exceeds Health Guide	Kidney Concentration (μg/g tissue)	Exceeds Health Guide	Radiation Dose CEDE, rem (mSv)	Exceeds Health Guide
Level I						
Soldiers in or on a US vehicle when a DU munition penetrated it.	[1]237	Yes	[1]4.38	Yes	[1]4.8 (48)	No
Soldiers who entered US vehicles to rescue occupants immediately after friendly-fire DU impacts.	[1]237	Yes	[1]4.38	Yes	[1]4.8 (48)	No
Level II						
Explosive Ordnance Disposal (EOD)	2.5	No	0.05	No	0.016 (0.16)	No
Unit maintenance personnel	7.6	No	0.15	No	0.047 (0.47)	No
Logistics Assistance Representatives (LARs)	2.5	No	0.05	No	0.016 (0.16)	No
Battle Damage Assessment Team (BDAT)	7.6	No	0.15	No	0.047 (0.47)	No
144th Service and Supply Co.	2.5	No	0.05	No	0.016 (0.16)	No
Radiation Control (RADCON) team.	3.8	No	0.075	No	0.023 (0.23)	No
Cleanup at Camp Doha's North Compound.	[2]NR	-	0.095	No	0.065 (0.65)	No
Level III						
Personnel exposed to smoke at Camp Doha.	[2]NR	-	2.8×10^{-7}	No	3.0×10^{-6} (3.0×10^{-5})	No
Personnel exposed to smoke from burning Abrams tanks.	0.28	No	0.007	No	0.007 (0.07)	No
Personnel who entered DU-contaminated equipment.	8.2	[3]No	0.012	No	0.01 (0.1)	No
Personnel exposed to smoke from Iraq's DU-impacted equipment.	0.44	No	0.02	No	0.001 (0.01)	No

Notes:
1. Status means exceeds health guideline or not.
2. NR – not separately reported by PNNL.
3. The total DU intake for 100 vehicle-hours appears to exceed the 8 mg inhalation guideline; however that guideline pertains to soluble uranium, which in this estimate is about 1.1 milligrams. The remainder is insoluble uranium.

Fetter and von Hippel (1999) provide a detailed analysis of exposure and dose based on available evidence. They report that measurements taken inside an M1A1 tank after it was struck by a single 120-mm depleted uranium penetrator corresponded to average and maximum 15-minute intakes of 12 and 26 mg. Also, estimates derived from the concentration of uranium in the urine of 14 soldiers that were in struck vehicles, but who do not have evidence of retained shrapnel, are consistent with inhalation of up to 25 mg of depleted uranium. Fetter and von Hippel (1999) consider that "taking into account various uncertainties and the possibility of multiple penetrator strikes, it is possible that individuals inside struck vehicles could inhale 50 or more milligrams of depleted uranium aerosol." These assessed numbers, particularly those based on measurements of urinary uranium in survivors of the "friendly fire" incidents are a factor of ten less than the upper limit estimates from OSAGWI (2000) and 100 times less than the "worst case" estimates from the Royal Society.

It is also useful to consider the physical considerations for inhalation of large masses of dust required in the estimates described above. This approach has been taken by several authors (UNEP 1999). These authors have noted that instantaneous inhalation of more than one gram (1000 mg) of dust is unendurable, and assuming 10% of the dust to be depleted uranium, the maximal intake of depleted uranium would be 100 mg. This would give a maximal effective dose of 10mSv from acute inhalation of depleted uranium (Royal Society, 2001). Longer exposures to depleted uranium aerosols and dusts may also be assessed and "it is very unlikely that any dust except very close to a point of impact would be more than 10% depleted uranium. An air concentration of 100 mg per cubic metre would be noticeably dusty, and normal breathing rates are of the order of one cubic metre per hour. Hence to inhale 100 mg of depleted uranium, someone would need to inhale dusty air that was heavily contaminated with depleted uranium for ten hours." (Royal Society, 2001, p9). It is unlikely that anyone would incur a dose greater than 10 mSv in these battlefield conditions. The Royal Society worst case scenario of inhalation of 5000 mg of depleted uranium is based on considerably greater theoretical exposures: breathing air carrying 50,000 mg of dust per cubic metre at 50 litres per minute for one minute, followed by 5,000 mg of dust per cubic metre at 50 litres per minute for 10 minutes (Royal Society, 2001 Annexe C). These levels do not appear to be supported by the current though limited information available from "friendly fire" survivors.

Retained Depleted Uranium Fragments

No NATO forces are recorded to have suffered "friendly fire" incidents or have retained fragments of depleted uranium due to service in the Balkans. As a result of "friendly fire" incidents during the Gulf War, US DoD has reported that depleted uranium munitions struck a number of US Bradley Fighting Vehicles and Abrams Tanks (OSAGWI, 1998). As would be expected the probability of death or serious injury is very high in vehicles struck by anti-tank weapons. Of 113 soldiers in about 30 US vehicles struck by depleted uranium penetrators in the Gulf War, 13 were killed and about 50 incurred significant wounds—a casualty rate of over 50 percent (Fetter and von Hippel, 1999). The risks associated with inhaled, ingested or implanted depleted uranium are very small in comparison given the intention of all such strikes to be “total kill”.

In addition to inhaled aerosols generated at impact, personnel in vehicles at the time of impact by a depleted uranium penetrator may also retain fragments of uranium in their bodies. Such fragments deliver a radiation dose to a relatively small volume of surrounding tissue. As the fragments dissolve gradually in body fluids, uranium also is transported to other organs and excreted in the urine.

As discussed in Section Five, the US Baltimore VA Medical Follow-up Program results provide the most relevant information about these survivor’s health and medical conditions. Extensive testing in

1993-1994, 1997, and 1999 has failed to document kidney abnormalities, even in veterans with retained depleted uranium fragments who are excreting elevated levels of uranium in their urine. Their testing included measuring retinol-binding protein and $ß_2$-microglobulin, which would indicate the presence or absence of proximal tubular damage (McDiarmid et al, 2000; McDiarmid, GWI Conference 2001 Washington DC). While these veterans have medical disorders resulting from their wartime injuries, they exhibit none of the known clinical manifestations from uranium's chemical or radiological toxicity. No adverse kidney effects have been observed and no cancers have been recorded in this population (McDiarmid et al, 2000, McDiarmid, 2001).

From the time of the Gulf War through 1998, no Gulf War veterans identified as having Level I (140 people) or Level II (127 people) depleted uranium exposures have been hospitalised in military facilities for kidney disease (nephritis, nephrotic syndrome, and nephrosis) of the type associated with depleted uranium's chemical toxicity (OSAGWI, 2000).

Summary

The current sound medical-scientific evidence provides no evidence to show that a person having external or internal exposure to depleted uranium at any realistic level will develop an illness caused by depleted uranium.

The estimates of depleted uranium intake, chemical dose, and radiation dose calculated by the US DoD (OSAGWI, 2000) for Level II and III participants indicate those veterans experienced air concentrations well below the short-term exposure limits. These estimates are far below any relevant US Federal or industrial guideline for chemical or radiation exposure (OSAGWI, 2000). The toxicology of uranium has been examined in Section Five and the Expert Committee has concluded that harmful medical effects from depleted uranium exposure for Level II or III personnel are not supported by the current sound medical-scientific evidence.

Recent risk assessments by the Royal Society show that while studies of large cohorts of veterans are vitally important to explore and understand the experiences and exposures which may effect the health status of veterans, in the case of exposure to depleted uranium most would have had only very low or negligible exposure. Even during the Gulf War only a very small fraction of the troops present would have received measurable exposure to depleted uranium and based on our current knowledge of uranium toxicity the risk is below measurable levels. Even for their worst case estimates the risk was still so low that no observable increase in lung cancer (or other cancer) mortality would be seen in a cohort of 10,000 veterans followed for 50 years (Royal Society, 2001).

Operation Allied Force utilised an aerial bombing program incorporating depleted uranium munitions to drive the Yugoslav army out of Kosovo in 1999. NATO troops were not positioned in proximity to the areas being bombed and were not present at target sites at the time of the bombing. From the historical data available the Expert Committee considers that it is highly unlikely that any Australian personnel serving with NATO forces in the Balkans would have experienced any more than Level III exposure. On the basis of the available sound medical-scientific evidence and under realistic assumptions of exposure and dose the Expert Committee concluded that depleted uranium could not produce any adverse health effects in Australian troops serving with NATO forces in the Balkans conflict.

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Appendix One

What is sound medical-scientific evidence?

The term "sound medical-scientific evidence" is defined in Section 5AB2 of the Veterans' Entitlements Act (VEA):

"sound medical-scientific evidence", in relation to a particular kind of injury, disease or death, has the meaning given by subsection (2).

(2) Information about a particular kind of injury, disease or death is taken to be **sound medical-scientific evidence** if:

(a) the information:

(i) is consistent with material relating to medical science that has been published in a medical or scientific publication and has been, in the opinion of the Repatriation Medical Authority, subjected to a peer review process; or

(ii) in accordance with generally accepted medical practice, would serve as the basis for the diagnosis and management of a medical condition; and

(b) in the case of information about how that kind of injury, disease or death may be caused - meets the applicable criteria for assessing causation currently applied in the field of epidemiology.

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Appendix Two

Epidemiology

This is the study of variations in disease frequency among population groups, and the factors that influence these variations. The principle objective of this discipline has been to determine factors which may cause or contribute to disease processes in humans, so that preventive measures may be applied.

Epidemiologic observations have a long history, with much work developed through the study of acute epidemic diseases such as cholera and typhoid. The discipline has burgeoned over the latter half of the Twentieth century, with interest in the study of the cause, treatment and prevention of cancer, cardiovascular and other chronic disease, and of course the advent of computer storage and analysis systems. Epidemiology generally seeks to serve the following objectives:

1. Provision of data necessary for planning and evaluating health care
2. Identification of determinants of disease so as to enable prevention
3. Evaluation of methods used to control disease
4. Observation of natural history of disease
5. Classification of disease

The increasing use of epidemiology for compensation purposes introduces questions of "standards of proof" which have social equity and political implications in the decision making process. This requires a legislative base to set the standards of proof. Here the Expert Committee is tasked to examine the nature and strength of the sound medical scientific evidence concerning the adverse health effects of exposure to depleted uranium.

Association and Causation

Association is the term used to describe the statistical dependence between two variables. In epidemiology, it is the degree to which the rate of disease in persons with an exposure of interest is either higher or lower than the rate of disease among those without that exposure. Such an association does not mean, or even imply, that the observed relationship is one of cause and effect. (Hennekens and Buring, 1987).

Making judgements about causality from epidemiologic data involves a logical process which addresses two major areas:

1. Whether for any individual study, the observed association between an exposure and disease is valid. An assessment of validity requires a consideration of the likelihood of alternative explanations for the results and chance (the luck of the draw), bias (any systematic error in the study for example in subject selection, information gathering or reporting), or confounding (the observed effect being due to other variables not adequately considered in study design or analysis of the results*)*; and

2. Whether the body of the evidence considered supports a judgement of causality. In this process standard epidemiological criteria are used (Hennekens and Buring, 1987).

Epidemiologic criteria used to assist in the assessment of causality

The Expert Committee considered the individual studies with respect to the above and then, in considering the available evidence used standard epidemiological criteria alluded to in the Terms of Reference to make a judgement regarding causality. The Bradford Hill criteria (Bradford-Hill, 1965)[1] (and more contemporary versions) are widely accepted in the interpretation of epidemiological studies for the purpose of assessing the possibility of a causal association.

The exact description of these epidemiologic criteria varies between authors and here effort has been made to consider both internal study validity (for individual studies) and factors important in the body of evidence (the applicable evidence available from epidemiological, clinical, toxicological and other research) in these criteria. In assessing the evidence the Expert Committee has used the following epidemiological criteria to decide if any suggested association in fact represents causation.
The criteria include:

1. Statistical significance (ie the possibility of chance being responsible for an apparent association; and study power)
2. Strength of association
3. Consistency of association between studies
4. Possibility of bias in measurement of exposure or outcome
5. Possibility of selection or confounding bias
6. Time sequence
7. Dose response
8. Biological plausibility (theoretical coherence, biological coherence and factual coherence)

Approaches for Epidemiological Study

It is important to stress some key words in the definition of epidemiology, which is in an alternate textbook definition "the study of the distribution and determinants of disease frequency in human populations." (Hennekens and Buring, 1987, p3) "Humans" distinguishes the approach from research using animal or other systems in experiments. "Populations" contrasts the practise of individual investigation as in clinical research. "Frequency" indicates the attempt to quantify the occurrence of disease, and the risk attributable to various causes. The term "distribution and determinants" points to the two major approaches of epidemiology:

1. examination of the distribution of disease frequency in populations (this can produce hypotheses about the causes of disease) known as descriptive studies; and
2. analytical studies which test these hypotheses by reviewing personal characteristics or exposures among individuals within the groups.

[1] Sir Austin Bradford Hill, as well as other prominent statisticians and epidemiologists including Mervyn Susser and Kenneth Rothman have described how the subjective likelihood (or the correct judgement) of a causal relationship is increased when evidence relating to an association meets criteria devised to consider the available evidence.

Descriptive studies
Descriptive studies use population based statistics on mortality, disease incidence, and survival. Other registries eg hospital based may also be useful. Obviously the studies concern populations and not individuals and measures of any exposures are usually broad and may be subject to confounding or interfering factors. Selection of free living populations may introduce biases and confounding into the calculations. Examination of national and international trends, migrant studies and time trends has provided valuable insights into the causation of a number of chronic diseases eg breast, prostate and lung cancers. Such descriptive material is also collected for birth defects, and there is good comparative data available for many of these conditions.

Analytical epidemiology
Analytical epidemiology has provided much useful information concerning the discovery and/or confirmation of a number of lifestyle and other environmental exposures as causes of chronic disease including cancer. Examples of this include cigarette smoking, where, for smokers of 40 or more cigarettes per day there is a risk of lung cancer of more than twenty times that of a nonsmoker. Another well documented example is occupational asbestos exposure and the development of mesothelioma, where the relative risk is well over 100 fold that of the unexposed population.

A. Cohort studies identify groups of individuals with and without a particular exposure, and follow them over time to examine disease incidence and/or mortality rates. These may be current or past exposures. An association is suggested when rates of disease or death differ between the groups. These are able to directly measure incidence and mortality rates related to a particular exposure (especially with prospective design) but they require large numbers of exposed individuals particularly when considering uncommon diseases, before significant differences may be noted.

B. Case-control studies or case-referent studies identify people with a particular disease (case), and a group of people without the disease (controls), and then collect information about past exposures, eg by interview or questionnaire. They provide a method of studying rare diseases but may be subject to recall and other biases, and difficulty in measuring past exposures.

Data Presentation and Interpretation

The odds ratio (OR) is a measure of association used in case control studies to estimate the odds of exposure in cases to the odds of exposure in controls. This approximates but is not synonymous with the "relative risk" (RR) the measure of association used in cohort studies. The term relative risk (RR) is used to describe the comparison of the risk of a known exposed group versus a known unexposed group developing a specific condition. Thus if the relative risk is one the risk is the same for both groups and exposure is not seen to be associated with the development of the particular condition ie there is no increase in the risk of a studied outcome with the exposure of interest.. If the RR (or OR) is 1.5 then the risk for the studied outcome in the exposed versus the unexposed group is increased by 50%. An RR (or OR) of 2 implies a doubling of risk, and an RR (or OR) of less than one implies a reduction of risk. Problems in decision making occur when the described increase in risk is weak (under a two to three fold ie 200-300% increase) and particularly when the relative risk is close to one, eg 1.1 or 1.3 rather than the 2,000% increases for cigarettes and lung cancer and the 10,000% increase for occupational asbestos exposure and mesothelioma. Many epidemiologists are reluctant to accept as real, increases in risk of less than 100% (RR<=2) as likely to be causative unless the "Bradford Hill" types of criteria are stringently applied to the body of evidence pertinent to the putative association, and overall, a considered case can then be made to

support causality. As described, the Expert Committee has chosen to utilise a version of these criteria in its consideration of the evidence regarding depleted uranium exposure and adverse health effects.

Another term, the "confidence interval" (CI), is used to describe the range of relative risk (or odds ratio) rates within which the actual result lies, to within, for example, a 95% probability. Thus if the confidence interval includes one then the result could have occurred due to chance and no true effect may exist. If the 95% confidence limits exclude one it does not exclude the possibility of a chance result, rather it indicates that chance would explain the observed (or a greater) risk estimate only one out of 20 times.

Epidemiologic studies need careful examination. The size of the population studied is important - the larger the sample size the greater the power (or ability) to detect a specified risk, the smaller the sample size the weaker the power. Negative results from small studies may not be conclusive as only large studies may confidently exclude or include low to moderate levels of risk.

When examining any study results, consideration of the possibility of a non-causal association is necessary. The observed association between exposure and disease may result from bias, confounding, chance, or cause-and-effect. Many types of study bias have been described including selection, information, recall, and interviewer bias. Confounders are variables which may themselves account for all or part of an apparent association between an exposure and a disease. They may also obscure an association. Chance is considered above in the discussion of study power and Confidence Intervals. The potential for confounding has also been highlighted earlier.

Study types

Study design has an effect on the quality of evidence which may be gained. In this instance the level of evidence available is at best observational (cohort or case control studies). The following is broadly the division of available study designs and how these could be considered in the information gathering guidelines.

Analytic Studies

1 **Intervention Studies**
- 1a Randomised Controlled Trial
- 1b Controlled Trial

2 **Observational Studies**
- 2a Cohort-Prospective
- 2b Cohort-Retrospective

3 Case Control

Descriptive Studies

4 **Population:** Correlational

5 **Individual:** Case Series, Case Reports

Emphasis is placed on primary research published in the leading peer reviewed journals of either broad or discipline-specific type.

Nature of the Evidence, Association and Causation

The Expert Committee has been asked to consider the nature and strength of the evidence. Any assessment of the literature may reveal associations. An observed association between exposure and disease may result from bias, confounding, chance, or cause-and-effect. The assessment of cause and effect is then made applying standard epidemiological criteria to body of evidence

1. Statistical significance and power

If the criterion of statistical significance is satisfied then the evidence is supportive. The failure of a test to reach statistical significance in the presence of adequate statistical power provides evidence against the association, however in the absence of adequate statistical power it may not necessarily detract from the association.

2. Strength of association

The greater the strength of association the more likely it is to be causal. Confounding is less likely to explain a strong association because the strength of the association between the confounding variable and the outcome must also be strong. While a strong association is highly supportive of causality. A weak association may not necessarily detract from the evidence of causality however adequate consideration of potential confounding or bias is essential.

3. Consistency of replication

Consistency of the evidence or lack of evidence in the face of study diversity in time, place, circumstances and population, as well as research design, strongly supports or detracts from a causal hypothesis.

4. Possibility of bias in measurement of exposure or outcome

Consideration of any systematic error in the study in information gathering or in reporting of the assessment of the exposure or outcome under investigation. Absence of bias in the studies considered to show a positive association supports the existence of a putative association. The presence of bias detracts from the conclusions which may be drawn from the information.

5. Possibility of selection or confounding bias

Consideration of any systematic error in the study in subject selection; or the possibility of the observed effect being due to other variables not adequately considered in study design or analysis of the results. Absence of bias or confounding in the studies considered to show a positive association supports the existence of a putative association. The presence of bias or uncontrolled confounding detracts from the conclusions which may be drawn from the information.

6. Time sequence

The chemical exposure must precede the disease or injury. This criterion is compatible with, but does not necessarily support causality. Reversal of the order of exposure and disease or injury is the most decisive basis available for rejection of causality.

7. Dose response

A response which is in proportion to the chemical level of exposure is strongly persuasive of a causal relation. However, its absence does not necessarily detract from the association.

8. Biological plausibility (theoretical coherence, biological coherence and factual coherence)

Theoretical coherence: Findings plausible in terms of preexisting theory are supportive of the association. Conversely, findings that are implausible in terms of pre-existing theory detract from the evidence.

Factual coherence: Compatibility of a new result with pre-existing facts is supportive of the association. Incompatible pre-existing facts strongly detract from evidence of causality.

Biological coherence: Pre-existing knowledge which identifies a mechanism by which the chemical exposure may produce the disease or injury is supportive of case for the association being causal. Observations from species other than humans may also be used to support the potential mechanism of action. Incoherence between biological knowledge and study observations detracts from the case for a causal association.

Strength of Evidence

The strength of the evidence may be described in a number of ways.

There are a number of classifications which may be referred to in this process. The IARC classification has wide recognition. Professor Holman (1997) in his Technical Appendix to the Pearce Report notes the falsificationist approach of the IARC and that "the results of application of the IARC system are frequently translated into public policy through regulatory decisions of governments to control exposures to carcinogenic substances. It classifies a body of empirical information on a putative causal relationship as providing sufficient evidence of carcinogenicity; limited evidence of carcinogenicity; inadequate evidence of carcinogenicity; or evidence suggesting lack of carcinogenicity in humans".

IARC definitions (IARC, 2000):

(i) *Carcinogenicity in humans*

The applicability of an evaluation of the carcinogenicity of a mixture, process, occupation or industry on the basis of evidence from epidemiological studies depends on the variability over time and place of the mixtures, processes, occupations and industries. The Working Group seeks to identify the specific exposure, process or activity which is considered most likely to be responsible for any excess risk. The evaluation is focused as narrowly as the available data on exposure and other aspects permit.

The evidence relevant to carcinogenicity from studies in humans is classified into one of the following categories:

Sufficient evidence of carcinogenicity: The Working Group considers that a causal relationship has been established between exposure to the agent, mixture or exposure circumstance and human cancer. That is, a positive relationship has been observed between the exposure and cancer in studies in which chance, bias and confounding could be ruled out with reasonable confidence.

Limited evidence of carcinogenicity: A positive association has been observed between exposure to the agent, mixture or exposure circumstance and cancer for which a causal interpretation is considered by the Working Group to be credible, but chance, bias or confounding could not be ruled out with reasonable confidence.

Inadequate evidence of carcinogenicity: The available studies are of insufficient quality, consistency or statistical power to permit a conclusion regarding the presence or absence of a causal association between exposure and cancer, or no data on cancer in humans are available.

Evidence suggesting lack of carcinogenicity: There are several adequate studies covering the full range of levels of exposure that human beings are known to encounter, which are mutually consistent in not showing a positive association between exposure to the agent, mixture or exposure circumstance and any studied cancer at any observed level of exposure. A conclusion of 'evidence suggesting lack of carcinogenicity' is inevitably limited to the cancer sites, conditions and levels of exposure and length of observation covered by the

available studies. In addition, the possibility of a very small risk at the levels of exposure studied can never be excluded.

In some instances, the above categories may be used to classify the degree of evidence related to carcinogenicity in specific organs or tissues.

(ii) *Carcinogenicity in experimental animals*

The evidence relevant to carcinogenicity in experimental animals is classified into one of the following categories:

Sufficient evidence of carcinogenicity: The Working Group considers that a causal relationship has been established between the agent or mixture and an increased incidence of malignant neoplasms or of an appropriate combination of benign and malignant neoplasms in (a) two or more species of animals or (b) in two or more independent studies in one species carried out at different times or in different laboratories or under different protocols. Exceptionally, a single study in one species might be considered to provide sufficient evidence of carcinogenicity when malignant neoplasms occur to an unusual degree with regard to incidence, site, type of tumour or age at onset.

Limited evidence of carcinogenicity: The data suggest a carcinogenic effect but are limited for making a definitive evaluation because, e.g. (a) the evidence of carcinogenicity is restricted to a single experiment; or (b) there are unresolved questions regarding the adequacy of the design, conduct or interpretation of the study; or (c) the agent or mixture increases the incidence only of benign neoplasms or lesions of uncertain neoplastic potential, or of certain neoplasms which may occur spontaneously in high incidences in certain strains.

Inadequate evidence of carcinogenicity: The studies cannot be interpreted as showing either the presence or absence of a carcinogenic effect because of major qualitative or quantitative limitations, or no data on cancer in experimental animals are available.

Evidence suggesting lack of carcinogenicity: Adequate studies involving at least two species are available which show that, within the limits of the tests used, the agent or mixture is not carcinogenic. A conclusion of evidence suggesting lack of carcinogenicity is inevitably limited to the species, tumour sites and levels of exposure studied.

(*b*) *Other data relevant to the evaluation of carcinogenicity and its mechanisms*

Other evidence judged to be relevant to an evaluation of carcinogenicity and of sufficient importance to affect the overall evaluation is then described. This may include data on preneoplastic lesions, tumour pathology, genetic and related effects, structure-activity relationships, metabolism and pharmacokinetics, physicochemical parameters and analogous biological agents.

Data relevant to mechanisms of the carcinogenic action are also evaluated. The strength of the evidence that any carcinogenic effect observed is due to a particular mechanism is assessed, using terms such as weak, moderate or strong. Then, the Working Group assesses if that particular mechanism is likely to be operative in humans. The strongest indications that a particular mechanism operates in humans come from data on humans or biological specimens obtained from exposed humans. The data may be considered to be especially relevant if they show that the agent in question has caused changes in exposed humans that are on the causal pathway to carcinogenesis. Such data may, however, never become available, because it is at least conceivable that certain compounds may be kept from human use solely on the basis of evidence of their toxicity and/or carcinogenicity in experimental systems.

For complex exposures, including occupational and industrial exposures, the chemical composition and the potential contribution of carcinogens known to be present are considered by the Working Group in its overall evaluation of human carcinogenicity. The Working Group also determines the

extent to which the materials tested in experimental systems are related to those to which humans are exposed.

(*c*) *Overall evaluation*

Finally, the body of evidence is considered as a whole, in order to reach an overall evaluation of the carcinogenicity to humans of an agent, mixture or circumstance of exposure.

An evaluation may be made for a group of chemical compounds that have been evaluated by the Working Group. In addition, when supporting data indicate that other, related compounds for which there is no direct evidence of capacity to induce cancer in humans or in animals may also be carcinogenic, a statement describing the rationale for this conclusion is added to the evaluation narrative; an additional evaluation may be made for this broader group of compounds if the strength of the evidence warrants it.

The agent, mixture or exposure circumstance is described according to the wording of one of the following categories, and the designated group is given. The categorization of an agent, mixture or exposure circumstance is a matter of scientific judgement, reflecting the strength of the evidence derived from studies in humans and in experimental animals and from other relevant data.

- ***Group 1: The agent (mixture) is carcinogenic to humans.***
  The exposure circumstance entails exposures that are carcinogenic to humans.

This category is used when there is *sufficient evidence* of carcinogenicity in humans. Exceptionally, an agent (mixture) may be placed in this category when evidence of carcinogenicity in humans is less than sufficient but there is *sufficient evidence* of carcinogenicity in experimental animals and strong evidence in exposed humans that the agent (mixture) acts through a relevant mechanism of carcinogenicity.

- ***Group 2***

This category includes agents, mixtures and exposure circumstances for which, at one extreme, the degree of evidence of carcinogenicity in humans is almost sufficient, as well as those for which, at the other extreme, there are no human data but for which there is evidence of carcinogenicity in experimental animals. Agents, mixtures and exposure circumstances are assigned to either group 2A (probably carcinogenic to humans) or group 2B (possibly carcinogenic to humans) on the basis of epidemiological and experimental evidence of carcinogenicity and other relevant data.

- ***Group 2A: The agent (mixture) is probably carcinogenic to humans.***
  The exposure circumstance entails exposures that are probably carcinogenic to humans.

This category is used when there is *limited evidence* of carcinogenicity in humans and *sufficient evidence* of carcinogenicity in experimental animals. In some cases, an agent (mixture) may be classified in this category when there is *inadequate evidence* of carcinogenicity in humans and *sufficient evidence* of carcinogenicity in experimental animals and strong evidence that the carcinogenesis is mediated by a mechanism that also operates in humans. Exceptionally, an agent, mixture or exposure circumstance may be classified in this category solely on the basis of *limited evidence* of carcinogenicity in humans.

- ***Group 2B: The agent (mixture) is possibly carcinogenic to humans.***
  The exposure circumstance entails exposures that are possibly carcinogenic to humans.

This category is used for agents, mixtures and exposure circumstances for which there is *limited evidence* of carcinogenicity in humans and less than *sufficient evidence* of carcinogenicity in experimental animals. It may also be used when there is *inadequate evidence* of carcinogenicity in humans but there is *sufficient evidence* of carcinogenicity in experimental animals. In some instances, an agent, mixture or exposure circumstance for which there is *inadequate evidence* of carcinogenicity in humans but *limited evidence* of carcinogenicity in experimental animals together with supporting evidence from other relevant data may be placed in this group.

- ***Group 3: The agent (mixture or exposure circumstance) is not classifiable as to its carcinogenicity to humans.***

This category is used most commonly for agents, mixtures and exposure circumstances for which the *evidence of carcinogenicity is inadequate* in humans and *inadequate or limited* in experimental animals.
Exceptionally, agents (mixtures) for which the *evidence of carcinogenicity is inadequate* in humans but *sufficient* in experimental animals may be placed in this category when there is strong evidence that the mechanism of carcinogenicity in experimental animals does not operate in humans.
Agents, mixtures and exposure circumstances that do not fall into any other group are also placed in this category.

- ***Group 4: The agent (mixture) is probably not carcinogenic to humans.***

This category is used for agents or mixtures for which there is *evidence suggesting lack of carcinogenicity* in humans and in experimental animals. In some instances, agents or mixtures for which there is *inadequate evidence* of carcinogenicity in humans but *evidence suggesting lack of carcinogenicity* in experimental animals, consistently and strongly supported by a broad range of other relevant data, may be classified in this group.

Classifications of Evidence of Causality (Holman, 1997)

I Sufficient evidence of causality: The evidence indicates that an association exists between the factor and disease or injury in which chance, confounding and bias can be ruled out with reasonable confidence.

II Limited evidence of causality: Association has been observed between the factor and disease or injury for which a causal interpretation is considered to be credible, but chance, confounding or other bias cannot be ruled out with reasonable confidence.

III Inadequate evidence of causality: The available evidence is of insufficient quality, consistency or statistical power to permit a conclusion regarding the presence or absence of a causal association.

IV Evidence suggesting lack of causality: There are several adequate studies covering the full range of levels of exposure to the factor that human populations are known to encounter, which are consistent in not showing as association between exposure to the factor and the disease or injury.

National Academy of Science

The NAS committee (Fulco et al, 2000) use a standard format to assess the qualitative and quantitative aspects of evidence and to categorise association utilised a fiver tier division adapted from the International Agency for Research on Cancer (IARC). The five categories are:

1. Sufficient evidence of a causal relationship
2. Sufficient evidence of an association
3. Limited/suggestive evidence of an association
4. Inadequate/insufficient evidence to determine whether an association exists
5. Limited/suggestive evidence of no association

where the following applies:

"Sufficient evidence of a Causal Relationship

Evidence is sufficient to conclude that a causal relationship between the exposure to a specific agent and a health outcome in humans. The evidence fulfills the criteria for sufficient evidence of association and satisfies several of the criteria used to assess causality: strength of associationm dose-response relationship, consistency of association, temporal relationship, specificity of association and biological plausibility.

Sufficient evidence of an association

Evidence is sufficient to conclude that there is a positive association. That is, a positive association has been observed between an exposure to a specific agent and a health outcome in human studies in which chance, bias and confounding could be ruled out with reasonable confidence.

Limited/suggestive evidence of an association

Evidence is suggestive of an association between exposure to a specific agent and a health outcome but is limited because chance, bias and confounding could not be ruled out with confidence.

Inadequate/insufficient evidence to determine whether an association exists

The available studies are of insufficient quality, consistency, or statistical power to permit a conclusion regarding the presence or absence of an association between an exposure to a specific agent and a health outcome in humans.

Limited/suggestive evidence of no association

There are several adequate studies, covering the full range of levels of exposure that human beings are known to encounter, that are mutually consistent in not showing a positive association between exposure to a specific agent and a health outcome at any level of exposure. A conclusion of no association is inevitably limited to the conditions, level of exposure, and length of observation covered by the available studies. In addition, the possibility of a very small elevation in risk at the levels of exposure studied can never be excluded."

(Fulco et al, 2000, pp83-4)

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Appendix Three

The physical and chemical properties of uranium and selected uranium compounds

Table A3.1:
Absorption (solubility) types for uranium compounds (ICRP-71, 1995b)

Type	Typical compounds
F = fast	UF_6, UO_2F, $UO_2(NO_3)_2$
M = moderate	UO_3, UF_4, UCl_4, U_3O_8
S = slow	UO_2

Table A3.2:
The physical and chemical properties of uranium and selected uranium compounds (ATSDR, 1999b)

Property	Uranium	Uranium Dioxide	Uranium Trioxide	Uranium Octaoxide	Uranium Tetrafluoride	Uranium Hexafluoride
Atomic/ Molecular Wt	238.0289	270.03	286.03	842.08	314.02	352.02
Chemical Formula	U	UO_2	UO_3	U_3O_8	UF_4	UF_6
Synonyms	Uranium I	Uranium Oxide	Uranyl Oxide	Uranium Octaoxide	Uranium Fluoride	UN2977; uranium fluoride (fissile)
CAS No	7440-61-1	1344-57-6	1344-58-7	1344-59-8	10049-14-6	7783-81-5
Colour	Silvery	Brown-Black	Yellow-Red	Olive green-black	Green	Colourless
Physical State	Solid	Solid	Solid	Solid	Solid	Solid
Melting Point ^oC	1135	2878	Decomposes	Decomposes at 1300	960	64.5-64.8
Boiling Point ^oC	4131	No data	NR	NR	No data	56.2
Autoignition temperature	20% (cloud), 100 ^oC (layer)	NR	NR	NR	NR	NR
Solubility **Water** **Other solvents**	Insoluble Soluble in acids	Insoluble Soluble in HNO_3	Insoluble Soluble in HCl, HNO_3	Insoluble Soluble in HNO_3, H_2SO_4	Insoluble Soluble in concentrated acids/alkalis	Decomposes Soluble in CCl_4 and chloroform
Density g/cm^3	18.95	10.96	7.29	8.30	6.70	4.68 at 21^oC
Conversion Factor	1microgm = 0.67pCi	1microgm = 0.59pCi	1microgm = 0.56pCi	1microgm = 0.57pCi	1microgm = 0.45pCi	1microgm = 0.45pCi

TableA3.2: The physical and chemical properties of uranium and selected uranium compounds (continued)

Property	Uranium Tetrachloride	Uranyl Fluoride	Uranyl acetate, dihydrate	Uralyl nitrate, hexahydrate	Ammonium diuranate	Uranium peroxide
Atomic/ Molecular Wt	379.84	308.03	424.15	502.13	624.22	302.03
Chemical Formula	UCl_4	UO_2F_2	$UO_2(CH_3COO)_2.2H_20$	$UO_2(NO_3)_2.6H_20$	$(NH_4)_2U_2O_7$	UO_4
Synonyms	Uranium (IV) chloride	Uranium oxyfluroide; uranium fluoride oxide	Bis(Acetate-B) dioxouranium	Bis(Nitrate-O) dioxouranium; hexahydrate	Ammonium uranate (IV)	No data
CAS No	10026-10-5	13536-84-0	541-09-3	13520-83-7	7783-22-4	19525-15-6
Colour	Dark green	Pale yellow	Yellow	Yellow	Reddish yellow	Pale yellow
Physical State	Solid	Solid	Solid	Solid	Solid	Solid
Melting Point ^oC	590	Decomposes at 300 ^oC	Loses 2 H_20 at 110	60.2	No data	Decomposes
Boiling Point ^oC	792	NR	Decomposes at 275 ^oC	Decomposes at 100 ^oC	No data	No data
Autoignition temperature	NR	NR	NR	NR	NR	NR
Soluability **Water** **Other solvents**	 Soluble Soluble in ethanol	 Soluble Soluble in ethanol	 7.7g/100 ml at 15 ^oC Soluble in ethanol	 Miscible in water at 60 ^oC Soluble in ethanol	 Practically Insoluble Soluble in acids	 Decomposes No data
Density g/cm^3	4.87	6.37	2.893 at 15 ^oC	2.807 at 13 ^oC	No data	No data
Conversion Factor	1microgm = 0.42pCi	1microgm = 0.52pCi	1microgm = 0.38pCi	1microgm = 0.32pCi	1microgm = 0.51pCi	1microgm = 0.53pCi

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Appendix Four

Uranium Decay Series

Uranium-238 series (ICRP 1983)

Nuclide	Type of decay	Half-life	Average emitted energy per transformation		
			Alpha energy (MeV)	Beta energy (MeV)	Gamma energy (MeV)
Uranium-238 ^{238}U	α	4.479 10^9y	4.26	0.010	0.001
↓					
Thorium-234 ^{234}Th	β	24.1 d	-	0.059	0.009
↓					
*Protactinium-234m ^{234m}Pa (98.87%) +	β	1.17 m	-	0.820	0.013
Protactinium-234 (0.13%)	β	6.7 h		0.422	1.75
↓					
Uranium-234 ^{234}U	α	2.45 10^5y	4.84	0.013	0.002

*Branched decay

Uranium-235 series (ICRP 1983)

Nuclide	**Type of decay**	**Half-life**	**Average emitted energy per transformation**		
			Alpha energy (MeV)	**Beta energy (MeV)**	**Gamma energy (MeV)**
Uranium-235 ^{235}U	α	7.04 10^8y	4.47	0.048	0.154*
↓					
Thorium-231 ^{231}Th	β	25.52 h	-	0.163	0.026

*also a 0.186 Mev

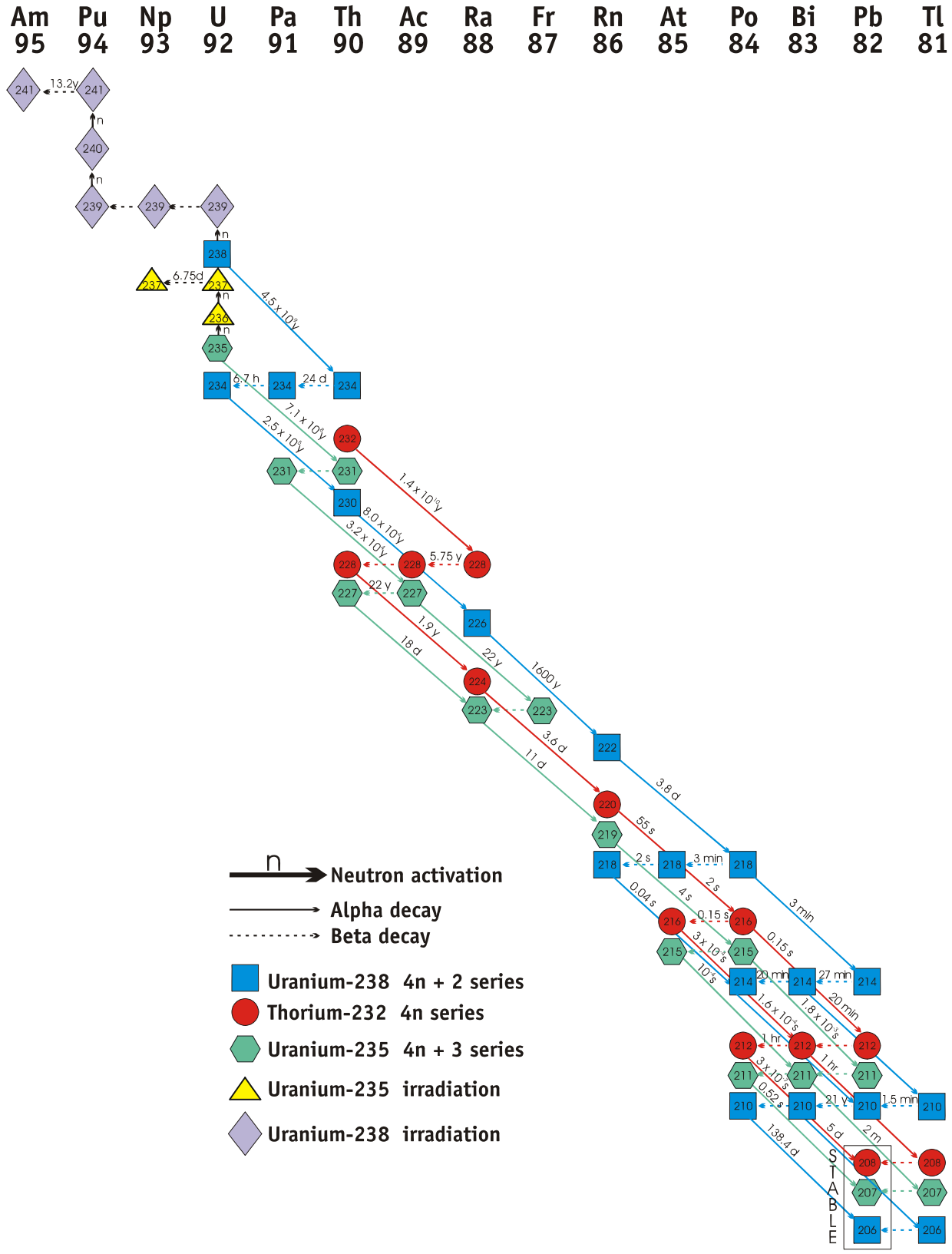

Figure A4.1 Uranium Decay Series and Neutron Irradiation of Uranium

Appendix Five

Ionizing Radiation

Glossary

(ATSDR, 1999a; ATSDR, 1999b; European Commission Report, 2001)

Absorbed dose (D): the energy absorbed per unit mass averaged over a tissue or an organ. The unit for absorbed dose is the gray.

Becquerel (Bq): the special name of the unit of activity. One Becquerel is equivalent to one transition (radioactive decay) per second: 1 Bq = 1 s^{-1}

Committed effective dose (E): the sum of the committed organ or tissue equivalent doses (H) resulting from an intake, each multiplied by the appropriate tissue weighting factor. The unit for committed effective dose is the Sievert.

Committed equivalent dose (H): the sum over time (years) of the equivalent dose rate in a tissue or organ that will be received by an individual as a result of an intake. A period of 50 years is assumed for adults and up to age 70 for children. The unit for committed equivalent dose is Sievert.

Deterministic radiation detriment: effect of radiation that has a threshold dose above which damage is certain to occur (contrary to stochastic risk) and below which there is no effect.

Decay series: the decay of a radionuclide by spontaneous transformation produces progeny nuclides. The rate of decay is specific to each nuclide and is expressed as the activity in Becquerel (Bq) indicating the number of transformations per second.

Dose: a measure of the radiation received or "absorbed" by a target (man, biota, …). The quantities termed absorbed dose, organ dose, equivalent dose, effective dose, committed equivalent dose or committed effective dose are used depending on the context. The modifying terms are often omitted when they are not necessary for defining the quantity of interest.

Dose limit: maximum reference for the doses resulting from the exposure of workers, apprentices and students and members of the public to ionizing radiation covered by the BSS Directive. It applies to the sum of the relevant doses from external exposures in the specified period and the 50-year committed doses (up to age 70 for children) from intakes in the same period.
The limit on effective dose for exposed workers is 100 mSv in a consecutive five-year period, subject to a maximum effective dose of 50 mSv in any single year.
The limit on equivalent dose for the skin is 500 mSv in a year. It applies to the dose averaged over any area of 1 cm^2, regardless of the area exposed.
The dose limit for members of the public is 1 mSv in a year (effective dose); in a single year and in special circumstances a higher effective dose may be authorised provided that the average over five consecutive years does not exceed 1 mSv per year. The limit on equivalent dose for the skin is 50 mSv in a year.
The dose limits to organs (e.g. skin) preclude the occurrence of deterministic effects. The limits on effective dose are set so as to ensure an equitable distribution of doses among a given population for planned practices. Dose limits do not apply in de facto situations where

intervention may be required.

The following qualifications are used when comparing doses to the dose limits for justified practices:

- above the dose limit: unacceptable
- below the dose limit: tolerable
- after optimisation of the protection measures: acceptable
- more than a factor 100 below the dose limit for members of the public (i.e. 10 μSv): negligible.

There is no standard qualification for doses below "negligible"; such doses are of no radiation protection concern but are nevertheless calculated (down to the order of nSv) for the sake of completeness of the assessment.

Effective dose (E): the sum of the weighted (over organ and type of radiation) equivalent doses in all the tissues and organs of the body from internal and external irradiation. The unit for effective dose is the Sievert.

Environment pathways: the different possible ways one can imagine for transfer of radioactivity in the environment from the source term to man.

Equivalent dose (H_T): the absorbed dose, in a tissue or organ weighted for the type and quality of radiation. The unit for equivalent dose is the Sievert. The radiation weighing factor equals 1 for γ and β rays, 20 for α particles.

Exposure: the process of being subject to irradiation. Exposure can be either external or internal exposure (see dose)

Intervention: a human activity that prevents or decreases the exposure of individuals to radiation from sources which are not part of a practice or which are out of control, by acting on sources (e.g. clean up), transmission pathways and individuals themselves

Ionizing radiation: the transfer of energy in the form of particles or electromagnetic capable of producing ions directly or indirectly.

Isotopes: isotopes of an element have the same number of protons but different numbers of neutrons forming the atomic nucleus. Accordingly they have a different mass; radioactive isotopes are called radionuclides and are identified by their mass numbers.

Natural radiation sources: sources of ionizing radiation from natural terrestrial or cosmic origin.

Progeny: see decay series and secular equilibrium

Reprocessing: when uranium fuel (enriched in ^{235}U) is used in a nuclear power plant, the amount of ^{235}U decreases and transuranic or fission products (^{236}U, ^{239}Pu, ^{234}Am, ^{99}Tc) will appear in the fuel. After some time the percentage of ^{235}U is to low and the downloaded fuel becomes waste or can be reprocessed to recover fissile material. The recovered uranium can be enriched for reuse as nuclear fuel. The residue of this enrichment is a type of depleted uranium (DU) which therefore contains ^{236}U and possibly very small amounts of transuranics and fission products.

Secular equilibrium: as a result of radioactive decay, a nuclide produces progeny nuclides. With time, progeny and parent nuclides are in equilibrium when production and decay are at the same rate, in equilibrium, the activities of parent and progeny nuclides are identical.

Sievert (Sv): the special name of the unit of equivalent or effective dose. One Sievert is equivalent to one joule per kilogram: $1\ Sv = 1\ J.kg^{-1}$,

One Sievert equals to thousand milliSievert (1000 mSv) or one million microSievert (1,000,000 μSv)

Stochastic risk: a risk of radiation effect that is statistically related to the dose; an increase in dose gives an increased probability of the effect (contrary to deterministic risk); the main stochastic risks are cancer induction and genetic detriment.

Tonne (t): this is a unit for weight; 1 tonne = 1000 kilograms (kg), 1 kg = 1000 grams (g) and 1,000,000 μg

Transuranics, Fission products: see reprocessing

Radiation Unit Conversion Chart

10 μSv	=	0.01 mSv			=	0.001 rem	=	1 mrem
100 μSv	=	0.1 mSv			=	0.01 rem	=	10 mrem
1,000 μSv	=	1 mSv	=	0.001 Sv	=	0.1 rem	=	100 mrem
		10 mSv	=	0.01 Sv	=	1 rem	=	1,000 mrem
		100 mSv	=	0.1 Sv	=	10 rem		
		1,000 mSv	=	1 Sv	=	100 rem		
				10 Sv	=	1,000 rem		
10 μGy	=	0.01 mGy			=	0.001 rad	=	1 mrad
100 μGy	=	0.1 mGy			=	0.01 rad	=	10 mrad
1,000 μGy	=	1 mGy	=	0.001 Gy	=	0.1 rad	=	100 mrad
		10 mGy	=	0.01 Gy	=	1 rad	=	1,000 mrad
		100 mGy	=	0.1 Gy	=	10 rad		
		1,000 mGy	=	1 Gy	=	100 rad		
				10 Gy	=	1,000 rad		

NOTE: Sievert is equivalent to rem; Gray is equivalent to rad.

Radiation Dose from Natural Sources (UNSCEAR 2000)

All living organisms are continually exposed to ionizing radiation, which has always existed naturally. The sources of that exposure are cosmic rays that come from outer space and from the surface of the Sun, terrestrial radionuclides that occur in the Earth's crust, in building materials and in air, water and foods and in the human body itself. Some of the exposures are fairly constant and uniform for all individuals everywhere, for example, the dose from ingestion of potassium-40 in foods. Other exposures vary widely depending on location. Cosmic rays, for example, are more intense at higher altitudes, and concentrations of uranium and thorium in soils are elevated in localized areas. Exposures can also vary as a result of human activities and practices. In particular, the building materials of houses and the design and ventilation systems strongly influence indoor levels of the radioactive gas radon and its decay products, which contribute significantly to doses through inhalation."

The major portion of the body burden of the decay series of uranium in the general population comes from ingestion of food and drinking water, giving a committed effective dose of 110 μSv per year for adults as compared to 5.8 μSv through inhalation, excluding radon (1.2 mSv).

This dose corresponds to 5% of the average annual dose due to internal and external exposure to natural sources of radiation (2.4 mSv). It relates essentially to ^{210}Pb and ^{210}Po; ^{238}U, with an annual ingestion of 0.5 mg per year, accounts only for 0.25 μSv and 0.021 μSv for inhalation. External exposure from all natural U^{238} in soil is also negligible. U^{238} series together with other primordial radionuclides, Th^{232} and K^{40}, cause a world-average annual external exposure of about 1 mSv per year.

The annual worldwide per caput effective dose is determined by adding the various components, as summarized in Table A5.1. The annual global per caput effective dose due to natural radiation sources is 2.4 mSv. However, the range of individual doses is wide. In any large population about 65% would be expected to have annual effective doses between 1 mSv and 3 mSv, about 25% of the population would have annual effective doses less than 1 mSv and 10% would have annual effective doses greater than 3 mSv."

Table A5.1:

Average radiation dose from natural sources

(UNSCEAR 2000)

Source	Worldwide average annual effective dose (mSv)	Typical range (mSv)
External exposure		
Cosmic rays	0.4	0.3-1.0 [a]
Terrestrial gamma rays	0.5	0.3-0.6 [b]
Internal exposure		
Inhalation (mainly radon)	1.2 *	0.2-10 [c]
Ingestion	0.3 **	0.2-0.8 [d]
Total	2.4	1-10

[a] Range from sea level to high ground elevation

[b] Depending on radionuclide composition of soil and building materials

[c] Depending on indoor accumulation of radon gas

[d] Depending on radionuclide composition of foods and drinking water

* 0.0058 mSv due to ^{238}U decay series

** 0.110 mSv due to ^{238}U decay series

Radiation Dose from Some Other Sources

Radiation is an inevitable component in our environment and is capable of both positive and negative effects. For members of the general population the largest source of exposure to ionizing radiation is natural in origin and comes from the radioactive gas radon, usually inside houses (National Radiological Protection Board, 1998). In addition to accounting for around half the average annual dose from all sources of ionizing radiation, radon exposures are variable, and doses that are currently illegal in the nuclear industry are not uncommon in high radon areas of the UK such as Devon and Cornwall.

A large body of epidemiological data exists on the risks of radiation exposure. This has been compiled from observations including on the survivors of the atomic bombs at Hiroshima and Nagasaki, on workers receiving occupational exposures, and on patients exposed to ionizing radiation during diagnosis or treatment (UNSCEAR, 1994; UNSCEAR, 2000). There is often, however, a marked gap between perceptions of the size of these risks and the values estimated from the data.

Darby, in a recent editorial points out that "people are usually surprised to learn that the atomic bombs at Hiroshima and Nagasaki from 1950 to 1990 caused a total of only around 400 deaths from cancer in the roughly 50 000 exposed survivors. This is just under 10% of all deaths from cancer occurring in that population" (Darby, 1999).

While it is clear high doses of ionizing radiation can induce carcinogenesis, the effects of chronic exposure to low levels of ionizing radiation are not clear. Several models have been put forward to extrapolate the risk at low doses of radiation from high dose exposures. Safety levels for radiation workers have been determined using these models. However, there is no convincing evidence low-dose radiation is harmful. Some believe that exposure to low levels of radiation may yield beneficial effects, a process known as hormesis. Hormesis is a stimulatory effect brought about by low-level exposure to a substance that is toxic at high levels (Murphy, 1991).

Table A5.2:

Average contribution of various sources of natural background radiation in the world. (Mettler and Upton, 1995 p39)

	Annual Effective Dose	
	In Areas of Normal Background	In Areas of Elevated Exposures
Element of exposure	mSv	mSv
Cosmic rays	0.38*	2.0
Terrestrial gamma rays	0.46	4.3
Internal irradiation	0.25	0.6
Radon and its decay ıcts		
Inhalation of radon 222	1.2	10
Inhalation of radon 220	0.07	0.1
Ingestion of radon 222	0.005	0.1
Total	**2.5**	#

* Includes 10 microSv from cosmogenic radionuclides.
# A total value is not given because elevated doses from all elements of exposure would not be expected to coincide at a single location.

Table A5.3:

Cosmic Ray Doses to a Person Flying in Aircraft (Dose/Roundtrip)

(Mettler and Upton, 1995 p45)

Route	Subsonic Flight	Supersonic Flight
	μ Gy	μ Gy
Los-Angeles-Paris	48	37
Chicago-Paris	36	26
New York-Paris	31	24
New York-London	29	22
Los Angles-New York	19	13

Comparison of some medical investigations are presented below.

Table A5.4:

Estimated Fluoroscopy Dose Rates. (Mettler and Upton, 1995 p402)

Examination	Gy/examination
Barium swallow	0.085
Uppergastrointestinal	0.021
Barium enema	0.020
Chest	0.012

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Appendix Six

Development of Depleted Uranium Munitions

Depleted uranium munitions have been developed as part of the armamentarium of conventional weapons. During the late 1950s, the primary material used for armour-piercing projectiles was tungsten carbide. While this was a great improvement over the preceding high carbon steel, tungsten munitions could break up before penetrating layered armour. This lead to the development of new alloys and materials capable of penetrating any armour and a succession of metal alloys were evaluated (OSAGWI, 1998, Tab D).

In the early 1960s, the US Army tested a four-alloy "UQuad" containing depleted uranium in experimental tests on the 105mm and 120mm Delta Armour Piercing Fin Stabilised, Discarding Sabots. By the mid-1970s the superiority of depleted uranium rounds in penetration of hard targets had become clear (OSAGWI, 1998, Tab D).

UK sources indicate that military trials using depleted uranium ammunitions began at Eskmeals in 1963. Later depleted uranium ammunition was tested at West Freugh (ceasing in 1990) and into the sea at Kirkcudbright (Annex A and B to 585/1/3/623/HDRPS dated 1 Dec 1993 - Responses to Lord Parry's Questions: House of Lords 1993 United Kingdom Ministry of Defence (2001). Depleted Uranium - Documents explaining the Ministry of defence position on the risks and health hazards. Ministry of Defence WebSite <www.mod.uk/index.php3?page=2441>).

OSAGWI, 1998 records that since the selection of depleted uranium for the XM774 cartridge, all major developments in tank ammunition have selected depleted uranium, including the 105mm M833 series and the 120mm M829 series (the latter being the primary anti-armour round used in the Gulf War). The latest generation of the 105mm M900 series and the 25mm M919 for the Bradley Fighting Vehicle are depleted uranium alloy munitions.

An extensive US research effort in the 1970s saw the development and evaluation of depleted uranium alloyed with ¾ percent by weight titanium (U-3/4Ti). This alloy allowed the depleted uranium core to withstand high acceleration without breaking up. In the early 1970s the US Airforce developed the GAU-8/A air to surface gun system for the A-10 close air support aircraft. This aircraft was designed to counter an attack by Soviet/Warsaw Pact armoured formations on NATO's Central Region, and was literally designed and built around the GAU-8 gun system. This eight-barrelled 30-mm cannon was designed to blast through the top armour of even the heaviest enemy tanks and to enhance the new cannon's striking power, the USAF used the depleted uranium U-3/4Ti, a 30mm round (OSAGWI, 1998).

These 30 mm rounds are composed of a propellant charge and a solid depleted uranium tip (known as the 'penetrator') weighing 275-300 gm, which is coated on the outside with aluminium, 0.8mm thick (Lockheed Martin, 1995 in WHO, 2001a). The penetrator does not contain an explosive charge, instead it relies on the pyrophoric properties of uranium. The kinetic energy of the 30 mm round is equivalent to about 0.1 pounds of TNT and on contact with a hard target a large proportion of this kinetic energy is converted into heat in less than a millisecond (Fetter and von Hippel, 1999). The impact of the penetrator at high velocity results in its fragmentation and a portion becomes aerosolized. Sufficient heat is generated to ignite the aerosolized uranium metal. The penetrator fragments then burn vigorously at a high temperature and, in effect, melt their way

through steel plating into the interior of an armoured vehicle. Once inside the confined space of a tank, the heat from the burning penetrator fragments ignites flammable vapours and munitions within the vehicle. The ignition of flammable components within a tank is reinforced by the use of high explosive rounds fired in conjunction with depleted uranium ones (OSAGWI, 2000).

OSAGWI note that a "comprehensive Environmental Assessment of the GAU-8 ammunition was released on January 18, 1976." The report is said to have stated that the proposed action was expected to have no significant environmental impact and that the "biomedical and toxicological hazards of the use of depleted uranium in this program are practically negligible" (OSAGWI, 1998 Tab D). In 1978 the A-10 aircraft was deployed to United States Air Forces in Europe (Richard and Davitt, 1980).

OSAGWI, 1998 record that the US Navy's Phalanx Close-In Weapon System, or CIWS was designed for terminal (last-ditch) defence against sea-skimming missiles. They evaluated a wide range of materials before deciding on depleted uranium alloyed with 2 percent molybdenum (DU-2Mo). Phalanx production commenced in 1978, however, in 1988 the Navy opted to replace the depleted uranium with tungsten for the CIWS 20mm round (OSAGWI, 1998 Tab D; 2000).

As with the US, allied nations have also seen Naval use of depleted uranium diminish in recent decades. The Australian Defence Force used depleted uranium munitions in some Navy ships prior to 1986 as part of the Phalanx anti-missile system on these ships. After 1986, the RAN switched to a tungsten carbide based munition for these weapon systems.

Current Use of Depleted Uranium in Offensive and Defensive Weapons

OSAGWI (1998) records that depleted uranium was at that time used in:

- Kinetic cartridges for the US Army's 25mm BUSHMASTER cannon (M2/3 Bradley Fighting Vehicle), the 105mm cannon (M1 and M60 series tanks) and
- The 120mm cannon (M1A1 and M1A2 Abrams Tank)
- The Heavy Armour variant of the M1A1, the M1A1 (HA), also employs layered depleted uranium for increased armour protection.
- US Army Special Forces use small calibre depleted uranium ammunition on a limited basis. The
- Marines use depleted uranium tank rounds in their own M1-series tanks as well as a 25mm depleted uranium round in the GAU-12 Gatling gun on Marine AV-8 Harriers.
- The US Army uses small amounts of depleted uranium as an epoxy catalyst for two anti-personnel mines: the M86 Pursuit Deterrent Munition and the Area Denial Artillery Munition.
- The US Air Force uses a 30mm depleted uranium round in the GAU-8 Gatling gun on the A-10.
- The 20mm depleted uranium round developed by the US Navy for use in its shipboard PHALANX Close In Weapons System (CIWS) remains in service; however, since FY 1990, the Navy has procured only tungsten rounds for the CIWS.

Appendix Seven

(a) Locations of depleted uranium ordnance expended during Deny Flight-Deliberate Force, 1993-95 in Bosnia (grid co-ordinates)

NATO 24 Jan 2001: Data concerning the locations of depleted uranium ordnance expended during Allied Operations Deny Flight-Deliberate Force, 1993-95 in Bosnia (grid co-ordinates)

This list of grid locations do not represent the exact locations where DU munitions struck the ground but rather where NATO aircraft engaged targets. These co-ordinates are estimates based on pilots' mission reports and confirmed by commanders of units that expended the DU ordnance. Further information is being sought and will be provided when available.

A-10 Employment of 30mm Munitions Bosnia 1993-1995

#	Date	Target	Rounds	Location
1	5-Aug-94	76mm AT Self-Prop Gun	860	Vic Sarajevo
2	22-Sep-94	T-55 Tank	120	Vic Sarajevo
3	30-Aug-95	Warehouse	UNKWN	Vic Sarajevo
4	30-Aug-95	Artillery/Bunker	UNKWN	Vic Sarajevo
5	30-Aug-95	120mm artillery	UNKWN	Vic Sarajevo
6	30-Aug-95	AAA	UNKWN	Vic Sarajevo
7	30-Aug-95	Mortar Position	UNKWN	Vic Sarajevo
8	30-Aug-95	Mortar Position	UNKWN	Vic Sarajevo
9	5-Sep-95	Hadzici Mil Repair Facility	800	434932.7N 181122.9E
				440522.0N 185655.7E
				440525.1N 185653.7E
10	7-Sep-95	Han Pijesak Army Storage	700	440527.2N 185653.5E
				440539.6N 185649.7E
				440540.0N 185645.0E
				440522.0N 185655.7E
				440525.1N 185653.7E
11	7-Sep-95	Han Pijesak Army Storage	700	440527.2N 185653.5E
				440539.6N 185649.7E
				440540.0N 185645.0E
				440539.6N 185649.7E
12	7-Sep-95	Han Pijesak Army Storage	500	440540.0N 185645.0E
				440540.3N 185642.5E
				440539.6N 185649.7E
13	7-Sep-95	Han Pijesak Army Storage	500	440540.0N 185645.0E
				440540.3N 185642.5E
14	9-Sep-95	Hadzici Mil Repair Facility	350	434939.1N 181117.3E
15	9-Sep-95	Hadzici Mil Repair Facility	350	434939.1N 181117.3E
16	11-Sep-95	Hadzici Ammo Storage Depot	400	4348N 1812E
17	11-Sep-95	Hadzici Ammo Storage Depot	400	4348N 1812E
18	11-Sep-95	Hadzici Ammo Storage Depot	550	4348N 1812E
19	11-Sep-95	Hadzici Ammo Storage Depot	550	4348N 1812E

Appendix Seven

(b) Locations of depleted uranium ordnance expended during Operation Allied Force (grid co-ordinates)

This list of grid locations do not represent the exact locations where DU munitions struck the ground but rather where NATO aircraft engaged targets. These co-ordinates are estimates based on pilots' mission reports and confirmed by commanders of units that expended the DU ordnance. Further information is being sought and will be provided when available.

Kosovo			
Number	**Date**	**Location (UTM)**	**Total Number of Rounds**
1	06-Apr-99	34TDM717863	UNKWN
2	07-Apr-99	34TDM551901	110
3	08-Apr-99	34TDN665117	150
4	08-Apr-99	34TDN834190	UNKWN
5	15-Apr-99	34TEM580880	250
6	15-Apr-99	34TEM680995	UNKWN
7	16-Apr-99	34TEM643964	UNKWN
8	17-Apr-99	34TEM1885	200
9	27-Apr-99	34TDM433974	UNKWN
10	27-Apr-99	34TDM680690	UNKWN
11	30-Apr-99	420348N0203450E	UNKWN
12	30-Apr-99	34TEM208935	UNKWN
13	30-Apr-99	34TDN402102	UNKWN
14	5-May-99	34TDM515938	210
15	6-May-99	34TDM717863	UNKWN
16	7-May-99	34TDM503893	400
17	7-May-99	34TDN387039	500
18	7-May-99	34TDM771627	100
19	9-May-99	34TDN416092	200
20	10-May-99	34TEN148478	200
21	11-May-99	34TEN187470	700
22	11-May-99	34TEM019990	150
23	11-May-99	34TDM913976	65
24	12-May-99	34TDN505044	110
25	13-May-99	34TDN7735	570
26	14-May-99	34TDM723693	170
27	14-May-99	34TEM105920	UNKWN
28	14-May-99	34TDM525911	300
29	14-May-99	34TEM126888	90
30	15-May-99	34TDM7462	210
31	15-May-99	34TDN514102	320
32	15-May-99	34TEM1995	200
33	15-May-99	34TEM6496	130
34	15-May-99	34TDN719403	UNKWN
35	15-May-99	34TDM741622	UNKWN
36	16-May-99	34TDM745682	90
37	17-May-99	34TDM755619	170
38	17-May-99	34TEM540821	120
39	22-May-99	34TEM209103	UNKWN
40	25-May-99	34TDM624931	120
41	25-May-99	34TEM620945	300
42	25-May-99	34TEM632934	150
43	26-May-99	34TDM588998	UNKWN

44	26-May-99	34TDM5597	170
45	28-May-99	34TEN472112	100
46	28-May-99	34TEM625882	200
47	28-May-99	34TDM43159425	300
48	28-May-99	34TDM659950	50
49	28-May-99	34TEM189923	90
50	29-May-99	34TEN178432	350
51	29-May-99	34TDM695654	190
52	29-May-99	34TEM335844	UNKWN
53	29-May-99	34TDM580994	UNKWN
54	29-May-99	34TDM659950	50
55	29-May-99	34TCM01479634	230
56	29-May-99	34EM335844	80
57	30-May-99	34TEM1691	480
58	30-May-99	34TCM01479634	250
59	31-May-99	34TDM54938	200
60	31-May-99	34TDM6573	970
61	1-Jun-99	422550N 0202630E	200
62	1-Jun-99	34TDM663705	540
63	1-Jun-99	34TDM597858	400
64	1-Jun-99	34TDM782603	500
65	1-Jun-99	34TEM625882	970
66	2-Jun-99	34TDM728675	80
67	2-Jun-99	34TDM728675	70
68	2-Jun-99	34TDM5892	600
69	2-Jun-99	34TDM743720	400
70	2-Jun-99	34TDM503893	400
71	2-Jun-99	34TDN387039	500
72	2-Jun-99	34TDM771627	100
73	3-Jun-99	34TEN362171	150
74	3-Jun-99	34TDM503893	470
75	3-Jun-99	34TDM740590	370
76	3-Jun-99	34TDN59223216	700
77	5-Jun-99	34TDN393005	280
78	5-Jun-99	34TDN4002	120
79	5-Jun-99	34TDN389042	400
80	5-Jun-99	34TDN393005	200
81	5-Jun-99	34TDN387005	560
82	5-Jun-99	34TDN603245	320
83	5-Jun-99	34TDM67256935	286
84	6-Jun-99	34TDM409873	UNKWN
85	6-Jun-99	34TDM412883	907
86	6-Jun-99	34TDN4002	120
87	6-Jun-99	34TDM936785	970
88	6-Jun-99	34TDN474090	745
89	6-Jun-99	34TDM396948	100
90	6-Jun-99	34TDM396948	100
91	6-Jun-99	34TDN474090	200
92	6-Jun-99	34TDN464082	440
93	7-Jun-99	34TDM7439471956	140
94	7-Jun-99	34TDM545937	225
95	7-Jun-99	34TDN886168	370
96	7-Jun-99	34TDM592764	610
97	7-Jun-99	34TDN465083	530
98	7-Jun-99	34TDN534026	655
99	7-Jun-99	34TDN4310	560
100	8-Jun-99	34TDN528123	1320
101	8-Jun-99	34TDM771631-DM762600	400
102	8-Jun-99	34TDN863422	670
103	8-Jun-99	34TDN528123	1000

104	9-Jun-99	34TDM755645	200
105	UNKWN	34TDM772630	500
106	UNKWN	34TEM625882	970
107	17-Apr-99	34TEM170852	UNKWN
108	UNKWN	34TEM6308785128	UNKWN
109	UNKWN	34TEN17012908	UNKWN
110	UNKWN	34TDM5359283702	UNKWN
111	27-May-99	34TEM397979	UNKWN
112	28-May-99	34TEM631852	180

FRY (Serbia)			
Number	**Date**	**Location (UTM)**	**Total Number of Rounds**
5	15-Apr-99	34TEM580880	250
6	15-Apr-99	34TEM680995	UNKWN
7	16-Apr-99	34TEM643964	UNKWN
33	15-May-99	34TEM6496	130
38	17-May-99	34TEM540821	120
41	25-May-99	34TEM620945	300
42	25-May-99	34TEM632934	150
46	28-May-99		200
65	1-Jun-99	34TEM625882	970
106	11-Jun-99		970
108	UNKWN	34TEM6308785128	UNKWN
112	28-May-99	34TEM631852	180

FRY (Montenegro)			
Number	**Date**	**Location (UTM)**	**Total Number of Rounds**
55	29-May-99	34TCM01479634	230
58	30-May-99		250

112 strikes -- 96 different targets attacked:

85 Targets in Kosovo
10 Targets in FRY/Serbia (other than Kosovo)
1 Target in FRY/Montenegro

Targets hit more than once (16 targets):

1=15 7=33 16=70=74 17=71 18=72=101=105 30=35 46=65=106
48=54 52=56 55=58 59=94 77=80 78=86 88=91 89=90 100=103

Appendix Eight

Animal Studies

Summary information for animal studies for inhalation, ingestion and dermal exposure to a variety of uranium compounds and NOAELs and LOAELs for acute and chronic exposures to uranium compounds for ingestion and inhalation (ATSDR, 1999b).

Inhalation

Animal studies are available for acute, intermediate and chronic duration inhalation exposures to uranium for mortality, as well as renal, respiratory, haematological, cardiovascular, and gastrointestinal tract effects. No animal studies were located regarding the endocrine, metabolic, dermal, or ocular effects of uranium in animals following acute, intermediate or chronic-duration inhalation exposures to uranium (ATSDR, 1999b).

Mortality
Animal studies indicate that inhalation exposure to high concentrations of uranium can be fatal. Death resulting from renal failure caused by absorbed uranium. The acute-duration LC_{50} (concentration causing 50% death) for uranium hexafluoride (UF_6) gas for guinea pigs has been estimated at 35,011 mg U/m^3 for a 2 minute exposure limit, and for Long-Evans rats, 26,098 mg U/m^3 for a 5 minute exposure limit, and 8,114 mg U/m^3 for a 10 minute exposure limit (Leach et al, 1984). In other acute lethality studies, rats, mice and guinea pigs suffered 10, 20 and 13% mortality following a 10-minute inhalation of UF_6 gas corresponding to 637 mg U/m^3 (ATSDR, 1999b). In intermediate duration studies rabbits and cats were found to be the most sensitive species to uranium (Voegtlin and Hodge 1949, ATSDR, 1999b).

Insoluble uranium compounds were also lethal to animals by inhalation route but at higher concentrations than with soluble compounds (ATSDR, 1999b). In one inhalation study examining the effects of UO_2 in monkeys no increased mortality was observed at concentrations of 5 mg UO_2/m^3 for 5 years (Leach et al, 1970). However, Rothstein, 1949 observed increased mortality in a variety of animals exposed to 19 mg UO_3/m^3 for six hours a day, 5.5 days a week, for four weeks. "These exposures were cited as being 50 to more than 100,000 times higher than the current maximum permissible concentrations for inhalation exposure to soluble and insoluble uranium compounds, set by the American Conference of Governmental Industrial Hygienists" (ACGIH, 1993 in Harley et al, 1999).

Kidney
Inhaled uranium, generally of soluble compounds such as halides, has been shown to have low level metallotoxic effects on the renal system in short and intermediate duration animal studies using guinea pigs, mice, rats, cats, rabbits and dogs (ATSDR, 1999b). Insoluble uranium oxides appear far less toxic to the kidneys. Dogs, monkeys and rodents exposed to insoluble uranium dioxide (UO_2) at a concentration of about 5 mg U/m^3 for up to five years did not experience renal damage (Leach et al., 1970; Leach et al, 1973). Nevertheless other animal studies have observed nephrotoxic effects. Dygert (1949) observed evidence of renal damage as indicated by a moderate degree of regenerative tubular changes in some animals exposed to 17 mg triuranium octaoxide (U_3O_8)/m^3 for 26 days. Evidence for renal injury following exposure to uranium trioxide (UO_3) was observed in rabbits at a concentration of 19 mg UO_3/m^3 for four weeks, or a concentration of 22 mg UO_2/m^3 for 30 days (Rothstein, 1949).

Lung

Animal studies have demonstrated that the pulmonary toxicity of uranium is subject to considerable interspecies variability, as well as being dependent on the chemical form of the inhaled uranium (ATSDR, 1999b). Acute duration inhalation experiments have been limited to uranium hexafluoride gas (UF_6). Hydrofluoric acid is produced when UF_6 contacts water and is a potent respiratory irritant. Gasping and mucosal irritation was noted in rats and mice after 10 minute exposures at 637 mg U/m^3 (Spiegl 1949). Intermediate-duration exposure to uranium compounds have also caused pulmonary toxicity. Changes were most noticeable after UF_6 gas (Spiegl, 1949), but milder effects of rhinitis were observed with other uranium compounds including uranium tetrafluoride (Dygert, 1949) and uranyl fluoride (Rothstein 1949). Uranium dioxide and triuranium octaoxide did not cause toxicity ((Dygert, 1949; Rothstein 1949).

In chronic-duration experiments a total of 3,100 test animals including rats, rabbits, guinea pigs and dogs were exposed to aerosols containing 0.05 – 10 mg U/m^3 of various uranium compounds for 7-13 months; pulmonary histology revealed no signs of injury attributable to uranium (ATSDR, 1999b). Exposure of 200 rats, 110 dogs and 25 monkeys to 5.8 mgU/m^3 as UO_2 dust for 1-5 years for 5.4 hours/day, 5 days a week did not result in acute histological damage in the lungs of dogs or rats. Heavy deposits of particulate matter were found in the lungs and cumulative alpha radiation doses in the lungs of dogs and monkeys was greater than 500 rads (5 Gray). Minimal fibrosis, suggestive of radiation injury, was observed in the tracheobronchial lymph nodes of some dogs and monkeys and lungs of monkeys (http://www.atsdr.cdc.gov/Tox Profiles/phs9029.t12.gif accessed 21 May 2001; Leach et al., 1970; Leach et al., 1973).

Cancer

Chronic inhalation studies on beagle dogs (32 exposed, 13 with five year exposure and six control) and monkeys (eight exposed, six with five year exposure and five control) maintained for up to 6.5 years after exposures of 1-5 years to UO_2 aerosol at a concentration of 5.8 mg/m^3 and approximately one micrometer median particle diameter were undertaken by Leach et al, 1973. The magnitude of the uranium concentration at the end of the five year exposure was approximately 2,000 µg U/gm (dogs), and 4,000 µg U/gm (monkeys) in the lungs. Leach estimated the lung burden in dogs to be 151 mg of uranium at the end of the exposure period. The estimated radiation dose from the experiment (alpha radiation accumulated over the five year exposure plus 2 - 6.5 years post exposure) to the dog lungs was 600-700 rads (6-7 Gray). This study conveyed a considerable loading of particulate matter and internal radiation to the lungs of the exposed animals. Pulmonary neoplasms (2 adenoma and 2 adenocarcinoma) and atypical epithelial proliferation were noted in nine of 13 dogs, while only one of five controls had bronchiolar proliferation. This finding in dogs has not been replicated in the same or other species of animals and may represent an idiosyncratic effect in dogs. Extensive lung fibrosis but no pulmonary neoplasms were seen in seven maximally exposed monkeys (Leach et al, 1973).

The magnitude of the uranium concentration in the tracheobronchial lymph nodes at the end of the five year exposure was approximately 60,000 µg U/gm (dogs), and 70,000 µg U/gm (monkeys). Most dogs and all monkeys in the maximally exposed group demonstrated fibrosis of the tracheobronchial lymph nodes. Monkeys displayed replacement of lymphoid tissue with pigment and fibrosis. Lymphoma was noted in one of seven exposed monkeys while no abnormalities were noted in the tracheobronchial nodes of five control monkeys. No lymphoma was noted in any of the exposed dogs. The estimated cumulative radiation dose to the nodes of maximally

exposed monkeys was at least 7000 rads (70 Gray) (Leach et al, 1970). This is a very large dose of internal radiation.

Golden Syrian hamsters (four study groups each with 102 animals) inhaling 19 mg U/m^3 as uranium ore dust for 16 months showed no apparent increase in tumours of the liver, kidney, spleen, trachea, lungs or heart compared to unexposed controls (Cross et al 1981a). In the hamsters, uranium ore dust provoked inflammatory and proliferative responses and alveolar hyperplasia but these changes did not progress with time.

Neurological Disturbance

High dose animal experimentation has demonstrated that inhalation of high concentrations of UF_6 gas by cats and dogs causes muscle weakness and gait disturbance (Dygert 1949). This provides limited evidence that large doses of at least some uranium compounds, many orders of magnitude above that experienced by humans, may cause central nervous system alterations in animals. No studies of inhalation of uranium oxides and musculoskeletal disorders were found.

Gastrointestinal and Liver

Dogs but not other species appear susceptible to gastrointestinal effects after inhalation of high concentrations of uranium compounds (ATSDR, 1999b). Little hepatic toxicity is reported in animal inhalation experiments; Rothstein, 1949 reported moderately fatty livers in some of the rabbits and rats tested after exposure to concentrations of 19 mg UO_3/m^3 for four weeks, while other species of animals examined in the study exhibited no such effect. In another animal study, animals exposed to 22 mg UO_2/m^3 for 30 days showed no hepatic effects (Rothstein, 1949). Dygert (1949) found no hepatic effect in animals exposed to concentrations of up to 17 mg U_3O_8/m^3 for 26 days. Likewise, no hepatic effects were observed in animals exposed to 10 mg UO_2/m^3 for one to two years (Stokinger, 1953).

Haematological

Rats exposed to a weighted mean of 19 mg UO_3/m^3 for four weeks (with short periods approaching 30 to 40 mg UO_3/m^3) exhibited significant differences in myeloid and lymphoid cells of the bone marrow but found no significant haematological change (Rothstein, 1949). Several other animal inhalation studies also found no haematological effects. Exposures as high as 22 mg UO_2/m^3 for 30 days or 17 mg U_3O_8/m^3 for up to 40 days in a variety of animal models found no adverse effects on the blood (Dygert, 1949; Rothstein, 1949; Leach et al, 1970; Leach et al, 1973).

Other

Studies examining immune system effects have found animals exposed to UO_2 dusts at a concentration of 5.0-5.8 mg U/m^3 for one to five years did not exhibit any pathologic changes in their spleens (Leach, 1970, 1973). No significant adverse effects on body weight were observed in several animal inhalation studies (Dygert, 1949, Rothstein, 1949; Leach, 1970, 1973). One study was identified which considered possible skin and endocrine effects and Stokinger et al, 1953 found no histopathological evidence of endocrine or dermal effects in rats exposed to 0.2 mg U/m^3 as uranium tetrachloride for one year.

No information on cardiovascular, reproductive, developmental, or genotoxic toxicological effects of inhaled uranium compounds was identified in a search of the primary literature or recent reviews (Harley et al. 1999; ATSDR, 1999b; Fulco et al, 2000).

Oral Consumption

Oral administration of a range of uranium compounds has been undertaken in a variety of animal models spanning mice, rats, dogs, rabbits, cattle and sheep. The oral toxicity of uranium compounds is well described in several reviews (Voegtlin and Hodge, 1949, 1953; ATSDR, 1999b). The chemical toxicity of ingested uranium is determined largely by the water solubility of the compound and therefore, ease of uptake from the gastrointestinal tract. In a summary of the oral toxicity in both rats and dogs the relative toxicity of uranium compounds was considered as follows: very toxic compounds included uranium tetrachloride, uranium peroxide, and uranyl fluoride; toxic compounds included uranium nitrate hexahydrate, uranyl acetate, ammonium diuranate, sodium diuranate, uranium trioxide, and high grade uranium ore (carnitite); practically non toxic compounds were uranium tetrafluoride, triuranium octaoxide, and uranium dioxide (Maynard and Hodge, 1949). For various end points and animal species ATSDR has reported minimal effect levels in the range of 1-10 mg U/kg of body weight per day (ATSDR, 1999b).

Mortality
Animal studies show that very high oral intake levels of soluble uranium compounds can be lethal. The oral LD_{50} (lethal dose, 50% mortality rate) for male Sprague Dawley rats has been estimated to by 114 mg U/kg following single gavage administration of uranyl acetate dihydrate (Domingo et al, 1987). No mortality was observed in mice dosed at 0.02 to 20 mg UO_2/day/mouse for a year and no evidence of toxicity appeared in mice dosed 100 mg U_3O_8/day/mouse for a year (ATSDR, 1999b).

Kidney
The kidney may be affected after systemic absorption of uranium after ingestion or inhalation, with proximal tubular epithelium the principal site of chemical toxicity however glomeruli and interstitial tissues have also been found to be damaged in some acute and intermediate duration experiments. Independent of route of absorption, uranium has a low order metallotoxicity in mammals (ATSDR, 1999b).

Lung
Respiratory effects from oral exposure to uranium are unlikely. In acute, intermediate and chronic duration experiments on rats, rabbits and dogs no respiratory effects were reported following ingestion of uranium oxides or other uranium compounds (Maynard and Hodge, 1949; Maynard et al, 1953; Domingo et al, 1987; Gilman et al, 1998a; Gilman et al, 1998b; Gilman et al, 1998c; ATSDR, 1999b). Cardiovascular effects from oral exposure to uranium are also unlikely. In acute and intermediate duration experiments on rats and rabbits cardiovascular effects have not been observed in animals following ingestion of uranium oxides or other uranium compounds (Gilman et al, 1998a; Gilman et al, 1998b; Gilman et al, 1998c; ATSDR, 1999b).

Neurological Disturbance
No histopathological changes in nervous tissue have been reported after oral exposure to uranium in studies on rats and rabbits (Gilman et al, 1998a; Gilman et al, 1998b; Gilman et al, 1998c). One study, by Domingo et al, 1987 have reported clinical neurotoxicity (piloerection, exophthalmos and decreased pupillary size) in Sprague-Dawley rats after acute exposure to doses of between 11-717mg U/kg as uranyl acetate dihydrate.

Reproductive
Limited animal studies have demonstrated some effect on reproductive function but generally not of histopathological damage to reproductive tissues. Intermediate studies of oral ingestion of uranium as uranyl nitrate in rats and rabbits have shown no changes in reproductive effects or reproductive organ weights (Gilman et al, 1998a; Gilman et al, 1998b; Gilman et al, 1998c).). Llobet et al (1991) found male mice treated with high doses of uranyl acetate dihydrate had normal testicular size and spermatogenesis, but a non dose related decrease in pregnancy rate after mating. Chronic duration studies on rats have shown testicular degeneration after two years of high oral doses of uranyl nitrate hexahydrate (Maynard et al, 1953).

The teratogenic potential of ingested uranium has been considered using mice. Dose related developmental effects, such as reduced foetal body weight and length, and external malformations (cleft palate and skeletal malformations) have been observed in one study with high dose oral uranyl acetate dihydrate administered during gestation (Domingo et al., 1989). A no-effect-level of below 5 mg uranyl acetate dihydrate per kg per day during pregnancy was suggested. In the study maternal toxicity was evident in all 20 pregnant mice. No studies were identified which examined developmental effects associated with uranium oxide ingestion. No studies were located that reported genotoxic effects in animals following oral exposure to uranium for any duration.

Other
Intermediate and chronic studies of oral uranium exposure in rats, mice and rabbits have not identified immune system effects attributable to uranium oxide exposure (Maynard et al., 1953; Gilman et al, 1998a; Gilman et al, 1998b; Gilman et al, 1998c;). No gastrointestinal effects occurred in animal experiments of intermediate and chronic oral exposure to a range of uranium compounds (Maynard and Hodge, 1949; Gilman et al, 1998a; Gilman et al, 1998b; Gilman, 1998c). Although hepatic effects have been observed in animals dosed with very high levels of insoluble uranium (ATSDR, 1999b), dogs dosed with up to 10 g UO_2/kg/day for one year exhibited no hepatic effects (Maynard and Hodge, 1949). No harmful effects on body weight were seen in intermediate-duration oral studies of dogs given up to 10 g UO_2/kg/day for one year (Maynard and Hodge, 1949; Dygert et al, 1949; Maynard et al, 1953). No animal data were available regarding musculoskeletal, metabolic, dermal or ocular effects following oral exposure to uranium (ATSDR, 1999b).

Skin Contact

Animal studies in rabbits, rats, mice and guinea pigs have demonstrated that soluble uranium compounds, when applied to bare skin, may be absorbed through the skin in sufficient amounts to cause renal toxicity (Orcutt, 1949a; DeRey et al , 1983; DeRey et al, 1984). Rabbits have proved the most sensitive animal model with LD_{50} (due to renal failure) of 28mg U/kg as uranyl nitrate in ethereal solution (Orcutt, 1949a). Water insoluble compounds (uranium tetrafluoride, uranium dioxide, uranium peroxide, triuranium octaoxide) when applied to the skin have not been associated with weight loss or deaths (Orcutt, 1949a).

Several studies have demonstrated the soluble uranium compounds when applied to the skin can cause irritation at the site of the application (Orcutt, 1949b; De Rey et al, 1983). No such effects were seen in similar rabbit experiments following an application of 618mg U/kg as uranyl fluoride, 666mg U/kg as uranium trioxide, 195 mg U/kg as sodium diuranate, 198 mg U/kg as

ammonium diuranate, 410 U/kg uranium peroxide, 458 mg U/kg as uranium dioxide or 147 mg U/kg as triuranium octaoxide in 50% aqueous solution (Orcutt, 1949a).

No studies regarding respiratory, cardiovascular, gastrointestinal, haematological, immunological, musculoskeletal, hepatic, endocrine, reproductive, developmental, genotoxic or cancer effects of uranium following acute, intermediate or chronic duration skin exposure to uranium were identified (ATSDR, 1999b).

Figure A8.1 Acute oral exposure to uranium compounds (ATSDR, 1999b)

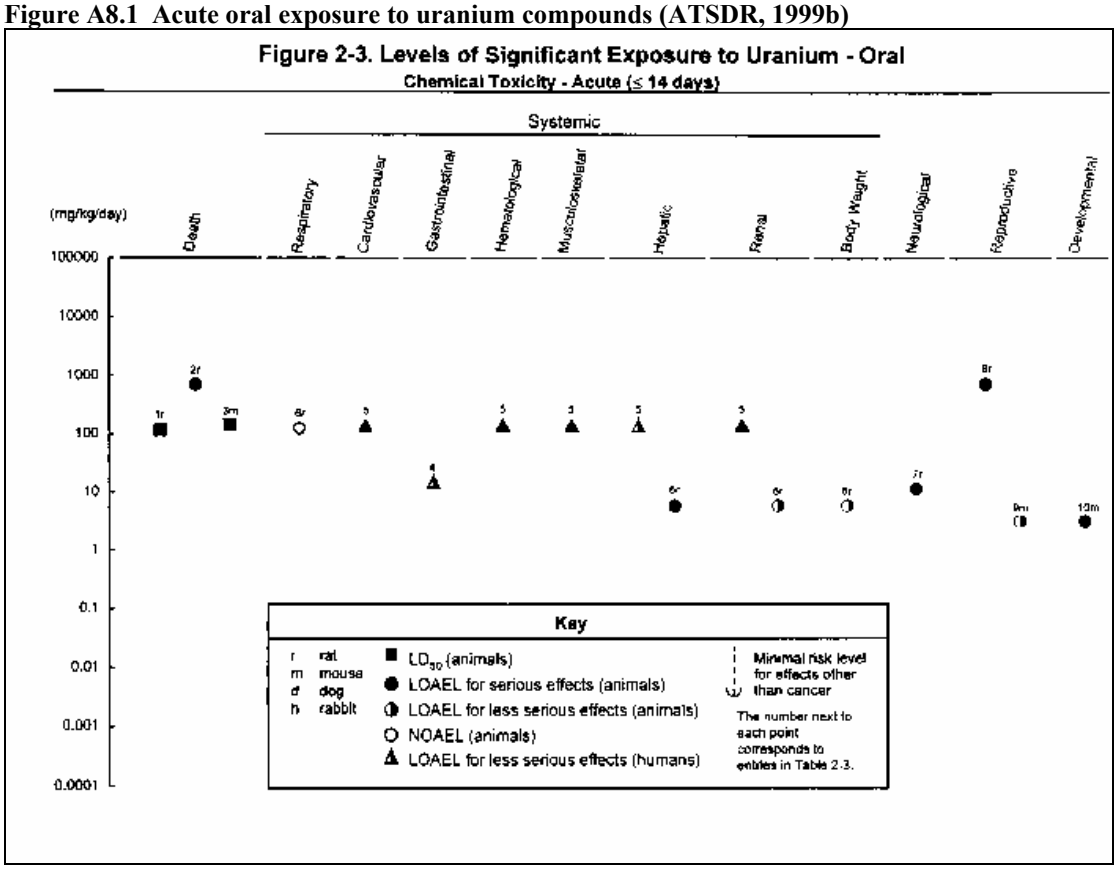


Figure A8.2 Chronic oral exposure to uranium compounds (ATSDR, 1999b)

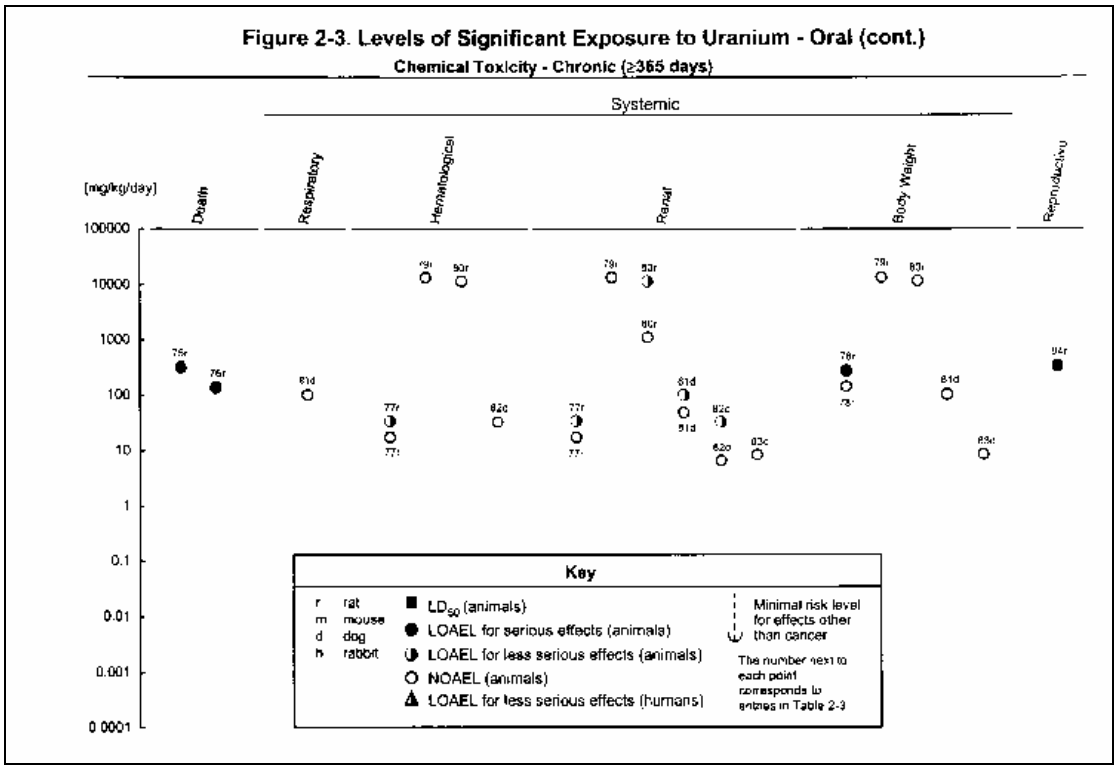


Figure A8.3 Acute inhalation exposure to uranium compounds (ATSDR, 1999b)

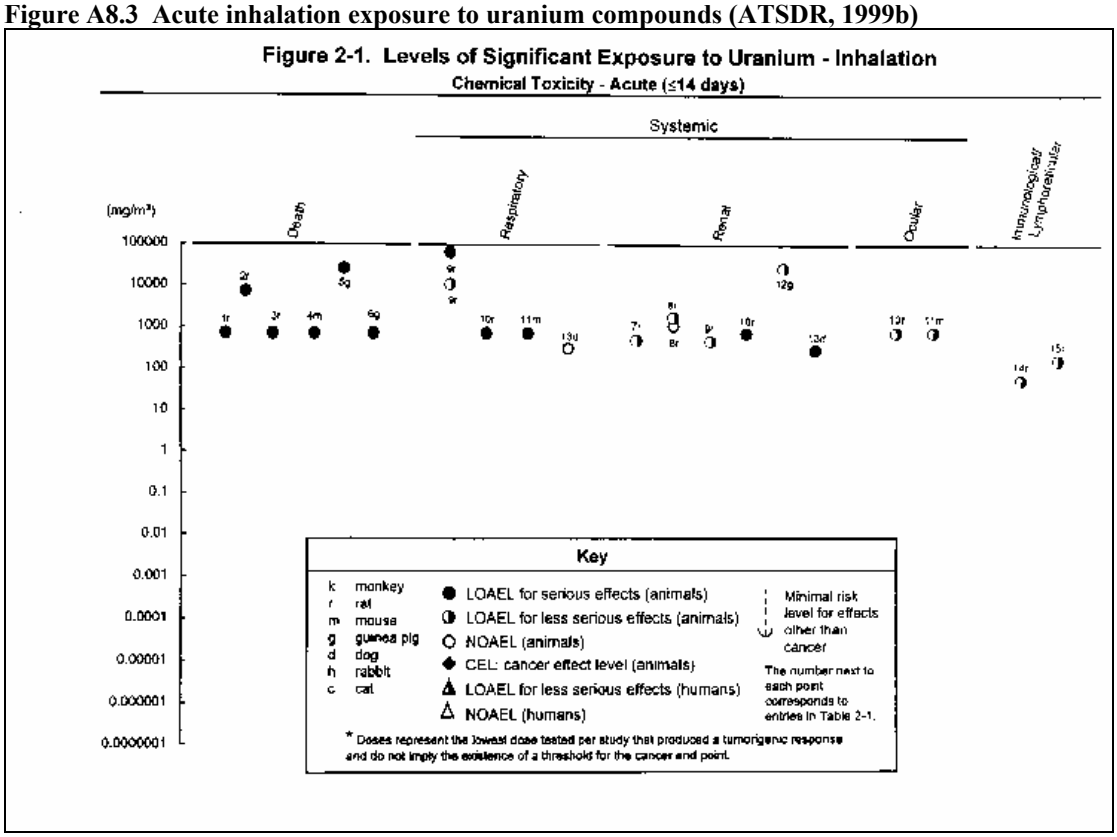


Figure A8.3 Chronic inhalation exposure to uranium compounds (ATSDR, 1999b)

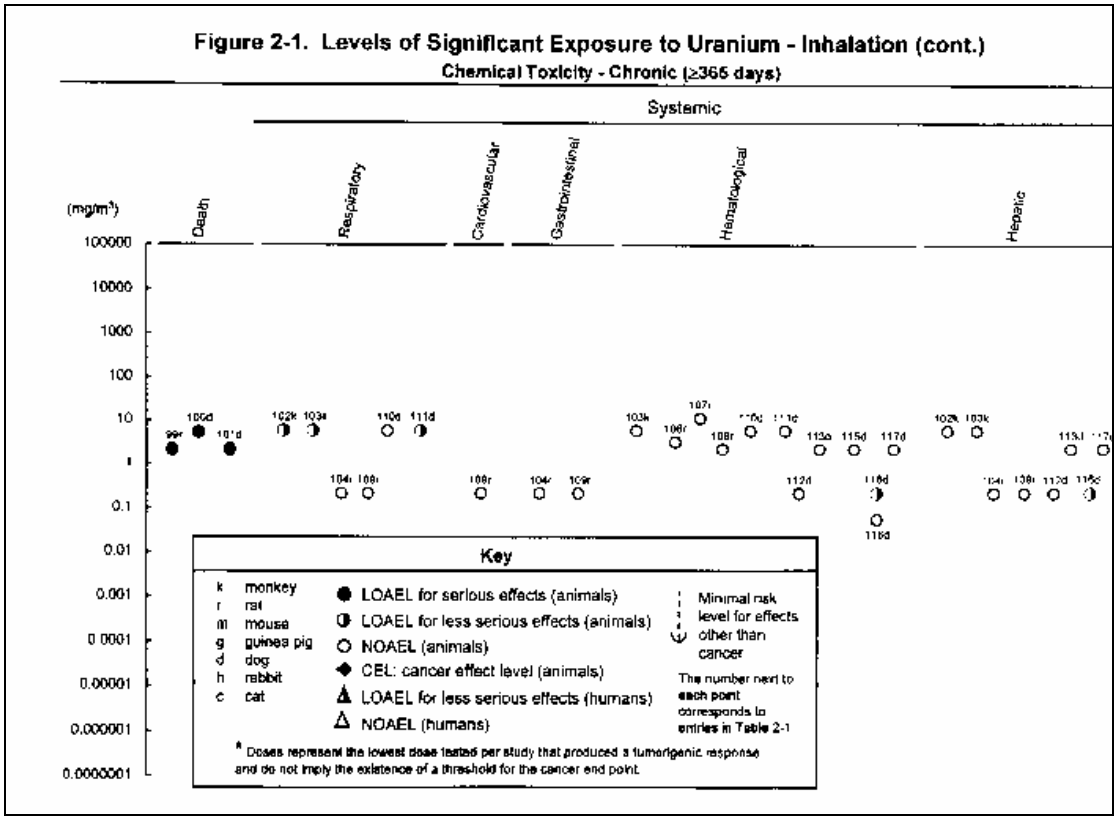
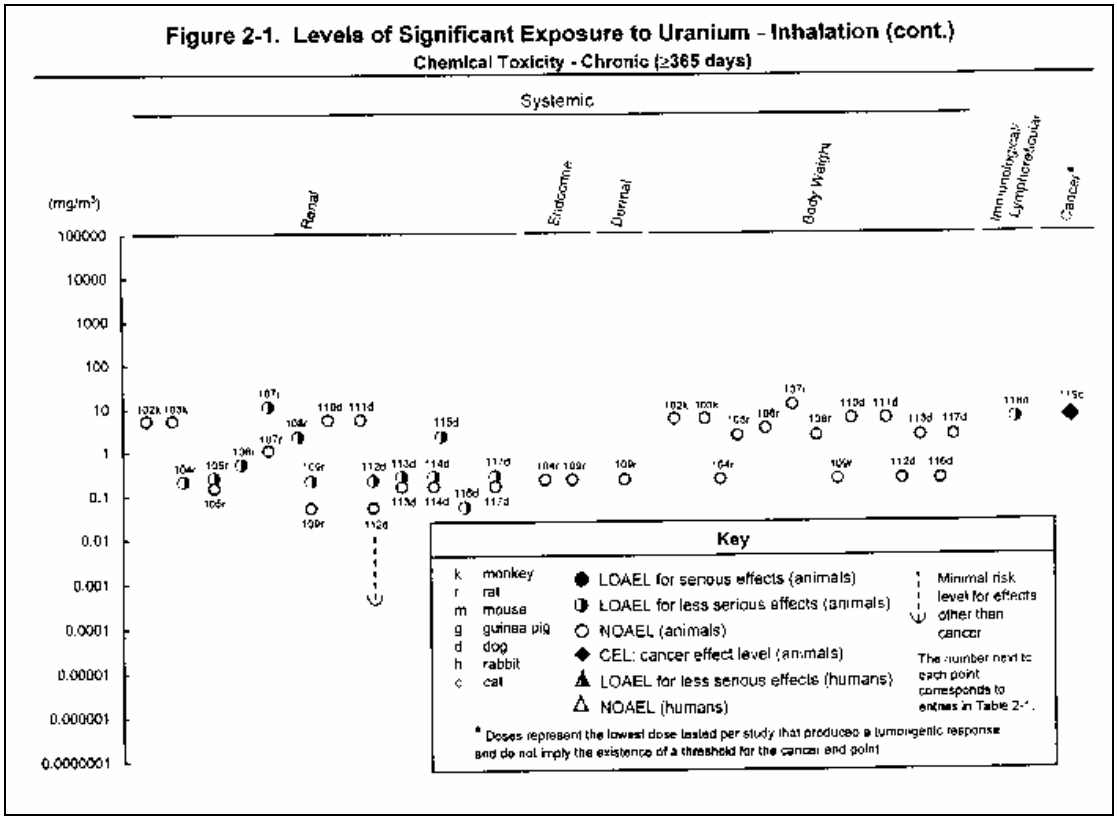


Figure A8.5 Chronic inhalation exposure to uranium compounds (cont'd) (ATSDR, 1999b)



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Appendix Nine

Epidemiological Studies of Uranium Exposed Workers
(Beral and Darby, 2001)

Summary Mortality Tables for Cohort Studies of Uranium Process Workers
(Beral and Darby, 2001)

Table A9.1
Epidemiological Studies of Uranium Exposed Workers (Beral and Darby, 2001)

Reference	Plant and location	Population studied	Exposure Data	Total workers in results	Total No. of deaths	Earlier refs with same/ overlapping population
McGeoghegan and Binks, 2000a	Springfields Lancashire	Radiation workers at plant fabricating uranium (U) fuel and UF_6	External monitored radiation dose	13,960	3,476	None relevant
Dupree-Ellis et al, 2000	Mallinckrodt, Missouri	Workers at a U processing plant	External monitored radiation dose	2,514	1,013	Dupree et al, 1995
Ritz, 2000	Rocketdyne/ Atomics Int, California	Workers involved in nuclear fuel assembly and disassembly	Internal radiation dose to lung	2,297	433	None relevant
McGeoghegan and Binks, 2000a	Capenhurst, Cheshire	Radiation workers at U enrichment plant	External monitored radiation dose	3,244	585	None relevant
Ritz 1999a	Fernald Feed, Ohio	U processing workers	External monitored radiation dose and Internal radiation dose to lung	4,014	1,064	Dupree et al, 1995; Ritz 1999b
Frome et al, 1997	4 Federal nuclear Oak Ridge, Tennessee,	Workers involved in U enrichment or nuclear fuel fabrication	External monitored radiation dose and internal monitoring status	~67,600	22,724	Dupree et al, 1995; Loomis et al, 1996; Wing et al 1991; Frome et al 1990; Checkoway et al, 1988; Checkoway et al, 1985; Poledak et al, 1981; Cookfair et al, 1983
Teta and Ott, 1988	Linde, NY	Workers at a site previously contaminated with U and other radionuclides	Not available	8,146	1,160	None relevant
Cragle et al, 1988	Savannah River, South Caroline	Workers at a nuclear fuel production plant	Not available	9,860	1,091	None relevant

Table A9.1 (continued)
Epidemiological Studies of Uranium Exposed Workers (Beral and Darby, 2001)

Reference	Plant and location	Population studied	Exposure Data	Total workers in results	Total No. of deaths	Earlier refs with same/ overlapping population
Beral et al, 1988	Atomic Weapons Est, Berkshire UK	Workers at a weapons production plant	External radiation dose and uranium monitoring status	3,044	137	None relevant
Dupree et al, 1987	Linde, NY	Workers at a U processing facility monitored for U exposure	Estimated internal lung dose	995	429	None relevant
Brown and Bloom, 1987	Portsmouth, Ohio	Workers at a U enrichment facility	Not available	5,244	483	None relevant
Stayner et al, 1985	Polk, Florida	Workers at a phosphate fertilizer production facility	Not available	3,160	155	None relevant
Waxweiler et al, 1983	Seven mills Rocky Mt region	Uranium mills and mines	Not available	2,002	533	Archer et al, 1973; Wagoner et al 1964
Hadjimichael et al, 1983	United Nuclear, Connecticut	Workers at a nuclear fuels fabrication plant	Not available	3,512	219	None relevant

Summary mortality tables for cohort studies of uranium process workers

From Valerie Beral and Sarah Darby, 2001 (Appendix Three, Royal Society Report, 2001)

Table A9.2. Ratio of observed number of deaths from all causes in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)*
McGeoghegan and Binks, 2000a	3476	0.84 (0.81-0.87)
Dupree-Ellis et al, 2000	1013	0.90 (0.85-0.96)
Ritz et al, 2000	433	0.72 (0.66-0.80)
McGeoghegan and Binks, 2000b	585	0.83 (0.76-0.90)
Ritz 1999a	1064	0.84 (0.79-0.89)
Frome et al, 1997	22724	1.00 (0.99-1.01)
Teta and Ott, 1988	1160	0.87 (0.82-0.92)
Cragle et al, 1988	1091	0.75 (0.71-0.80)
Beral et al, 1988	137	0.88 (0.74-1.04)
Dupree et al, 1987	429	1.18 (1.07-1.30)
Brown and Bloom, 1987	483	0.68 (0.62-0.74)
Stayner et al, 1985	155	0.82 (0.69-0.96)
Waxweiler et al, 1983	533	0.88 (0.81-0.96)
Hadjimichael et al, 1983	219	0.82 (0.71-0.93)
Summary value	33502	0.86 (0.79-0.93)
Test for heterogeneity: $\chi^2_{13} = 414.05$; $P<0.001$		

*Estimated ratio of observed number of deaths (O) to that expected in the general population (E), with 95% confidence intervals (CI)

Table A9.3. Ratio of observed number of deaths from all cancers in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	971	0.86 (0.81-0.91)
Dupree-Ellis et al, 2000	283	1.05 (0.93-1.17)
Ritz et al, 2000	133	0.87 (0.73-1.03)
McGeoghegan and Binks, 2000b	178	0.88 (0.76-1.02)
Ritz 1999a	332	1.09 (0.98-1.22)
Frome et al, 1997	4679	0.98 (0.95-1.01)
Teta and Ott, 1988	259	0.99 (0.87-1.12)
Cragle et al, 1988	216	0.74 (0.65-0.85)
Beral et al, 1988	37	0.88 (0.62-1.21)
Dupree et al, 1987	74	1.06 (0.83-1.32)
Brown and Bloom, 1987	125	0.85 (0.71-1.02)
Stayner et al, 1985	22	0.76 (0.48-1.15)
Waxweiler et al, 1983	82	0.75 (0.59-0.93)
Hadjimichael et al, 1983	51	0.85 (0.65-1.15)
Summary value	7442	0.91 (0.85-0.97)
Test for heterogeneity: $\chi^2_{13} = 52.26$; P<0.001		

Table A9.4. Ratio of observed number of deaths from stomach cancer in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	92	0.92 (0.74-1.13)
Dupree-Ellis et al, 2000	4	0.38 (0.12-0.89)
Ritz et al, 2000	6	1.18 (0.43-2.57)
McGeoghegan and Binks, 2000b	15	0.90 (0.51-1.49)
Ritz 1999a	15	1.34 (0.75-2.21)
Frome et al, 1997	176	0.73 (0.63-0.85)
Teta and Ott, 1988	11	0.83 (0.42-1.49)
Cragle et al, 1988	7	0.58 (0.23-1.19)
Beral et al, 1988	2	0.50 (0.06-1.81)
Dupree et al, 1987	7	1.65 (0.66-3.39)
Brown and Bloom, 1987	10	1.69 (0.81-3.10)
Stayner et al, 1985	3	0.41 (0.09-1.21)
Waxweiler et al, 1983	3	0.40 (0.08-1.17)
Hadjimichael et al, 1983	14	0.82 (0.45-1.38)
Summary value	365	0.76 (0.62-0.89)
Test for heterogeneity: $\chi^2_{13} = 18.29$; P>0.1; NS		

Table A9.5. Ratio of observed number of deaths from colorectal cancer in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	118	0.96 (0.80-1.15)
Dupree-Ellis et al, 2000	36	1.18 (0.83-1.64)
Ritz et al, 2000	15	1.11 (0.62-1.83)
McGeoghegan and Binks, 2000b	19	0.87 (0.53-1.36)
Ritz 1999a	35	1.05 (0.73-1.46)
Frome et al, 1997	425	0.74 (0.68-0.82)
Teta and Ott, 1988	37	1.19 (0.84-1.64)
Cragle et al, 1988	21	0.68 (0.42-1.04)
Beral et al, 1988	6	1.34 (0.49-2.91)
Dupree et al, 1987	6	0.95 (0.35-2.06)
Brown and Bloom, 1987	10	0.82 (0.39-1.51)
Summary value	728	0.91 (0.78-1.04)
Test for heterogeneity: $\chi^2_{10} = 17.45$; $0.05<P<0.10$		

Table A9.6. Ratio of observed number of deaths from liver cancer in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	22	1.18 (0.74-1.78)
Dupree-Ellis et al, 2000	2	0.42 (0.07-1.30)
McGeoghegan and Binks, 2000b	2	0.60 (0.07-2.16)
Ritz 1999a	8	1.62 (0.70-3.20)
Frome et al, 1997	78	0.78 (0.62-0.97)
Teta and Ott, 1988	7	1.71 (0.69-3.52)
Cragle et al, 1988	4	0.85 (0.23-2.18)
Summary value	123	0.84 (0.62-1.07)
Test for heterogeneity: $\chi^2_6 = 6.93$; $P>0.1$; NS		

Table A9.7. Ratio of observed number of deaths from lung cancer in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	360	0.85 (0.77-0.95)
Dupree-Ellis et al, 2000	98	1.02 (0.83-1.24)
Ritz et al, 2000	46	0.81 (0.59-1.08)
McGeoghegan and Binks, 2000b	67	0.90 (0.69-1.14)
Ritz 1999a	112	1.01 (0.83-1.21)
Frome et al, 1997	1849	1.18 (1.13-1.24)
Teta and Ott, 1988	97	1.07 (0.87-1.31)
Cragle et al, 1988	83	0.83 (0.66-1.03)
Beral et al, 1988	11	0.65 (0.32-1.16)
Dupree et al, 1987	21	0.97 (0.60-1.48)
Brown and Bloom, 1987	48	0.88 (0.65-1.17)
Stayner et al, 1985	10	1.13 (0.54-2.08)
Waxweiler et al, 1983	26	0.83 (0.54-1.21)
Hadjimichael et al, 1983	18	0.92 (0.54-1.45)
Summary value	2846	0.94 (0.83-1.05)
Test for heterogeneity: $\chi^2_{13} = 58.70$; P<0.001		

Table A9.8. Ratio of observed number of deaths from bone cancer in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	2	0.67 (0.08-2.42)
Dupree-Ellis et al, 2000	1	1.20 (0.07-5.26)
McGeoghegan and Binks, 2000b	0	0.00 (0.00-8.02)
Ritz 1999a	0	0.00 (0.00-3.70)
Frome et al, 1997	25	1.19 (0.77-1.76)
Teta and Ott, 1988	0	0.00 (0.00-2.63)
Cragle et al, 1988	1	0.63 (0.02-8.65)
Hadjimichael et al, 1983	1	1.55 (0.02-8.65)
Summary value	30	0.93 (0.53-1.33)
Test for heterogeneity: $\chi^2_{7} = 4.58$; P>0.1; NS		

Table A9.9. Ratio of observed number of deaths from prostate cancer in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	68	0.89 (0.69-1.12)
Dupree-Ellis et al, 2000	23	1.15 (0.74-1.70)
Ritz et al, 2000	7	0.73 (0.29-1.50)
McGeoghegan and Binks, 2000b	11	0.79 (0.39-1.41)
Ritz 1999a	25	1.44 (0.93-2.12)
Frome et al, 1997	319	1.01 (0.90-1.13)
Teta and Ott, 1988	15	0.93 (0.52-1.53)
Cragle et al, 1988	8	0.87 (0.38-1.72)
Beral et al, 1988	6	2.81 (1.03-6.10)
Stayner et al, 1985	2	1.83 (0.22-6.63)
Waxweiler et al, 1983	6	0.71 (0.26-1.54)
Summary value	490	0.98 (0.89-1.07)
Test for heterogeneity: $\chi^2_{10} = 8.00$; P>0.1; NS		

Table A9.10. Ratio of observed number of deaths from bladder cancer in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	40	0.92 (0.65-1.25)
Dupree-Ellis et al, 2000	8	1.16 ((0.48-2.36)
Ritz et al, 2000	3	0.89 (0.18-2.59)
McGeoghegan and Binks, 2000b	8	1.04 (0.45-2.05)
Ritz 1999a	8	1.15 (0.50-2.27)
Frome et al, 1997	105	0.76 (0.62-0.92)
Teta and Ott, 1988	15	1.09 (0.61-1.79)
Cragle et al, 1988	6	1.04 (0.38-2.26)
Beral et al, 1988	1	0.61 (0.02-3.40)
Stayner et al, 1985	2	1.82 (0.22-6.57)
Summary value	196	0.83 (0.71-0.96)
Test for heterogeneity: $\chi^2_9 = 3.81$; P>0.1; NS		

Table A9.11. Ratio of observed number of deaths from kidney cancer in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	13	0.60 (0.32-1.03)
Dupree-Ellis et al, 2000	8	1.17 (0.54-2.18)
Ritz et al, 2000	5	1.26 (0.41-2.94)
McGeoghegan and Binks, 2000b	2	0.49 (0.06-1.77)
Ritz 1999a	5	0.63 (0.20-1.46)
Frome et al, 1997	109	0.92 (0.76-1.11)
Cragle et al, 1988	3	0.38 (0.08-1.10)
Beral et al, 1988	3	4.30 (0.90-12.70)
Waxweiler et al, 1983	3	1.12 (0.23-3.25)
Summary value	151	0.78 (0.59-0.96)
Test for heterogeneity: $\chi^2_8 = 9.16$; P>0.1; NS		

Table A9.12. Ratio of observed number of deaths from brain cancer in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	18	0.67 (0.39-1.05)
Dupree-Ellis et al, 2000	12	1.57 (0.84-2.64)
Ritz et al, 2000	6	1.31 (0.48-2.84)
McGeoghegan and Binks, 2000b	7	1.39 (0.56-2.86)
Ritz 1999a	12	1.24 (0.64-2.17)
Frome et al, 1997	151	1.09 (0.92-1.28)
Teta and Ott, 1988	4	0.43 (0.12-1.09)
Cragle et al, 1988	7	0.55 (0.22-1.13)
Beral et al, 1988	1	0.85 (0.02-4.76)
Hadjimichael et al, 1983	5	2.23 (0.72-5.20)
Summary value	223	0.91 (0.65-1.17)
Test for heterogeneity: $\chi^2_9 = 17.02$; P<0.05		

Table A9.13. Ratio of observed number of deaths from thyroid cancer in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	4	2.15 (0.59-5.51)
McGeoghegan and Binks, 2000b	0	0.00 (0.00-11.52)
Ritz 1999a	0	0.00 (0.00-6.76)
Frome et al, 1997	3	0.33 (0.07-0.96)
Summary value	7	0.38 (0.00-0.81)
Test for heterogeneity: $\chi^2_3 = 2.10$; P>0.1; NS		

Table A9.14. Ratio of observed number of deaths from non-Hodgkin lymphoma in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	15	0.63 (0.35-1.04)
Dupree-Ellis et al, 2000	10	0.77 (0.37-1.42)
Ritz et al, 2000	4	0.44 (0.12-1.14)
McGeoghegan and Binks, 2000b	5	1.09 (0.35-2.55)
Ritz 1999a	8	1.67 (0.72-3.29)
Frome et al, 1997	187	0.87 (0.75-1.00)
Teta and Ott, 1988	8	0.94 (0.41-1.85)
Cragle et al, 1988	14	0.65 (0.36-1.10)
Beral et al, 1988	1	1.44 (0.04-8.07)
Brown and Bloom, 1987	8	1.57 (0.68-3.11)
Waxweiler et al, 1983	4	0.91 (0.25-2.33)
Hadjimichael et al, 1983	2	0.48 (0.05-1.73)
Summary value	266	0.82 (0.71-0.92)
Test for heterogeneity: $\chi^2_{11} = 8.99$; P>0.1; NS		

Table A9.15. Ratio of observed number of deaths from Hodgkin's disease in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	9	1.24 (0.57-2.36)
Dupree-Ellis et al, 2000	2	0.92 (0.15-2.83)
McGeoghegan and Binks, 2000b	2	1.77 (0.21-6.39)
Ritz 1999a	6	2.04 (0.74-4.43)
Frome et al, 1997	40	0.77 (0.55-1.05)
Teta and Ott, 1988	2	0.51 (0.06-1.85)
Brown and Bloom, 1987	4	1.54 (0.43-4.01)
Waxweiler et al, 1983	3	2.31 (0.48-6.75)
Summary value	68	0.83 (0.61-1.06)
Test for heterogeneity: $\chi^2_7 = 5.00$; P>0.1; NS		

Table A9.16. Ratio of observed number of deaths from leukaemia in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	32	1.00 (0.68-1.41)
Dupree-Ellis et al, 2000	11	1.11 (0.57-1.89)
Ritz et al, 2000	8	1.46 (0.63-2.88)
McGeoghegan and Binks, 2000b	4	0.69 (0.19-1.78)
Ritz 1999a	13	1.16 (0.62-1.98)
Frome et al, 1997	180	0.98 (0.84-1.13)
Teta and Ott, 1988	11	1.00 (0.50-1.79)
Cragle et al, 1988	18	1.46 (0.86-2.30)
Dupree et al, 1987	6	0.89 (0.32-1.93)
Brown and Bloom, 1987	8	1.29 (0.56-2.56)
Stayner et al, 1985	2	0.53 (0.06-1.91)
Waxweiler et al, 1983	0	0.00 (0.00-0.82)
Hadjimichael et al, 1983	2	0.84 (0.09-3.03)
Summary value	295	0.90 (0.67-1.14)
Test for heterogeneity: $\chi^2_{12} = 25.63$; P<0.05		

Table A9.17. Ratio of observed number of deaths from genito-urinary diseases in uranium workers compared to that expected in the general population (Beral and Darby, 2001)

Reference	Total number of deaths	O/E (95% CI)
McGeoghegan and Binks, 2000a	28	0.57 (0.38-0.83)
Ritz et al, 2000	5	0.78 (0.25-1.81)
McGeoghegan and Binks, 2000b	7	0.98 (0.39-2.02)
Ritz 1999a	3	0.21 (0.04-1.29)
Frome et al, 1997	270	0.83 (0.73-0.94)
Brown and Bloom, 1987	3	0.54 (0.11-1.56)
Stayner et al, 1985	2	0.89 (0.11-3.18)
Summary value	318	0.70 (0.54-0.87)
Test for heterogeneity: $\chi^2_6 = 8.04$; P>0.1; NS		

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Appendix Ten

IARC Monographs on the Evaluation of Carcinogenic Risks to Humans

Ionizing Radiation, part 2: Some Internally Deposited Radionuclides (Volume 78) (14–21 June 2000)

Internally deposited radionuclides that emit α particles are *carcinogenic to humans (Group 1).*
In making this overall evaluation, the Working Group took into consideration the following:

- α Particles emitted by radionuclides, irrespective of their source, produce the same pattern of secondary ionizations and the same pattern of localized damage to biological molecules, including DNA. These effects, observed in in-vitro systems, include DNA double-strand breaks, chromosomal aberrations, gene mutations, and cell transformation.
- All radionuclides that emit α particles and that have been adequately studied, including radon-222 and its decay products, have been shown to cause cancer in humans and in experimental animals.
- α Particles emitted by radionuclides, irrespective of their source, have been shown to cause chromosomal aberrations in circulating lymphocytes and gene mutations in humans *in vivo*.
- The evidence from studies in humans and experimental animals suggests that similar doses to the same tissues—for example lung cells or bone surfaces—from α particles emitted during the decay of different radionuclides produce the same types of both non-neoplastic effects and cancers.

Internally deposited radionuclides that emit β particles are *carcinogenic to humans (Group 1)*
In making this overall evaluation, the working group took into consideration the following:

- β Particles emitted by radionuclides, irrespective of their source, produce the same pattern of secondary ionizations and the same pattern of localized damage to biological molecules, including DNA. These effects, observed in in-vitro systems, include DNA double-strand breaks, chromosomal aberrations, gene mutations and cell transformation.
- All radionuclides that emit β particles and that have been adequately studied, have been shown to cause cancer in humans and in experimental animals. This includes hydrogen-3 (tritium), which produces β particles of very low energy, but for which there is nonetheless *sufficient evidence* of carcinogenicity to experimental animals.
- β Particles emitted by radionuclides, irrespective of their source, have been shown to cause chromosomal aberrations in circulating lymphocytes and gene mutations in humans *in vivo.*
- The evidence from studies in humans and experimental animals suggests that similar doses to the same tissues—for example lung cells or bone surfaces—from β particles emitted during the decay of different radionuclides produce the same type of non-neoplastic effects and cancers.

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Appendix Eleven

Illustrative Proportions of Depleted Uranium Around a Site

Extracted from Appendix Three, Report of the WHO Depleted Uranium Mission to Kosovo (WHO, 2001a)

The amount and fate of depleted uranium at an 'average' attack site has been hypothecated as a way of exploring if or how the potential for depleted uranium to become mobilized can be visualized. The 'typical' site situation developed using data available to the mission on the type of attacks by A-10 aircraft undertaken in Kosovo in 1999 is presented in Appendix 3. A summary of the results is given here:

1. Rounds fired
31000 rounds were fired from A-10 aircraft (rounded up to next one thousand)
divided by
112 separate attacks by A-10 aircraft in Kosovo

Hence, the average number of rounds fired during each attack = 300 (rounded up from 277)

Geographical sites that were attacked more than once were not considered in this hypothetical illustration.

2. Depleted uranium : high explosive rounds
Four depleted uranium (DU) rounds are fired for every high explosive (HE) round

Therefore, for an average of 300 rounds at a ratio of 4:1 (DU:HE) this would involve the use of 240 depleted uranium rounds at each site

3. Weight of depleted uranium
Each of the 240 rounds has a penetrator containing depleted uranium weighing 300 g

Therefore, the weight of depleted uranium that could be reasonably expected at a typical site is 72 kg

4. Fate of the depleted uranium rounds
Combat simulations in America found that less than 10% of the rounds fired by an A-10 aeroplane hit the target (Stolfi et al. 1979)

From the 240 rounds of depleted uranium at a typical site, 24 could be expected to hit the target and 216 would miss the target. Of the 216 rounds that missed the target, 80% (172 rounds) would land within a 100 m radius of the target and 20% (44 rounds) could land up to one nautical mile (1.85 km) radius from the target. This is based on details provide by KFOR.

4a. Rounds hitting the target and igniting

A round hitting the target should catch fire and less than 50% of the mass of the depleted uranium penetrator will burn, with the remainder as un-burnt fragments and dust.

Correspondingly, using a worst case of 50% of the mass igniting, the estimated amount of depleted uranium converted into uranium oxides as combustion products would be 3.6 kg:

24 rounds hitting the target X 50% of 300 g = 3600 g (3.6 kg)

4b. Rounds missing target and un-burnt

The rounds missing the target are regarded as less likely to ignite and will remain as depleted uranium metal.

216 rounds missed the target X 300 g together with the remaining un-burnt portion of the 24 rounds hitting the target X 50% of 300 g = 68400 g (68.4 kg)

The total quantity of depleted uranium metal at the typical site would be 68.4 kg

5. Surface or buried

For the 216 rounds that missed the target on a hard surface such as concrete, almost all of the penetrators would probably remain on the surface as intact objects or broken fragments.

On soft ground such as moist soil, it is assumed that the 80% of missed rounds expected to land within 100 m of the target (172 penetrators) would enter the soil and become buried at depths up to three m. This is because of the normal, reasonably steep angle of attack of the A-10 aircraft. The remaining 20% (44 penetrators) expected to land further away do so, in part, because of the shallowing angle of attack at the end of an air strike. The penetrators are therefore assumed to represent the likely number that may land on the surface.

6. Summary of depleted uranium at a target site

Spatial distribution of penetrators around an attack site:

Depleted uranium penetrators hitting the target	24	10%
Depleted uranium penetrators landing within 100 m radius	172	72%
Depleted uranium penetrators landing within 1.85 km radius	44	18%
Total at the typical site	240	

Likely form of uranium after an attack:

Depleted uranium as uranium oxides following combustion:	3.6 kg	5%
Depleted uranium metal	68.4 kg	95%
Total at the typical site	72 kg	

Likely location of penetrators on the surface and below ground:

Depleted uranium buried (172 x 300 g)	51.6 kg	72%
Depleted uranium on surface as oxides	3.6 kg	5%
Depleted uranium on surface as un-burnt fragments	3.6 kg	5%
Depleted uranium on surface as metal (44 x 300 g)	13.2 kg	18%
Total at the typical site	72 kg	

7. Penetrators likely to contribute to measurable surface radiation

The measurable surface radiation extends only a tiny distance from a piece of depleted uranium. Therefore, the weak radiation from all of the estimated 72% of the penetrators at a typical site with soft soil would be isolated from people walking over the ground.

Consequently, for the scenario an equivalent of about 68 penetrators (intact, fragments or dust) could be on the surface. This represents around 20.4 kg including the quantity becoming uranium oxide combustion products after ignition when the target was hit.

8. Degradation rate for buried penetrators

The rate of corrosion of uranium metal in the environment is slow. The presence of a small proportion of titanium will act to further slow the rate of environmental degradation. A figure of 500 years to decompose a 300 g object was discussed in section 3. Consequently, it is regarded as unlikely that the penetrators will degrade quickly once in the environment and hence they will only contribute a very slow leaching of uranium into the environment. Once leached, the uranium may well become sorbed in an immobile form to the soil or diluted substantially in soil and ground waters.

One literature reference refers to a typical natural uranium composition in soil as four US tons per square mile to a depth of 12 inches. The metric equivalent is 1.4 tonne per square kilometre to a depth of 30 cm.

Even if all of the depleted uranium at the typical site (72 kg) eventually degraded, all within one square kilometre of the target and none was removed, then the additional amount of uranium compared to the natural occurrence of uranium in the soil would be 5%.

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Appendix Twelve

Scenarios and corresponding effective doses from Depleted Uranium

Adapted from Ref: Appendix 8 - UNEP/UNCHS Balkan Task Force Report October 1999 and European Commission Report, 2001 including Brenk Systemplanung modelisation results (BS scenario) and other scenarios (*)

Scenario of exposure	Result / Comment
0. Common assumptions to all scenarios 10 kg DU particles < 10µm slow soluble oxide:*Type S* Area contaminated: 20 x 50 m = 1000 m2 BS scenario 1: 32 x 32 m = 1000 m2 BS scenario 2: radioactive build-up after 1000 years For BS, *effective dose* in this table are for adults. UNEP results are supposed for adults also. Based on this frame realistic scenario are used to evaluate an exposure …	This is the basic scenario described in UNEP October 1999 report Brenk Systemplanung (BS) gave some modelisation results with the same basic scenario (introducing a Darcy velocity in 5.5 = scenario 1) and assessing the impact in 1000 years time (considering then in 5.5 that yearly leaching of DU is reduced from 10% of initial amount – 1 kg – to 3.68 g.) … the results are given in this column. Comments on chemical toxicity come from UNEP report
1. Picked up solid pieces of DU In this scenario, fragment of penetrators are supposed to be picked up and either kept in a pocket, worn as an ornament or stored on a shelf on the bedside. β radiation : 2 $mSv.h^{-1}$ several weeks in pocket (reduced by 50%) * worn as ornament: 2.3 $mSv.year^{-1}$ 500h w_T = 1% tissue weighing factor Impact on 1 cm^2 / $2m^2$ body * on a shelf on the bedside	Resulting skin dose is *negligible* (0.6 $\mu Sv.year^{-1}$) skin *dose* may be high but NO *deterministic effects* skin dose = 0.6 $\mu Sv.year^{-1}$
2. Solid pieces of DU in or on ground … in this case external exposure is considered	see **1** or **6**
3. Instantaneous inhalation of DU dust after an attack Following the explosion of a shell (with DU penetrator) or a plane crash (with DU counterweights) followed by a fire, inhalation of the resulting "aerosol" is considered. 1 mg dust maximum instantaneous intake 10% DU in dust * 1992 Plane crash + fire in Bijlmer (NL) Dose to bystanders: fraction - all DU counterweights converted to aerosols	Resulting 100 mg DU intake " might lead to acute chemical toxicity and total effective radiation dose caused by inhalation of less than 10 mSv." plane crash *1 µSv – 0.7 mSv
4. Inhalation of resuspended DU People are considered breathing air loaded with DU contaminated dust, considering it has been produced by an explosion, dispersed and settled on the 1000 m^2 area around target site. Due to wind it is resuspended ready to be breathed All DU dust is included in 1 mm upper layer of soil 2 hours stay – 1 $m^3.h^{-3}$ breathing rate Dust concentration : 50 $\mu g.m^{-3}$ to 5 $mg.m^{-3}$	Resulting concentration in dust = 6 µg DU / mg dust intake : 0.6 – 60 µg of DU effective dose negligible (0.07 – 7 µSv)
5. Ingestion of DU … in different situations	
5.1 Soil in mouth In this scenario a child is supposed playing on a contaminated area and eats soils soil concentration: 6 µg DU / mg 1 g soil ingested (by a child playing) BS scenario 2: 0.031 µg DU / mg 44 $g.year^{-1}$ ingested (or 1g)	Effective dose negligible (4 µSv) possible acute chemical toxic effect for 60µg DU BS: 1 $\mu Sv.year^{-1}$ (or 4 µSv also for BS model with same assumptions)

Scenario of exposure	Result / Comment
5.2 Contaminated vegetables Member of the public is supposed eating vegetables growing on the area contaminated by dust before it is washed away by a rainfall. Area contamination (from "0"): 10000 mg.m^{-2} before washing by rainfall Intake: 60 kg.year^{-1} or 1 kg.week^{-1}, growing on ~ 1m^2 of land	Intake ~ 100 mg DU … "significant from chemical risk point of view. Resulting radiation dose will be (tolerable) of the order of 0.1 mSv." (Washing reduces contamination by 99%)
5.3 Contaminated hand Same soil characteristics as 5.1 but the quantity ingested is lower. 10-100 times lower soil ingestion than 5.1	Effective dose negligible: 0.04 to 0.4 µSv
5.4 Open wounds In this case, the skin is in direct contact with dust particles, supposed not cleaned away. Contact α and β-radiation: < 50 mSv.h^{-1}	"No acute *deterministic effect*".
5.5 Contaminated water The ground water table is supposed contaminated by dissolution of DU spread on the surface: the water is used for drinking. soil depth between ground water table and surface of the "bedrock" is 3m 10 kg DU contaminated area: 1000 m^2 leakage of DU: 10% per year of deposited amount of DU Water content of the ground 30% BS scenario 1: Darcy velocity 150 m.year^{-1} 500 l.year^{-1} BS scenario 2: … release of 3.68 g DU per year	Doses are negligible (tolerable in any case) ? In UNEP report : 1 g DU.m^{-3} in drinkable water. This is "above hygiene standards for chronic exposure … chemical toxic effects cannot be excluded. The annual radiation dose … about 1 mSv.year^{-1}" (UNEP considered a **stagnant groundwater** whereas BS, more realistic, considers it **flowing** and involving 16 times more water, equivalent ratio between doses) BS 1: 24 µSv.year^{-1} with conservative parameter value BS 2: 0.1 µSv.year^{-1}
5.6 Contaminated food The food is contaminated either by soil ingestion (cattle) or because vegetables are growing on the contaminated groundwater table. <u>Contaminated meat and milk …</u> <u>Contaminated plants by root uptake:</u> 10 kg DU on 1000 m^2 – 10 cm deep soil density 1500 kg.m^{-3}: 70 mg DU.kg^{-1} soil BS scenario 2: consumption 100 kg.year^{-1}	Dose negligible Meat and milk: "… less than 0.1 mSv per day …": keep –cattle away from contaminated areas. UNEP calculations give "effective dose by ingestion of 7 µSv.year^{-1} BS 2: 2.6 µSv.year^{-1}
6 External radiation The population is exposed externally to the dust spread over the area. mixing of DU in 10 cm (70 mg DU.kg^{-1} soil) - DU is 0.8% of γ radiation of natural U (17 nGy.h^{-1} or 0.02 mSv.year^{-1}) indoor occupancy: 0.8 BS Scenario 2: mixing of DU in 5 cm 12 h per day inside house – 900 h.year^{-1} in garden * Armor combat crew 1000 h.year^{-1} 0.1 to 1.8 µSv.h-*1*	Doses negligible for population 4 µSv.year^{-1} 2.7 µSv.year^{-1} * 0.1 to 1.8 mSv.year^{-1} (AEPI, 1995)
7 According to extension of the impact If the total amount of DU used and resulting contamination is considered spread on larger areas than previously supposed or if all target sites are near and considered in the same region, dispersion is higher and resulting contamination reduced. 10 tonnes of DU on 112 targets have been used that is an average of 100 kg on each 1000 m^{-2} target area (see "0") of which 90% non exploded = 10 kg * If dispersion of DU dust is higher, … 5.5 and 5.6: depending on homogeneity of acquifer.	Effective dose should be lower than those highlighted above.