

4 Industrial, commercial and military applications

Whereas the mining of uranium has taken place since the Middle Ages it is only in the last 100 years, and particularly the last 50 years, that mining has taken place on a large scale. Estimated total production of metallic uranium since recording began in 1920 is estimated at 1.5 million tonnes (British Geological Survey, 2000) although this is only a small fraction of the 10^{14} tonnes estimated to be present in the lithosphere. Recent production figures for a range of acknowledged producers are given in Annex 3.

Uranium by itself has relatively few industrial uses and is commonly removed from raw materials such as fertilizers containing phosphate as a waste product during processing, or because of environmental concerns where it is often removed with other heavy metals such as cadmium.

4.1 Historical uses

Glassware and ceramics Prior to the discovery of radioactivity, uranium was principally used in the colouring of ceramics and glass. Sodium and ammonium diuranates were used as a yellow glaze, although they could also be used at increased concentrations to produce cream, orange, brown, green or black glazes (Chu and Chu, 1975; Conrad, 1973). In glass, uranium is used typically at concentrations in the range of 0.1% to 2% by mass; it produces a fluorescent yellow or light green glow making it possible to easily identify this type of glass. Production of glass containing uranium continued until the middle of the 20th century and was called by various names depending on its colour. For example popular German names include ‘Annagelb’ for yellow glass and ‘Annagruen’ for green glass, whereas in the UK and the US such glass is usually referred to as ‘Vaseline glass’. Depleted uranium has been and is potentially still used as the basis of a yellow enamel powder used in the manufacture of badges and jewellery (NUREG, 1999).

Dentistry Until the early 1980s uranium and DU were also used in the production of dental porcelains to obtain a natural colour and fluorescence (Sairenji et al., 1982 Noël et al., 1988 and NUREG, 1999). Uranium concentrations of 170 to 13 300 mg/kg were found in 15 types of porcelain powders from one manufacturer (Noël et al., 1988) while Sairenji et al. (1982) determined uranium concentrations of 0.5–24.7 mg/kg in eight types of porcelain powders marketed in Sweden. The US National Regulatory Commission (NRC) standard for uranium in dentures is 500 mg/kg (NUREG, 1999).

Chemical catalysts Uranium has also occasionally been used as a catalyst in certain specialized chemical reactions and in photographic films. For example the oil and gas industry continue to use nickel-U material (10%–65% DU) in relatively large quantities as a catalyst. The nuclear industry has also used small quantities of DU to chemically absorb gaseous tritium for the purposes of transportation, and in the production of plutonium. In the period between the discovery of radioactivity and nuclear fission, uranium ore (typically as pitchblende) was mined principally for the extraction of radium (a decay product of uranium) which was used for medical purposes and the preparation of luminous paints.

Nuclear weapons and power production From 1938, when nuclear fission was first identified, uranium mining expanded due to the need for enriched uranium as a fuel for use in nuclear power stations and for nuclear weapons.

The first step in the process is mining; historically, this has involved subterranean or open-pit ore rock removal and now also involves liquid in-situ leaching of unconsolidated deposits (the latter method has been used extensively in the Czech Republic and the former East Germany). The next steps in the production of uranium fuel, aimed at concentrating the uranium, are usually carried out near the mine to save on transportation costs. Methods used depend on the nature of the ore, and may involve mechanical procedures such as crushing, screening and flotation, followed by acid or alkaline leaching, solvent extraction, or ion exchange and eventual precipitation. The product of these concentration steps contains perhaps 40%–70% uranium by weight and is generally shipped to a central processing plant to be further refined. This purification is either by digestion with nitric acid and extraction of the resulting uranyl nitrate into an organic solvent, or by conversion to UF_6 and fractional distillation of that volatile compound. At this stage, all naturally occurring radioactive progeny have been removed from the uranium which is considered to be chemically pure. However, due to the inherent radioactive nature of uranium, following this purification step the presence of other radioactive elements within the uranium decay chains will start to increase.



Figure 4.1 Nuclear fuel rods being loaded.

Whilst some nuclear reactor types such as the CANDU and Magnox use natural uranium as fuel, others require enriched uranium (see Figures 4.1 and 4.2). If such ^{235}U enrichment is required, the purified uranium in an appropriate chemical form is transferred to an isotope separation plant (see Annex 1). Isotope separation of ^{238}U , ^{235}U and ^{234}U may be achieved by a number of processes including gaseous diffusion, centrifugal or laser enrichment. The enriched uranium is then processed and fabricated into appropriate forms for use in nuclear reactors. The by-product of this enrichment process is DU, often in the form of UF_6 , as has been discussed above.

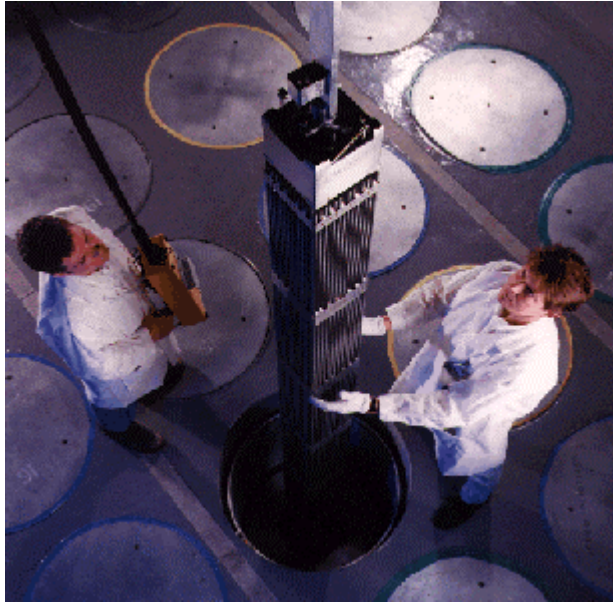


Figure 4.2 Uranium fuel assembly used in a pressurized water type of nuclear reactor.

4.2 Current applications

The four major uses of DU at the present time include radiation shielding, counterbalance weights and military armour and ammunition.

Radiation Shielding The density of DU makes it a suitable material for the shielding of gamma radiation. For this reason, uranium has been used extensively in the medical, research and transport sectors (NUREG, 1999) as radiation beam collimators and containers to transport nuclear sources. Thus, DU has often been used as a shield for radioactive sources in tele-therapy units (figure 4.3) used in the treatment of cancer and in linear accelerators. Typical quantities of DU used in such equipment range from tens to hundreds of kilograms.



Figure 4.3 Cobalt-60 tele-therapy unit

Current developments in waste management have also employed DU as a shielding material. For example, casks used for holding spent fuel in the nuclear power industry

have been constructed by combining DU with concrete (e.g. DUCRETE™ (www.starmet.com, 2001)). This achieves a significant increase in gamma-radiation shielding with thinner shield walls and much lighter weight casks than traditional storage casks. Appreciable quantities (i.e. thousands of kilograms) of DU have been used as shielding material in casks used for the transport of radioactive sources such as those used in the medical and engineering industries.

Counterbalance weights and ballast Vessels and equipment, such as boats and satellites require a large amount of weight to be carried in the form of ballast. The high density and relative availability of DU make it a potentially suitable material for this use by fulfilling the weight requirements while minimising the amount of space taken up by the ballast material.

Industries in which the use of DU has been cited for these purposes include the aircraft industry, military aerospace industry, and the oil and gas exploration and production industry (NUREG, 1999). It has also been suggested that it has been used in the manufacture of keels for yachts (Priest, 2001).

Wide-bodied aircraft such as the McDonnell-Douglas DC-10, Lockheed L-1011 and Boeing B-747 require heavy counterweights on control surfaces (usually, but not exclusively, the aileron) to enable proper flight control. These areas often have low surface clearance with insufficient space available for low density materials. Depleted uranium, lead and tungsten have all been used for counterweights due to their high density. A typical wide-bodied aeroplane such as the Boeing 747 ('jumbo jet') requires up to 1500 kg of counterweights (NUREG 1999). Not all of this material is DU however, and DU is now being replaced retrospectively with tungsten.

The plane which crashed into a block of flats in Amsterdam in 1992 carried 282 kg of DU counterweights and the Korean Boeing-747 which crashed near Stansted airport in England in January 2000 was estimated to be carrying approximately 425 kg of DU counterweights (Uijt de Haag et al., 2000). When used in aircraft DU is usually either plated (Ni and/or Cd) and painted or sheathed in an aluminium alloy. No data appears to have been generally issued on DU released from other civilian or military air crashes.

The Nuclear Regulatory Commission's report (NUREG, 1999) comments that 'It is unknown how many DU counterweights are currently installed in aircraft. It is estimated that approximately 15,000 weights may be associated with the Boeing-747 fleet (based on 550, Boeing-747 aircraft produced between 1968 and 1981 and spare parts) (Gallacher, 1994). However, the number of aircraft that contain DU counterweights is rapidly decreasing. Rather than refurbishing the DU (during maintenance operations), tungsten counterweights are used as a replacement.'

Similarly DU may be used as weights for geophysical exploration tools, as rotor tips in military helicopters (AEPI, 1995) and experimentally in many different forms of engineering applications such as counterweights in various engines (particularly during the 1980s when its commercial use was seen as a way of reducing stockpiles of such material). There is little evidence that the use of DU has continued into the production of any of these experimental applications, although the pressure to find appropriate commercial outlets for DU is undoubtedly still present.

Military Uses Depleted uranium (and associated uranium-titanium alloys [typically 0.75 wt % Ti]) have been employed by the military as a component of heavy tank armour and armour-piercing munitions (e.g. AEPI, 1995; Rao and Balakrishna Bhat, 1997). The high density and high melting point of DU make it an extremely effective material for neutralising anti-tank weapons. The DU is inserted into a sleeve attached to the regular steel armour, thus isolating the DU from the tank crew and those in contact with the external surfaces of the tank while utilising its protective characteristics.

The high density of DU and its various alloys also makes it suitable material for use in armour piercing munitions and to penetrate hardened targets. Depleted uranium also has advantages over similarly dense alternative materials, such as tungsten, in that it is:

- relatively inexpensive.
- non-brittle unlike tungsten.
- at the high temperatures and pressures involved during the impact of such weapons DU has been found to adiabatically shear (e.g. self sharpen) giving increased penetration.

There are four main types of DU munitions acknowledged as being currently in circulation, the 25 mm, 30 mm, 105 mm and 120 mm anti-tank rounds, although small amounts of DU have been used in the manufacture of other munitions (AEPI, 1995). A 30 mm round fired from ground attack aircraft contains a 0.27 kg DU penetrator. Heavy tanks fire 120 mm rounds containing a 4.85 kg DU penetrator. A 30 mm cannon as used in a ground attack aircraft can fire up to 4200 rounds per minute (although such munitions are typically only fired in relatively short bursts of say 120 to 195 rounds or two to three second bursts (CHPPM, 2000)) and a considerable mass of DU could be distributed in an attack zone, particularly if the attack is performed by a number of aircraft. Some of this DU may be released as particles should the penetrator impact on a sufficiently hard target, but the entire round may stay intact with only surface scaring even when impacting with relatively hard targets such as concrete.

DU counterweights may also be used in missiles (AEPI, 1995), warheads and military aircraft. For example some land-attack cruise missiles (Zajic, 1999; personal web site) and other strategic missile systems, such as the trident ballistic missile system, have been reported as using DU as counterweights, although this has not been substantiated by official sources. One use of DU within missile warheads might be to aid ground penetration. For example the now obsolete Pershing D-38 earth-penetrating missile carried an 80 lb (36.3 kg) DU penetrator. In one case recorded at a missile testing range, the warhead used in this type of missile penetrated the earth to a depth of approximately 200 ft (61 m) (Van Etten and Purtymun, 1994).

Data relating to the use of DU in a wide range of weapons systems produced outside of the USA and NATO are generally lacking, although DU weapons are widely available. For example they are thought to be in the possession of at least nine countries (Harley et al., 1999b). Numerical estimates of the relative proportion of DU munitions amongst arsenals of various countries are equally difficult to obtain, although it has been reported in AEPI, (1995) that US contractors had produced more than 55 million DU penetrators for small-calibre munitions (principally of the 30 mm type) and 1.6 million penetrators for tank ammunition. The potential use of DU in other forms of armaments to enhance hard target penetration is briefly reviewed with respect to the recent conflict in Serbia by Liolios, (2000).

Depleted uranium weapons are regarded as conventional weapons by NATO and are not subject to restriction. However, the extent of their use has, in the past, been unclear. It is now accepted and acknowledged that DU weaponry was employed by US forces in the Gulf War. A letter from Lord Robertson, Secretary General of NATO to the Secretary General of the United Nations, Mr Kofi Annan dated 7th February 2000 also confirmed the use of DU munitions during the Kosovo conflict. The use of DU weapons at various firing ranges associated with the development and proving of munitions and armour (e.g. at the Yuma and Aberdeen proving grounds in the USA (AEPI, 1995)) has also been widely acknowledged.

4.3 Summary

Depleted uranium is used in a variety of products. Its major uses include radiation shielding, counterbalance weights and ballast, military munitions and armour. The use of DU has been acknowledged in a number of military conflicts including the Gulf War, Bosnia and Kosovo and in various military firing ranges. Use in such conflicts has clearly demonstrated the military benefits of DU.

5 Factors influencing routes of intake and exposure

5.1 Introduction

Exposure to uranium occurs primarily by inhalation, ingestion or external irradiation, while secondary exposure may result from dermal absorption of particularly mobile forms of uranium. This is the case with many chemicals naturally present in our environment. Because of the identical chemical and biological behaviour of naturally occurring uranium and DU, it is possible to draw on our knowledge of exposure to other naturally occurring heavy metals (e.g. lead, arsenic etc.) in attempting to define the relative importance of exposure pathways.

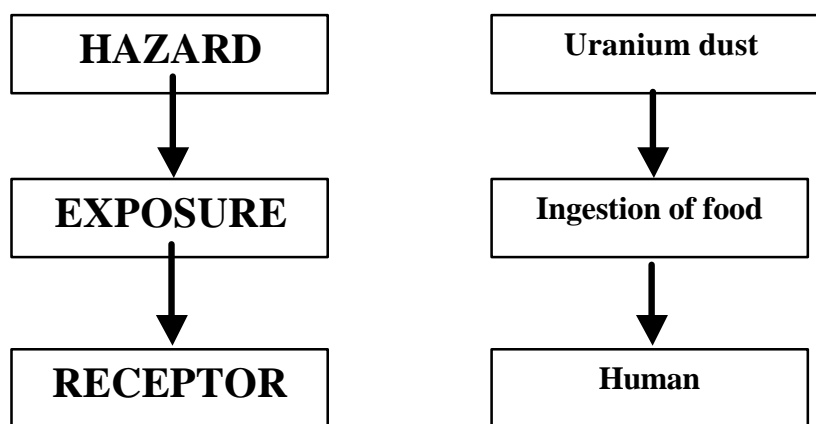


Figure 5.1 Example of a simplified pollutant linkage, for human exposure to uranium. In practice, however, such a linkage only represents one of a much larger number of potential linkages whose relative importance within a specific incident may have to be investigated (Ferguson et al., 1998).

Risk assessment models in which the potential health detriment associated with exposure to a given chemical is calculated are reliant on the definition of a ‘pollutant linkage’ such as that shown in Figure 5.1. Several different such linkages may occur at any one site associated with the presence of various hazards, environmental pathways and receptors such as humans, animals, groundwater, and buildings.

Exposure by external irradiation is not explicitly dealt with in this chapter, but is discussed in the following chapters where health risks associated with radiological issues are reviewed.

Studies on exposure to uranium have focussed on those chemical forms and modes of exposure of direct relevance to the nuclear industry. In comparison relatively few studies have been conducted on DU. It should be recognized that the extensive database on uranium will be of value in assessing, in many cases, the implications of exposure to DU since they have the same chemical and biological behaviour.

Scenarios used in previous studies on exposure of humans to DU in military situations have focussed on the likely form of compounds and exposure routes by which personnel or the local population may be exposed in the hours or days immediately following the use of DU in munitions and armour. In order to undertake longer term assessments a more in-depth analysis is required of the physiochemical transformations that control

the weathering and transport of uranium through various environmental pathways and compartments by which human exposure may occur. This more detailed analysis should be structured to evaluate the speciation and bioavailability of uranium under the prevailing local environmental conditions. It is well established that the total metal concentration in an environmental medium is an unreliable guide to hazard quantification, as different forms of a metal can have substantially different bioavailabilities (Thornton, 1996; Plant et al., 1996; Elless et al, 1997).

It is noted that regional or local exposure scenarios are often structured to estimate the risk of two different types of health detriment:

- (i) population detriment.
- (ii) maximum individual detriment.

Population detriment is a traditional public health measure that estimates the number of cases of a particular outcome or disease in an exposed population attributable to a specific source of contamination. The maximum individual detriment relates to the individual who suffers the largest incremental risk due to a particular scenario. The relative importance of different sources and pathways is likely to differ depending on whether population detriment or maximum individual detriment is being calculated. In the case of population detriment it is particularly important that not only is the average exposure estimated, but also that its spatial distribution, and the relative importance of various exposure routes amongst the local population are defined.

Children are not small adults and their exposure may differ from an adult in many ways. Unfortunately, despite their obvious importance little definitive data exists concerning how their uranium exposure differs from that of adults (ATSDR, 1999).

Examples that follow in this chapter illustrate the relative importance of various exposure routes when assessing exposure within the context of population and maximum individual detriment and suggests factors that need to be considered when undertaking more site-specific studies. Such treatment is important in understanding the relative proportion of total exposure that may be allocated to a specific pathway during the assessment of human health risks (WHO, 1994), particularly where substance exposure pathways may exist.

The potential relationships between exposure to uranium and DU and specific forms of health detriment are discussed in Chapters 8 and 9.

5.2 Exposure via inhalation

The pyrophoric nature of uranium is considered to be of special relevance to the assessment of human exposure due to the production of dust containing mixed oxides of uranium. This scenario is especially likely to occur immediately following the use of DU munitions or where DU may be accidentally or deliberately heated (e.g. in the welding of reclaimed battlefield scrap). Its relevance to aviation accidents remains a subject of debate.

The oxides considered to be of principal concern are uranium dioxide (UO_2), uranium trioxide (UO_3) and triuranium octaoxide (U_3O_8) (Harley et al., 1999b; CHPPM, 2000). The size distribution, morphology and exact chemical composition of each particle released during the use of penetrators and armour is highly variable (e.g. Patrick and

Cornette, 1977). Moreover, they may be chemically and mineralogically altered by weathering either following the impact with the target, or their initial release into the environment (e.g. uranium oxides may become hydrated, chemically reacting with other elements and species present in the soil and/or target such as silica, iron, phosphate and vanadium; Patrick and Cornette, 1977 Ebinger et al., 1990). The relative absorption behaviour (lung to blood) for uranium after the inhalation of different chemical forms is given in Table 5.1.

Table 5.1 Absorption types for uranium compounds (ICRP-71, 1995b). Note. Some preparations of U_3O_8 may also be considered as Type S.

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

F fast; M moderate; S slow

The biological solubility and bioavailability of the oxides U_3O_8 and UO_2 are relatively low (Type M and S), compared to other forms of uranium to which workers in the nuclear industry may be exposed (e.g. UO_3). This has been reported by Jette (1990) through lung-solubility analysis of particles ($< 10 \mu m$ in diameter) produced immediately following the impact of DU munitions. Some of these particles may, however, be removed by mucociliary transport into the gastro-intestinal (GI) tract, and reach the intestine where appropriate gut uptake factors need to be considered. For more information on deposition and clearance from the respiratory tract the reader is referred to Annex 4.

There is a lack of detailed mineralogical and chemical analysis of material liberated under battlefield conditions (or other conditions in which uranium dust has been liberated following combustion) and subsequently weathered. This limits the accuracy of exposure assessment. The availability of material specific information would inevitably increase the confidence of such assessments. The lack of such data has also highlighted a significant knowledge gap in performing detailed exposure assessments for scenarios involving military personnel (CHPPM, 2000).

The level of human exposure to dusts and aerosols derived from the impact of possible uncontrolled oxidation of DU is a function of the proximity of the human subject to the source of contamination, the degree to which uranium has become physically and chemically dispersed into the local environment and the particle sizes and density of the dusts produced.

Dust particles with a diameter less than $10 \mu m$ activity mean aerodynamic diameter (AMAD) are generally assumed to be respirable, larger particles being trapped in the upper extra-thoracic part of the respiratory tract from where they are either expectorated or swallowed. In the case of radiological exposure the respiratory tract is both a target organ and a route of entry to the systemic circulation. There are two lung models in current use; the ICRP Publication 30 lung model (ICRP-30, 1979) and the ICRP Publication 66 Human Respiratory Tract Model (ICRP-66, 1994a). The latter is used

internationally and is embodied in European Legislation and the Basic Safety Standards (1996). The principals underpinning these models are discussed in more detail in Annex 4. While these models are generally used for radiological exposure, their more detailed treatment of exposure via inhalation offers considerable scope in improving the assessment of chemical exposure via this potentially important route (e.g. CHPPM, 2000).

For the purposes of generalized environmental exposure modelling, approximately 73% of the dust in air is considered to be respirable and the concentration of contaminants in the dust is assumed to be equal to the concentrations in the soil (Muir et al., 1995). Local variations in such factors are high and locally determined factors should, if possible, be used during more extended assessments. The ICRP Human Respiratory Tract Model (ICRP-66, 1994a) offers a more comprehensive treatment in this respect, calculating deposition throughout the entire respiratory tract for a given particle size distribution.

The US EPA estimates that a typical exposure to uranium in air results in a total uranium intake estimated to be from 2 to 20 nanograms ^{238}U /day (US EPA, 2000).

Individuals likely to suffer enhanced exposure to uranium via inhalation are mainly those living or working in close proximity to primary sources of uranium dust such as mine tailings or areas in which DU has been used for various industrial or military activities. Agricultural workers, particularly those involved in cultivation practices such as vines that require the operators nose and mouth to be in close proximity to the soil's surface, or those working with heavy dust-generating machinery, are also likely to be exposed to enhanced levels of uranium containing re-suspended dusts. Limited data are available describing possible re-suspension of DU particles once deposited on soil. Data presented in CHPPM (2000) and cited to have been compiled in a short letter based report by Beyeler, suggests re-suspension factors for DU oxides to range from 3.3×10^{-8} (no mechanical disturbance) to 1.9×10^{-4} (vigorous work activity).

5.3 Exposure via ingestion

The impact on health after ingestion depends on the amount of bioavailable uranium compounds present in ingested material, which in turn is dependent on the concentration of bioavailable uranium in the environment. In most scenarios in which uranium has been released into the environment, it is assumed that it could contaminate the soil, or migrate to surface or groundwater. Using these as sources for drinking water, agricultural land and recreational purposes can lead to human exposure either directly or through the food chain. Therefore, information on the possible concentrations of soluble uranium compounds in these media, in the days and years following the contamination incident is essential. In this context, DU and even anthropogenically redistributed uranium has entered the environment only relatively recently. This, together with a lack of reliable field data where DU has been used for military purposes, significantly limits our knowledge of how, and on what time scale, such materials may become incorporated into the human food chain. This places a high degree of importance on the extrapolation of studies of uranium in natural systems. However, during any such extrapolation it is important to consider likely differences in the bioavailability of the specific chemical forms of DU encountered, compared with those of substances from which extrapolations are to be made.

Because of the diversity of the human food chain it is impossible to suggest a suitably generic dietary balance to cover all populations, scenarios and cultures. WHO has, however, suggested a typical diet to aid in the comparison of exposure; these data are identified in Annex 3.

Like many trace metals the bioavailability of uranium in food, water and soils, affects uptake into the body through the gastrointestinal tract. For example a high-phytate content in the diet may reduce uptake (Gibson, 1994; Golden and Golden, 1981) whereas the presence of lower molecular weight ligands, such as citrate, may promote absorption. However, under the near-neutral conditions of the upper intestine uranium, unlike many heavy metals, is likely to form relatively stable oxy-anion complexes (e.g. Brookins, 1988) that inhibit complexation with organic chelators. Therefore, although the bioavailability of uranium from foodstuffs is an important variable in the exposure assessment process, its extrapolation from the results of other studies on other heavy metals such as Zn, Pb, etc should be avoided. Although unable to determine uptake factors, Spencer et al., (1990) investigated intake and excretion patterns of naturally occurring uranium and calcium in humans. These studies confirmed the similar behaviour of uranium isotopes in the body and confirmed significant elimination of the total dietary intake via faecal excretion (urinary excretion being approximately 2% of total excretion (e.g. Leggett and Harrison, 1995).

ATSDR (1999) considers ingestion to be the major source of environmental exposure to uranium. Typical world-wide dietary intake is estimated at between 0.9 and 4.5 $\mu\text{g}/\text{day}$ with an average of 1.5 $\mu\text{g}/\text{day}$ (Linsalata, 1994). This is consistent with dietary intakes estimated from the excretion of uranium in urine amongst a group of 12 subjects from Utah (Singh et al, 1990). However, ATSDR (1999) cites a paper describing dietary intake as high as 2.9 to 4.5 mg/day for individuals living near a uranium mine (Yamamoto et al, 1971).

Various gut uptake factors and bioavailabilities have been suggested for ingested uranium and these are discussed in more detail in Chapter 7 with specific reference to the ingestion of water, foodstuff and soil/dust.

5.3.1 Staple foods

There are few data on the concentration of uranium in staple foods from an environment that could be considered to be contaminated with DU. Whereas within the context of the nuclear industry extensive compilations of concentration factors (CF) and concentration ratios (CR) exist in the literature (e.g. ICRP-29, 1978); IAEA, 1982, 1994; see Table-5.2). These have been extensively used in radiological assessments to estimate the likely concentration of a given radionuclide (such as uranium) in a foodstuff from the concentration of that radionuclide in a given substrate (eg. water, food or soil) to which the organism is exposed. In addition to data presented for generalized environments, a significant amount of data has also been collected for more varied local diets such as the Aboriginal homelands of northern Australia (Martin et al., 1998)

Cereals, root vegetables and fruits

Studies with grasses and wheat have shown that in broad terms the majority of uranium appears to be accumulated firstly by the roots, shoot and then seeds (Jain and Aery, 1997). This observation is consistent with observations in other plants that form the basis of some phyto-remediation methods for the removal of uranium from polluted waters and in studies of sites of uranium mineralisation (Basham et al., 1989). In plants used for the purposes of phyto-remediation it has been found that accumulation into

shoots and seeds can be minimized through harvesting during specific periods of growth (Dushenkov et al., 1997).

Consideration of the effects of DU on ecosystem function at military proving grounds highlighted a paucity of data on the impact of DU on the function of non-arable plants. Meyer and McLendon (1997) carried out studies of three species of grasses, *Buchloe dactyloids* (buffalograss), *Schizachyrium scoparium* (little bluestem) and *Aristida purpurea* (purple threeawn). These species were considered to be typical of the vegetation present in the arid and semi-arid conditions of most areas where DU has been tested and deployed. The study concluded that DU was relatively nontoxic to these grasses with no observed toxicity occurring at soil concentrations below 25 000 mg/kg-U. In contrast, Jain and Aery (1997) reported that contamination of agricultural land with uranium may have a detrimental effect on the productivity of wheat. (There is a marked reduction in a number of growth parameters, that included a decrease in seedhead number (spikes), seed number and seed weight (no seed head being produced at an additive level of 625 µg/l U in irrigation water).

Table 5.2 Examples of typical concentration factors for uranium (ICRP-29, 1978; IAEA, 1982, 1994).

Concentration Factor					
Leafy Vegetables*		Root Vegetables*		Fruits*	
Minimum	Maximum	Minimum	Maximum	minimum	maximum
1.2×10^{-4}	1.0×10^{-2}	2.0×10^{-4}	3.0×10^{-2}	4.0×10^{-4}	4
Grains and Cereals**					
Minimum		Maximum			
2.0×10^{-4}		1.3×10^{-3}			

* As Bq/g wet plant/Bq/g dry soil

** As Bq/g dry plant and soil

Note: Such factors vary with soil pH, organic carbon content, uranium content etc. (ATSDR, 1999) and site specific data should be applied where possible

To facilitate the assessment of exposure from the ingestion of foodstuffs WHO and various other international and national authorities have compiled a set of typical food consumption patterns. Examples of typical patterns are given in Annex 3. It is emphasized that these data reflect average consumption and that it is important during the assessment of dietary exposure that specific local consumption practices are investigated. Such practices may include the fermentation of food and an unusually strong dietary dependence on one or two specific foods.

Animal products

Like many other heavy metals uranium may become incorporated into the human food chain. The majority of bioaccumulation studies to date have been performed on natural rather than DU. This is an important consideration as the kinetics of bioaccumulation of DU into the food chain will be subject to a lag period, the magnitude of which will be governed by the weatherability and bioavailability of metallic uranium and any mixed oxides. Thomas and Gates (1999) investigated the presence of radionuclides including uranium in the lichen-caribou-human food chain. The observed concentration of

uranium in Caribou muscle were observed to range from 1% to 16% of that observed in lichen from the same environment.

Clulow et al. (1998) investigated and determined the concentration of uranium in water, sediments and fish from lakes near the uranium mining and milling operations at Elliot Lake, Ontario and from control lakes in adjacent non-industrialized watersheds. Bio-concentration of uranium was observed to occur from water, to gut material, to bone in lake trout and white fish. A systematic relationship was not established between sediment levels and those observed in fish tissues. The authors calculated a potential intake of uranium (from fish derived from industrialized areas) of 2.3 mg/annum.

Data on the concentrations of uranium and DU in exposed or unexposed farm animals are scarce. In relation to the assessment of exposure to contamination from anthropogenic sources, such as armed conflict, deposition of mostly insoluble uranium compounds is probably most relevant where it occurs on the soil or on vegetation and is taken up by grazing animals. However, it may also be necessary to investigate the potential for uptake from plants and other sources of animal feed as these uranium oxides become weathered and enter the soil pore water.

Table 5.3 Examples of bioconcentration factors for uranium transfer into animal products (ICRP-29, 1978; IAEA, 1982). Note: Such factors vary with environmental pH, uranium content etc. (ATSDR, 1999) and site-specific data should be applied where possible).

a) uptake from water into fish tissue

Environment	Biota	Concentration Factor	Range
Bq/kg wet weight fish per Bq/l water			
Fresh water	Fish	0.5	0.3 to 50

b) soil to plant transfer factors for all types of pasture, grass, browse and forage vegetation

Concentration Factors (Bq/kg dry plant per Bq/kg dry soil)	
Minimum	Maximum
1.0×10^{-5}	0.2

c) feed to milk transfer factors, fraction of daily intake taken into milk

Transfer Coefficient	
Minimum	Maximum
7.3×10^{-5}	6.1×10^{-4}

Typical soil ingestion values for cattle are estimated to be about 500 g soil per animal per day. Assuming a body weight of 400 kg this corresponds to about 1.25 g/kg of body weight per day. Data on soil intake for sheep and pigs are extremely limited (VHI, 1997) although values of 60 g and 500 g/day for these animals, respectively, have been extrapolated on a basis of body weight. Soil ingestion by goats can be considered to be negligible, as they are very selective grazers typically concentrating on the tops of grass leaves (although ingestion of dust deposited on these leaves could be considered). The uranium content measured in tissues of cattle herds grazing in pastures next to the Rocky Flats Plant in Colorado USA were slightly higher than in other cattle, reflecting

possible contamination from this source (Smith and Black, 1975). Concentrations of uranium in muscle from cattle exposed to elevated levels of forage (440 µg/kg) were similar to controls. Elevated levels of uranium were observed in liver (4 ×), kidney (4 ×) and bone (femur 12 ×) during studies based in New Mexico (Lapham et al, 1989). These results were interpreted as indicating that in cattle the muscle does not concentrate uranium (Lapham et al., 1989).

WHO (1998c) suggests an estimated average daily meat consumption of about 150 g per person per day, although, of course, this may be modified according to local dietary habits and socio-economic factors. For example higher values could be appropriate if a critical group approach were adopted in exposure assessment (see Annex 3).

5.3.2 Drinking water

As discussed earlier in Chapter 3 drinking water can contain naturally occurring uranium over a wide range of concentrations. The potential for additional exposure to uranium through the ingestion of drinking water in environments that are affected by the presence of DU is controlled by a variety of physical, chemical and hydrogeological factors similar to those that control exposure to other sources of chemical contamination. These include the proximity of the DU contamination to potential water resources (that may be currently used or developed in the future), the nature of the water resource (e.g. piped, blended, treated supply versus private water supply; groundwater versus surface water), the local geochemical environment that may promote or inhibit the weathering and transport of DU, physical dispersion during fluid migration and the magnitude of contamination by DU. Due to the complexity of these factors, it is important to undertake a site-specific assessment of the likelihood of DU entering drinking water where such contamination is known to be present.

By necessity, such an assessment is likely to be undertaken in an iterative manner modified in terms of scale and rigour as the actual likelihood and potential impacts of exposure to DU in, or from, the affected drinking-water resource is established. Similar investigations may also be required in areas in which potentially contaminated soils and machinery may be disposed of, or stockpiled. It may also be necessary, under such circumstances, to undertake a review of which water resources are currently used by members of the local population and the potential of water development activities changing this pattern in the future.

The first step in assessing exposure to uranium through drinking water must be the chemical determination of its concentration in water consumed by individuals. This may include monitoring the point of public/household supply, or private supply, or from potential sources such as springs used during camping excursions. It is essential during the collection of samples to be aware of various practices associated with the operation of the supply, such as the amount of water typically run-off prior to consumption, and the use of private water supplies.

The WHO recommends an average water consumption of two litres/day to be used in calculating human exposure to contaminants in drinking water. In practice this consumption rate may vary depending upon age, activity level and the consumption of water in the form of beverages and soft drinks. Where exposure through drinking water is suspected as being an important route of exposure it is recommended that these factors be carefully investigated.

It is unlikely that significant amounts of the more insoluble tetravalent species of uranium will be present in potable water supplies, except where it is carried on particulate material. In such cases the amount of material likely to be ingested should be estimated by sequential filtration procedures and included in subsequent exposure calculations using the appropriate gut uptake factor.

While it is not the objective of this report to specifically review potential remedial scenarios, it should be noted that many treatment methodologies exist for the removal of uranium from water (e.g. Varani et al., 1987) although few have been used to produce potable water.

5.3.3 Soil and dust

How much soil is ingested is a deceptively simple but highly relevant question that has led to a substantial discussion in the literature (e.g. Simon, 1998). Some studies considered to be influential (e.g. Kimborough et al., 1984) have been criticized for using ultraconservative soil ingestion rates with little empirical support (Paustenbach et al., 1986; Gough, 1991). Recent research has been dominated by mass-balance studies of 'conservative tracer elements', i.e. chemical elements that are present in soil but which are not significantly absorbed by passage through the gut (van Wijnen et al., 1990; Calabrese, 1989; Davies, 1990).

The ingestion of soil and/or dust occurs both within and outside the household environment and it is important to establish, where possible, the concentration of uranium in both environments. Alternatively, a general relationship between indoor dust and outdoor soil contaminant concentrations may have to be assumed.

For example, Keenan et al. (1989) and Murphy et al. (1989) have reported the proportion of locally derived soil particles in indoor dust to be in the order of 75% to 100%. This estimate was based mainly on a study of land contamination around a series of smelters. However, some studies such as those by Franzen et al. (1988) and Steele et al. (1990) indicated that in mining communities the proportion was much less than this (typically indoor concentrations were 14% to 15% of soil concentrations). This difference was considered to be due to the surface properties and moisture content of smelter particles which allowed them to adhere readily to shoes, clothing and pets, and thus to be tracked indoors more easily than other particles.

For uranium and DU, which may be derived from a variety of sources, it is impossible to suggest one value for the proportion of outdoor-derived dust in the indoor environment. Given the large range of observations and the lack of relevant information, the recommendation of a value of 75% (Keenan et al. 1989) would seem appropriate, even though this is almost certainly cautious in many cases.

Three distinct categories of soil ingestion may be considered, and these are discussed below along with suggested quantities of ingested material. From these examples it can be clearly seen that the amount of soil and/or dust ingested varies greatly, and that it is essential that the likely magnitude of geophagic activity be assessed in potentially exposed populations. During surveys of geophagic behaviour, it is essential that great care is exercised to prevent false negative results being obtained due to cultural taboos associated with this practice (e.g. being considered to be improper or of lower social status).

1. *Inadvertent ingestion of small quantities of soil and dust.*

It is likely that all members of an exposed population will have intakes by this route although exposure is likely to be greatest for children under seven years old. Sources of soil and dust are likely to be derived from both outdoors and indoors and the relative magnitude of exposure will depend greatly on the habits and behaviour of an individual. Despite the wide number of studies considerable uncertainties still exist in data relating to this activity (e.g. Simon, 1998). This is in part due to the difficulty in the methodological use of tracers to estimate such quantities, and also the highly individualistic nature of exposure. Inadvertent soil ingestion rates for studies performed in the USA are given in Tables 5.4 and 5.5 and illustrate the uncertainties and typical values determined in such studies. It should be noted that these values are generally slightly higher than those suggested by WHO (20 mg/day, Annex 3). The use of data based on mass consumed may be inappropriate for such a dense material as DU and its oxides and correction factors accounting for differences in density may need to be applied.

Table 5.4 Examples of soil ingestion estimates for children in the USA (mg/day) derived from tracer studies (note: negative values indicate error in mass balance).

Tracer		(Davies, 1990) N = 101			(Calabrese, 1989) N = 64		
		Mean	Median	Range	Mean	Median	Range
Al	39	25	-279 to 904	153	29	-75 to 6837	
Si	82	59	-404 to 535	154	40	-53 to 5549	
Ti	246	81	-5821 to 6182	218	55	-3069 to 6707	

Table 5.5 Soil ingestion rates for adults (mg/day) derived from tracer experiments (N = 6) (Calabrese et al., 1990).

Tracer	Al	Si	Ti
Mean	77	5	377
St Dev	65	55	517
Median	57	0.5	211

2. *Occasional deliberate consumption of soil and dust.*

Most young children indulge in this type of exploratory behaviour for a relatively short time, although there is hardly any quantitative information on the amount of soil deliberately ingested during these activities. This is due in part to the difficulty in separating the occasional consumption of soil from the habitual practice of geophagy. For the group of 64 US children studied by Calabrese et al. (1991) the median soil ingestion rate ranged from 9 to 96 mg/day according to tracer, but one child (a three and a half year old girl) ingested much greater quantities (up to 13.6 g/day). Earlier estimates of the amount of soil deliberately ingested as 5g/day

(US EPA, 1984) and 10 g/day (US EPA, 1989) have generally been based purely on 'judgement'.

3. *Geophagia*.

The term geophagia refers to the persistent and purposeful consumption of soil and/or dust, often in relatively large quantities. It is typically associated with children and pregnant females who are commonly subject to nutrient deficiencies. Geophagia should be considered as being distinct from pica, which also relates to the mouthing or eating of unusual objects, and should not be considered as only occurring in rural environments. Geophagia has been studied in both the United Kingdom and North America within the wider context of pica (e.g. Cooper, 1957; Bicknell, 1975; Barltrop, 1966; Morgan et al., 1988). However as Lacey (1990) comments 'The body of literature on pica is so fragmented that it is difficult to find a precise summary of the knowns and unknowns about the condition. There is little consistency in defining pica, classifying substances ingested, identifying key characteristics of practitioners, recommending treatment or projecting outcomes'. The fragmentary nature of this information therefore makes it extremely difficult to calculate exposure of populations or individuals via this route. The situation elsewhere is even more complicated, particularly in tribal cultures where geophagy is commonly practised. For example studies by Geissler et al. (1998) indicated that a large proportion of male and female children in Kenya practise geophagy up to the age of 16, with an average soil consumption rate of 25 g/day.

5.4 Dermal contact

Exposure to uranium through dermal contact in non-occupationally exposed populations is poorly studied. This is principally due to the relatively low abundance of uranium in the natural environment compared to the concentrations encountered in the workplace.

However, some uranium compounds (e.g. uranyl nitrate, uranyl fluoride, uranium trioxide) have been demonstrated to be chemically toxic to animals through dermal exposure (e.g. Orcutt, 1949, DeRey et al., 1983, Ubios et al., 1997 and Lopez et al., 2000), whereas the radiation dose received through dermal contact is minimal (AEPI, 1995). Potential health effects associated with such exposure are discussed in more detail in Chapter 8.

In the natural environment, dermal exposure to the more soluble forms of uranium is significantly less likely than in an occupational context, as uranium from these compounds is likely to have become translocated and dispersed from soils and sediments into surface waters and groundwater. For less soluble compounds, such as uranium phosphates associated with fertilizers and uranium oxides associated with DU, exposure through dermal contact is likely to result in a greater degree of hand to mouth transfer (see ingestion of soil above).

Human exposure through dermal contact can also result in contaminants and poisons entering the systemic circulation by physical transport following traumatic damage to the skin such as can be encountered in military situations in which DU has entered the body, or perhaps through abrasion of the skin, as has been suggested by the studies of Podocniosis (Price, 1990) and endemic Kaposi's sarcoma (Ziegler, 1993). In general, dermal contact as a route of uptake of uranium into the body is considered to be unimportant.

5.5 Workplace exposure

Conditions in the workplace where humans are potentially exposed are extremely variable and depend on the prevailing 'health and safety' culture. The potential hazards also depend on the type and chemical form of uranium containing materials handled within the workplace. Six major groups of workers can be identified.

1. Those primarily involved in the mining and milling of uranium ores.
2. Those involved in nuclear fuel fabrication and reprocessing
3. Those concerned with the handling and machining of metallic uranium (see Figure-5.2) and associated compounds during its industrial processing (e.g.-armaments and chemical industries).
4. Those involved in the handling of prefabricated components made of metallic uranium and associated compounds and alloys during assembly and manufacturing of industrial components (e.g. the aircraft industry and medical/research sectors).
5. Those involved in industries in which uranium is present as a contaminant or by-product. For example, in the extraction of other ores containing elevated levels of uranium (e.g. phosphorities), in the processing and agricultural application of phosphate fertilizers, or as a worker decommissioning and scrapping military vehicles.
6. Those involved in emergency services in the aftermath of accidents or incidents involving DU (e.g. factory fires, aircraft accidents) or following a plane crash or fire in a uranium storage facility.



Figure 5.2 Workshop for the machining of uranium.

In the second case, there is the potential for contact with pure uranium or uranium compounds such as UF_6 which is extremely toxic due to the release of HF when in contact with water. To date at least one case of attempted acute poisoning (non-fatal) has been recorded in which the subject, a uranium processing worker, deliberately consumed processed uranium (Pavlakakis et al., 1996).

Occupational exposure to DU may involve exposure to a number of different physical and chemical forms of uranium. During the production process the solubility of uranium compounds changes dramatically. Uranium can exist in biologically soluble forms such as uranium hexafluoride (UF₆), uranyl nitrate (UO₂(NO₃)₂) and uranium trioxide (UO₃), and relatively insoluble forms such as uranium dioxide (UO₂) and triuranium octoxide (U₃O₈). As discussed in Chapters 8 & 9, the severity of health effects may be associated with chemical or radiological toxicity and hence the exposure limits are controlled from the standpoint of both (ACGIH, 2000; ICRP, 1991; BSS, 1996; NIOSH, 1994).

The presence of high concentrations of 'available' uranium compounds in workplaces associated with cases 3 to 6 above is much less likely, although uranium concentrations as high as 400 mg/kg may be reached in ore and dusts associated with phosphate rocks and phosphogypsum. Elevated levels of uranium may also be inadvertently encountered in fly-ash from coal combustion, zircon sands used in the ceramics industry, ores and precipitates used in the production of titanium dioxide pigments and in the metal recycling industry (van der Steen, 1999). In the context of the metal recycling industry, out of a total of 3500 events in which radioactivity was detected in shipments of metals for recycling in the USA, 44 events (1.24%) were ascribed to the presence of uranium and/or DU (Yusko, 1999). These events included the melting of a shipment of recycled zinc containing DU at the Southern Zinc plant in Georgia, USA.

A contaminating incident (involving DU chips and dust) was also reported to have occurred when hammers and chisels were used to remove DU counterweights from an aircraft. Such occurrences highlight the need for promoting awareness of the presence and handling of such materials in the workplace.

5.6 Summary

Exposure to uranium and DU may take place through a wide variety of pathways. Environmental exposure may be due to inhalation and ingestion of food, drinking water and dust/soil. In the workplace exposure is more likely to be the result of dermal contact or inhalation.

In general, it is considered that ingestion of food and drinking water dominate background exposure to uranium. However, this is only likely to occur in the case of DU if it has become well mixed in the food chain and/or contaminated a source of drinking water. This is unlikely to occur in the short term and consequently exposure to DU will be dominated by ingestion and inhalation of any dust derived from the use of DU (exposure by handling metallic DU, for example by picking up penetrators from the battlefield, is dealt with in more detail in Chapter 9).

Data regarding bio-uptake of uranium into plants and animals indicates that bioaccumulation factors, while not being high may be in some cases significant over the longer term, particularly where local consumption patterns indicate a preference for foodstuffs shown to potentially bioaccumulate uranium (i.e. the kidneys of cattle).

In the absence of specific data on the solubility and bioavailability of DU related compounds and mixtures extrapolation from the behaviour of uranium may not always be valid. Similarly natural variations in parameters governing the bioavailability and mobility of uranium and DU are often highly variable indicating the need to collect site specific as well as material specific data.

