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Scientific Criteria Document for the Development of the Canadian Water Quality Guidelines for the Protection of Aquatic Life

Uranium

PN 1451

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NOTE TO READERS

The Canadian Council of Ministers of the Environment (CCME) is the major intergovernmental forum in Canada for discussion and joint action on environmental issues of national, international and global concern. The 14 member governments work as partners in developing nationally consistent environmental standards, practices and legislation.

This document provides the background information and rationale for the development of the Canadian Water Quality Guidelines for uranium. They were developed by the National Guidelines and Standards Office of Environment Canada. For additional scientific information regarding these water quality guidelines, please contact:

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AF	assessment or application factor
BAF	bioaccumulation factor
BC	British Columbia
BCF	bioconcentration factor
BEC	bound effect concentration
BLM	biotic ligand model
CANDU	Canadian Deuterium Uranium
CAS RN	Chemical Abstracts Service Registry Number
CCME	Canadian Council of Ministers of the Environment
CDF	cumulative distribution function
CWQG	Canadian water quality guideline
DL	detection limit
DNA	deoxyribonucleic acid
DOM	dissolved organic matter
DU	depleted uranium
dw	dry weight
EC_X	effective concentration – the concentration that causes the specified percentage of the population (represented in the X) of the experimental biota to show an observed effect. The effect may be immobilization, changes in reproductive potential, growth, or some other ecologically relevant endpoint
FIAM	free ion activity model
h	hour, e.g., 96 h is 96 hours
ICP	inductively coupled plasma
ICP-AES	inductively coupled plasma atomic emission spectrometry
ICP-MS	inductively coupled plasma mass spectrometry
ICP-OES	inductively coupled plasma optical emission spectrometry
IC_X	inhibitory concentration – the concentration of inhibitor that causes the specified percentage (X) of inhibition in the target (e.g., molecule, enzyme, cell, micro-organism)
K_d	partition or distribution co-efficient
LC_X	lethal concentration – the concentration that is lethal to the specified (X) percentage of the experimental biota
LOEC	lowest-observed-effect concentration
MATC	maximum acceptable toxicant concentration, calculated as the geometric mean of the LOEC and the NOEC
MDEC	minimum-detectable-effect concentration – the concentration at which the response becomes significantly ($p \leq 0.05$) lower than the control
MXR	multixenobiotic resistance protein
NGR	National Geochemical Reconnaissance
NGSO	National Guidelines and Standards Office
NOEC	no-observed-effect concentration
NOM	natural organic matter
NRG	no recommended guideline
ON	Ontario

PBiT	persistence (P), bioaccumulation (B), inherent toxicity (iT); these criteria are commonly used in hazard identification
pH	the negative log of the concentration of the hydrogen cation, $-\log[H^+]$
PHREEQCI	PH (pH), RE (redox), EQ (equilibrium), C (program written in C) and I refers to the interactive version of PHREEQC; a program used to calculate metal speciation in water
PNEC	predicted no-effect concentration
ppb	parts per billion
PSL	Priority Substances List
RNA	ribonucleic acid
SK	Saskatchewan
SSD	species sensitivity distribution
TTD	time to deterioration
U	uranium; unless otherwise specified, U represents the total amount of U present
URP	Uranium Reconnaissance Program
WHAM	Windermere humic aqueous model – a program used to calculate metal speciation in water
ww	wet weight

EXECUTIVE SUMMARY

Uranium (CAS RN 7440-61-1, atomic mass 238.03 g/mol) is a naturally occurring radioactive metal. It can occur in three isotopic forms, each with its own distinct radioactive properties. In comparison with its decay products, however, uranium itself has relatively low radioactivity (ATSDR 1999). It has a very high density. In addition to its radioactive properties, uranium also has chemical properties as an elemental metal. Not found in elemental form in nature, uranium exists as an important component of about 155 minerals, including oxides (which include pitchblende and uraninite), phosphates, carbonates, vanadates, silicates, arsenates and molybdates (Clark et al. 1997). Although uranium can exist in four different oxidation states, the uranyl ion UO_2^{2+} with the uranium in the +6 oxidation state (i.e., VI) is the most common in oxic waters (Choppin and Stout 1989; Clark et al. 1997; Langmuir 1978).

Methods used to measure uranium in environmental samples can detect and quantify uranium based on either its properties as an element or as a radioactive compound. Reported detection limits vary between methods and sample preparation, but generally are in the low parts per billion (ppb, or μg/L) for total uranium. Like other metals, uranium can exist as several different physical-chemical forms (or species) in water, including the free uranyl ion UO_2^{2+} or complexed forms such as $UO_2(CO_3)_2^{2-}$. Methods for measuring the concentrations of these species (as opposed to total uranium) are limited and technically challenging. More commonly, speciation is modelled using geochemical codes based on thermodynamic data, as in Markich et al. (2000).

Several areas in Canada, including parts of Saskatchewan and Ontario, contain naturally high concentrations of uranium ore deposits, which has led to past and present mining operations (Environment Canada and Health Canada 2003; Giancola 2003). The main use for mined uranium is as a fuel. Several processes are involved in the manufacture of fuel products from uranium ore, and potential release points into the environment include uranium mines and mills, uranium refining and conversion facilities and fuel fabrication facilities, power reactors and associated waste management facilities, research reactors (fission and activation products), and stand-alone waste management facilities (Environment Canada and Health Canada 2003).

As a naturally occurring element, the presence of uranium in water does not necessarily indicate pollution. As a result of geochemical processes, some areas naturally contain elevated concentrations of uranium in underlying rock. Superimposed on the mineral composition of the environment are abiotic processes that are crucial in determining the spatial and temporal variability in natural background. These processes include weathering, climate, soil type, pH, dilution (e.g., due to rainfall, snowmelt, other seasonal variations), and redox potential (Natural Resources Canada 2004). Based on 95th percentiles, background concentrations of uranium have been estimated to be 0.35 μg/L in northern Saskatchewan and 0.28 μg/L near Elliot Lake in Ontario (Environment Canada and Health Canada 2003). Concentrations of uranium in water bodies that may have been impacted by uranium facilities range from 0.11 to 1061 μg/L (Environment Canada and Health Canada 2003; Swanson 1985).

Similar to many other metals of potential concern, the environmental fate and behaviour of uranium are dependent on abiotic conditions, such as pH, hardness, alkalinity and natural organic matter. These abiotic factors influence the bioavailability, toxicity and mobility of uranium by altering the speciation, or physical-chemical forms, of uranium in aquatic systems. Although the speciation of uranium in

water is complex, modelling results show that conditions that favour the formation of the free ion UO_2^{2+} include low pH and low concentrations of natural organic matter, and possibly low alkalinity (Gilbin et al. 2003; Markich et al. 2000; Riethmuller et al. 2001). Uranium tends to partition into sediments (ATSDR 1999), as evidenced by high partition or distribution coefficient (K_d) values between 0.36 and 3.2 x 10^3 L/kg wet weight (ww) (Swanson 1985). Sediments have a cation exchange capacity, which allows reversible surface binding (adsorption) of trace elements (such as uranium) at exchange sites on the surface (Manahan 1994). Although the nature of the adsorption differs between mineral types, adsorption is highest at near-neutral pH values (Lenhart and Honeyman 1999; Sylwester et al. 2000; Zuyi et al. 2000). Adding to the burgeoning field of biogeochemistry, Lovely et al. (1991) showed that microbes can enhance the reduction of dissolved uranium in anaerobic sediments; microbial reduction is therefore an important pathway in uranium fate and behaviour, and other studies have shown similar interactions between microbes and uranium (Wang and Chen 2006).

In addition to altering the environmental fate and behaviour of uranium, water chemistry conditions (such as pH, hardness, alkalinity and natural organic matter) can influence the toxicity of uranium to aquatic organisms. For many inorganic metals, the most bioavailable form is the free ion, which is often the most toxic form, and so understanding the chemical conditions that lead to the free ion form facilitates the prediction of toxicity. For uranium in particular, the majority of evidence suggests that the free ion UO_2^{2+} is the most toxic form, although some studies suggest exceptions at different pH values (Fortin et al. 2004; Fournier et al. 2004; Gilbin et al. 2003; Markich et al. 2000). Potential explanations that may explain deviations with pH include competition for uptake between the hydrogen ion and the uranyl ion (Franklin et al. 2000) and the contribution of UO_2OH^+ to observed toxicity (Markich et al. 2000). Increases in hardness, defined as the sum of calcium (Ca^{2+}) and magnesium (Mg^{2+}), in some cases reduced the toxicity of uranium but in other studies had no effect on toxicity; some of these results are difficult to interpret, because hardness co-varied with alkalinity. There were very few studies that documented the effects of alkalinity and natural organic matter. No quantitative relationship could be established between any of these factors and the toxicity of uranium, so no modifications or adjustments were made to the data.

Although bioaccumulation is an important consideration for many environmental contaminants, the interpretation of quantities such as the bioconcentration factor is problematic for metals (McGeer et al. 2003). Based on evidence from metals such as zinc, cadmium, copper, lead, nickel and silver, experts have stated that metrics such as bioconcentration factors are not reliable indicators of hazard. Although there are currently not enough data to make the same conclusion for uranium, it is reasonable to assume that a similar conclusion could apply. Despite this lack of certainty, there is evidence that uranium does not biomagnify in food webs, and no evidence that it does (Environment Canada and Health Canada 2003; Simon and Garnier-Laplace 2004, 2005; Swanson 1985).

Uranium is assumed to be a non-essential element in aquatic organisms, as there have been no reports of a metabolic function for uranium in aquatic organisms. There are few studies on a mode of action of uranium toxicity in fish; one study investigated the histological effects of food-borne uranium on lake whitefish (*Coregonus clupeaformis*), showing that kidney damage was the primary effect (Cooley et al. 2000). Some studies have suggested the gill as the site of action in the fish (Bywater et al. 1991) and the Asiatic clam (*Corbicula fluminea*) (Tran et al. 2004). However, Simon and Garnier-Laplace (2004, 2005) found that the likely mode of action for uranium was through the digestive gland in both the Asiatic clam and the crayfish *Orconectes limnosus*.

The short- and long-term freshwater Canadian water quality guidelines (CWQGs) for uranium for the protection of aquatic life were developed based on the Canadian Council of Ministers of the Environment (CCME 2007) protocol using the statistical or Type A approach, as sufficient data were available. Insufficient data were available to derive a short- or long-term marine water quality guideline for uranium.

Canadian Water Quality Guidelines for the Protection of Aquatic Life

	Long-term exposure guideline (µg U/L)	Short-term exposure guideline (µg U/L)
Freshwater	15	33
Marine	NRG	NRG

NRG = no recommended guideline

SOMMAIRE

L'uranium (masse atomique de 238,03 g/mol; nº de registre CAS : 7440-61-1) est un métal radioactif d'origine naturelle. Il possède trois formes isotopiques qui possèdent des propriétés radioactives distinctes. Comparativement à ses produits de désintégration, l'uranium présente toutefois une radioactivité relativement faible (ATSDR, 1999). Ce métal élémentaire, dont la masse volumique est très élevée, possède des propriétés chimiques particulières en plus de ses propriétés radioactives. L'uranium élémentaire n'existe pas dans la nature, mais il constitue un important composant de quelque 155 minéraux, dont des oxydes (y compris la pechblende et l'uraninite), des phosphates, des carbonates, des vanadates, des silicates, des arséniates et des molybdates (Clark et coll., 1997). Même si l'uranium peut exister sous quatre états d'oxydation distincts, l'ion uranyle, UO_2^{2+}, avec l'uranium sous l'état d'oxydation +6 (c.-à-d. VI) est le plus commun dans les eaux oxygénées, (Choppin et Stout, 1989; Clark et coll., 1997; Langmuir, 1978).

Les méthodes de mesure de la concentration d'uranium dans des échantillons prélevés dans l'environnement permettent de détecter et de doser l'uranium en se basant sur ses propriétés d'élément ou de composé radioactif. Les valeurs de la limite de détection signalées varient en fonction de la nature des méthodes et de la préparation des échantillons, mais elles se situent habituellement dans la plage inférieure des parties par milliard (µg/L ou ppb), dans le cas de l'uranium total. L'uranium, tout comme d'autres métaux, peut exister sous plusieurs formes (ou espèces) physico-chimiques distinctes dans l'eau, y compris celles de l'ion uranyle libre UO_2^{2+} et du complexe $UO_2(CO_3)_2^{2-}$. Il existe peu de méthodes de mesure de la concentration de ces espèces (par contraste à l'uranium total) et leur élaboration pose de sérieux problèmes techniques. Dans la plupart des cas, la détermination des espèces chimiques présentes (la spéciation) est modélisée en utilisant des codes géochimiques basés sur des données thermodynamiques, comme dans l'étude de Markich et de ses collaborateurs (2000).

Plusieurs régions du Canada, notamment certaines zones de la Saskatchewan et de l'Ontario, recèlent de nombreux gisements de minerai à forte teneur naturelle en uranium dont l'exploitation minière a été réalisée par le passé ou est encore en cours (Environnement Canada et Santé Canada, 2003; Giancola, 2003). L'uranium extrait des mines est principalement utilisé comme combustible nucléaire. Plusieurs procédés sont nécessaires à la préparation de ces combustibles à partir de minerai uranifère et les points de rejet possibles dans l'environnement comprennent les mines et les usines de traitement d'uranium, les installations de raffinage et de conversion d'uranium et celles de fabrication de combustibles, les réacteurs nucléaires de puissance et les installations connexes de gestion de déchets, les réacteurs de recherche (produits de fission et d'activation) et installations autonomes de gestions des déchets (Environnement Canada et Santé Canada, 2003).

L'uranium étant un élément d'origine naturelle, sa présence dans l'eau n'est pas nécessairement un signe de pollution. Des processus géochimiques peuvent entraîner des concentrations naturelles d'uranium élevées dans la roche sous-jacente de certaines régions. En plus de la composition minérale du milieu naturel, il faut tenir compte de processus abiotiques qui sont essentiels à la détermination de la variabilité spatiale et temporelle du rayonnement de fond. Les processus en question comprennent l'altération atmosphérique, le climat, le type de sol, le pH, la dilution (p. ex. celle due aux précipitations, à la fonte des neiges ou à d'autres fluctuations saisonnières) et le potentiel d'oxydoréduction (Ressources naturelles Canada, 2004). Selon des estimations basées sur les valeurs du 95^{e} percentile, les concentrations de fond d'uranium seraient de 0,35 µg/L dans le nord de la Saskatchewan et de 0,28 µg/L à proximité d'Elliot Lake, en Ontario (Environnement Canada et Santé

Canada, 2003). Les concentrations d'uranium, dans des plans d'eau ayant pu être touchés par les activités d'installations de traitement ou d'utilisation d'uranium, se situent dans une plage de 0,11 à 1061 µg/L (Environnement Canada et Santé Canada, 2003; Swanson, 1985).

Comme dans le cas de nombreux autres métaux présentant des préoccupations potentielles, le devenir et le comportement de l'uranium dans l'environnement dépendent de conditions abiotiques telles que le pH, la dureté et l'alcalinité de l'eau, et la présence de matière organique naturelle. Ces facteurs abiotiques influent sur la biodisponibilité, la toxicité et la mobilité de l'uranium, car ils peuvent modifier la spéciation, ou les formes physico-chimiques, de l'uranium présent dans des systèmes aquatiques. Bien que la spéciation de l'uranium dans l'eau constitue un processus complexe, les résultats de modélisation indiquent que les conditions qui favorisent la formation de l'ion libre UO_2^{2+} comprennent entre autres un bas pH et de faibles concentrations de matière organique naturelle, et possiblement, une faible alcalinité (Gilbin et coll., 2003; Markich et coll., 2000; Riethmuller et coll., 2001). L'uranium a tendance à se lier et à se retrouver dans les sédiments (ATSDR, 1999), comme le démontrent les valeurs élevées de K_d qui se situent entre 0,36 et 3,2 x 10^3 L/kg, en p/p (Swanson, 1985). Les sédiments possèdent une capacité d'échange cationique qui permet une fixation en surface (adsorption), réversible, des éléments traces comme l'uranium, sur les sites d'échange à la surface des particules (Manahan, 1994). La nature du processus d'adsorption varie en fonction de celle des minéraux, mais il a été établi que l'adsorption est maximale à des valeurs de pH quasi-neutres (Lenhart et Honeyman, 1999; Sylwester et coll., 2000; Zuyi et coll., 2000). Parmi les progrès réalisés dans le domaine florissant de la biogéochimie, mentionnons le fait que Lovely et ses collaborateurs (1991) ont démontré que des microorganismes peuvent accroître la réduction de l'uranium dissous dans des sédiments anaérobies. La réduction microbienne constitue donc une voie importante au chapitre du sort et du comportement de l'uranium, ce que confirment les résultats d'autres études, qui indiquent qu'il existe des interactions semblables entre des microorganismes et l'uranium (Wang et Chen, 2006).

En plus de modifier le sort et le comportement de l'uranium dans l'environnement, les propriétés chimiques de l'eau (comme le pH, la dureté, l'alcalinité et la présence de matière organique naturelle) peuvent aussi influer sur la toxicité de l'uranium pour des organismes aquatiques. Dans le cas de nombreux métaux inorganiques, la forme la plus biodisponible est celle de l'ion libre, lequel constitue souvent la forme la plus toxique; il est donc important de bien comprendre la nature des conditions chimiques qui favorisent la formation d'ions libres, car cette information facilite les prévisions relatives à la toxicité. Dans le cas particulier de l'uranium, la plupart des résultats semblent indiquer que l'ion libre UO_2^{2+} constitue la forme la plus toxique, et ce, même si les résultats de certaines études laissent croire qu'il existe des exceptions à différentes valeurs de pH (Fortin et coll., 2004; Fournier et coll., 2004; Gilbin et coll., 2003; Markich et coll., 2000). Parmi les explications possibles de ces écarts en fonction du pH, mentionnons l'absorption compétitive de l'ion hydrogène et de l'ion uranyle (Franklin et coll., 2000) et la contribution de l'ion UO_2OH^+ à la toxicité observée (Markich et coll., 2000). L'augmentation de la dureté, définie comme la somme des concentrations des ions calcium (Ca^{2+}) et magnésium (Mg^{2+}), a entraîné une réduction de la toxicité de l'uranium dans certains cas particuliers, mais selon les résultats d'autres études, elle n'influe pas sur la toxicité; l'interprétation de certains des résultats est complexe, car la variation de la dureté est aussi fonction de celle de l'alcalinité. Il existe très peu d'études dans lesquelles sont mentionnés les effets de l'alcalinité et de la matière organique naturelle. Aucune relation quantitative n'a pu être établie entre ces divers paramètres et la toxicité de l'uranium, ce qui justifie le fait qu'aucun traitement ou aucune modification des données n'ait été effectué.

Bien que la bioaccumulation compte parmi les facteurs importants dont il faut tenir compte dans le cadre d'études sur de nombreux contaminants de l'environnement, l'interprétation de valeurs comme le facteur de bioconcentration constitue un problème dans le cas des métaux (McGeer et coll., 2003). D'après des données obtenues pour des métaux comme zinc, le cadmium, le cuivre, le plomb, le nickel et l'argent, les spécialistes ont établi que des paramètres tels que les facteurs de bioconcentration ne constituent pas des indicateurs de risque fiables. Les données actuellement disponibles ne sont pas suffisantes pour tirer la même conclusion dans le cas de l'uranium, mais il semble justifié de supposer qu'une conclusion similaire pourrait tout de même être formulée. Malgré cette incertitude, il existe des données qui prouvent qu'il n'y a pas biomagnification de l'uranium dans la chaîne alimentaire, et aucune donnée prouvant qu'il y a biomagnification (Environnement Canada et Santé Canada, 2003; Simon et Garnier-Laplace, 2004; Simon et Garnier-Laplace, 2005; Swanson, 1985).

Selon les hypothèses acceptées, l'uranium constitue un élément non essentiel pour les organismes aquatiques, car aucune étude ne signale que l'uranium possède une fonction métabolique dans ces derniers. Il existe peu d'études portant sur le mode d'action de la toxicité de l'uranium chez le poisson; dans une étude particulière, on a examiné les effets histologiques de l'uranium présent dans les aliments sur le grand corégone (*Coregonus clupeaformis*) et les résultats indiquent que les plus importants effets sont des dommages subis par les reins (Cooley et coll., 2000). Les résultats de certaines études laissent croire que les branchies constituent les sites d'action, chez le poisson (Bywater et coll., 1991) et chez la petite palourde asiatique (*Corbicula fluminea*) (Tran et coll., 2004). Toutefois, les recherches de Simon et Garnier-Laplace (2004; 2005) ont établi que le mode d'action le plus probable de l'uranium serait par le biais de la glande digestive, chez la palourde asiatique tout comme chez l'écrevisse *Orconectes limnosus*.

Les valeurs établies dans les Recommandations pour la qualité des eaux au Canada (RQEC) en vue de protéger la vie aquatique, pour l'exposition à l'uranium en eau douce, à court terme et à long terme, ont été déterminées en se basant sur le protocole du CCME (CCME, 2007), au moyen de l'approche statistique, ou approche de Type A, car les données disponibles étaient suffisantes pour justifier son emploi. Toutefois, les données n'étaient pas suffisantes pour calculer des valeurs recommandées du type RQEC pour l'exposition à l'uranium dans l'eau de mer, à court terme ou à long terme.

Valeurs des Recommandations pour la qualité des eaux au Canada en vue de protéger la vie aquatique

	Exposition à long terme (µg U/L)	**Exposition à court terme (µg U/L)**
Eau douce	15	33
Eau de mer	AR	AR

AR = aucune recommandation

1.0 INTRODUCTION

Canada has one of the world's richest deposits of uranium. While uranium is an important natural resource, mining and milling activities can redistribute it, and may cause concentrations in ambient water to exceed background concentrations, which in turn could lead to adverse environmental effects.

Canadian water quality guidelines (CWQGs) compile, synthesize and interpret aquatic toxicity data, providing an important tool in the evaluation of ambient water quality. CWQGs are numerical or narrative thresholds set to protect all forms of aquatic life over an indefinite exposure to substances of potential concern. Where ambient concentrations are below the CWQG, adverse affects are not expected to occur in the aquatic environment. The Water Quality Task Group of the Canadian Council of the Ministers of the Environment (CCME) is charged with overseeing the development of Canadian water quality guidelines for the protection of aquatic life. Recently, the protocol used to develop these guidelines was revised (CCME 2007). The goals of the revised protocol include (i) accounting for the unique properties of contaminants which influence their toxicity; and (ii) incorporating the species sensitivity distribution (SSD) method, which uses all available toxicity data (provided these data pass quality control criteria) in a more flexible approach. While the SSD approach has been used in several jurisdictions for water quality guideline development, it is a new concept in the derivation of CWQG, and consequently the uranium CWQG is one of the first where it has been applied.

The structure of the supporting document for uranium has been built to accommodate the changes in the protocol for guideline derivation. All of the customary components of scientific supporting documents have been included (physical and chemical properties, production and uses, environmental fate and behaviour, environmental concentrations, and toxicity data). In addition, new cornerstones of the protocol, such as bioavailability and toxicity modifying factors have been given special attention.

2.0 PHYSICAL AND CHEMICAL PROPERTIES

2.1 Identity

Uranium (U, CAS RN 7440-61-1) is a heavy, naturally occurring element (atomic number = 92, atomic weight = 238.029 g/mol) and is a member of the actinide series on the periodic table. Uranium is radioactive, and it decays by emitting an alpha (α) particle (2 neutrons and 2 protons) from its nucleus (Harley 1996). Although many of the decay products of uranium have a high specific activity associated with them, uranium itself has a relatively low radioactivity (0.67 µCi for a one gram sample) (ATSDR 1999).

In the environment, uranium can exist in three isotopic forms, each with characteristic relative abundance and radioactivity (Table 1). The isotope of interest for nuclear reactions is ^{235}U, because it can undergo fission. An enrichment process is used to increase the usefulness of uranium as a nuclear reactant, resulting in enriched uranium (^{235}U isotopic abundance is increased to 2–4%). Uranium with less ^{235}U than natural abundance (0.72%) is depleted uranium, or DU.

Not found in elemental form in nature, uranium exists as an important component of about 155 minerals, including oxides (which include pitchblende and uraninite), phosphate, carbonates,

vanadates, silicates, arsenates and molybdates (Clark et al. 1997). The economically important uranium-containing minerals are uraninite and pitchblende, with compositions of UO_2 and UO_3. Uranium hexafluoride is anthropogenically produced (ATSDR 1999). Due to the electron configuration properties of the actinide series (smaller energy differences between neighbouring electron orbitals), uranium can exist in more oxidation states than other metal contaminants of concern (Clark et al. 1997). Of the four possible uranium oxidation states (III, IV, V and VI), IV and VI are generally agreed to be the most common (Choppin and Stout 1989; Clark et al. 1997), although Langmuir (1978) suggests that U(V) as UO_2^{+} can also have appreciable thermodynamic stability in reduced waters with pH < 7. In oxic natural waters, uranium is present mainly in the U(VI) state (oxidized), either as the free cation UO_2^{2+} or complexed to a ligand to form molecules such as $UO_2(HPO_4)_2^{2-}$ and $UO_2(CO_3)_3^{4-}$ (Choppin and Stout 1989; Langmuir 1978).

In environmental toxicity testing, uranyl sulphate ($UO_2SO_4 \cdot 3H_2O$) and uranyl nitrate ($UO_2(NO_3)_2 \cdot 6H_2O$) are the most common uranium chemicals, although uranyl acetate ($UO_2(C_2H_3O_2)_2 \cdot 2H_2O$) has also been used. As both compounds are expected to ionize in water to form the hexavalent uranyl ion (UO_2^{2+}), which is the oxidation state most common in oxygenated aquatic systems (Langmuir 1978; Poston et al. 1984), these compounds are expected to be suitable surrogates for naturally occurring uranium. The physical and chemical properties of several uranium compounds are summarized in Table 2.

2.2 Detection methods for environmental samples

Methods used to measure uranium in environmental samples (including air, water and soil) can detect and quantify uranium based on either its properties as a chemical or as a radioactive compound. Common sample preparations include concentration, solvent extraction, acid digestion and filtration (ATSDR 1999).

Detection limits and recovery were used to evaluate the analytical precision and validity of toxicity studies. Surveying the available information, it is apparent that most reported detection limits are in the low parts per billion (ppb) range (Table 3). One ppb is equivalent to 1 µg/L, or 0.001 mg/L. Consistent with this observation, the New South Wales Environment Protection Authority stated that the current practical quantitation limit for uranium is 0.01 µg/L (parts per billion) in freshwater and 0.3 µg/L in salt water (ANZECC and ARMCANZ 2000 and references therein). Bywater et al. (1991) reported that measured values of uranium in one experiment were approximately 67% of nominal concentrations, despite attention to quality control, emphasizing the importance of measuring exposure concentrations in toxicological testing.

Speciation of metals, including uranium, in water is often related to the observed toxicity. However, most detection methods measure the total amount of uranium in a sample, and provide little or no information on its speciation in water. Techniques such as time-resolved laser-induced fluorescence spectroscopy (Moulin et al. 1995) or electrospray ionization-mass spectroscopy (Gresham et al. 2003) can be used to empirically investigate speciation. Methods used in direct speciation of actinides (e.g., solvent extraction, ion-exchange chromatography, precipitation and sorption) may also be applicable to uranium (Choppin and Stout 1989). More commonly, speciation is modelled using geochemical codes based on thermodynamic data, as in Markich et al. (2000). Outputs of different geochemical codes, however, can differ; in one example, PHREEQCI (a speciation model; PH [pH], RE [redox], EQ [equilibrium], C [program language] I [interactive version]) predicted $UO_2(HPO_4)_2^{2-}$ as the major

species, in contrast to another speciation model WHAM (Windermere humic aqueous model), which predicted UO_2^{2+} as the major species (Unsworth et al. 2002). This sensitivity analysis used the thermodynamic data provided with the models, and it is assumed that the models were run under similar chemistry conditions. Speciation is of direct importance when considering the toxicity of uranium to aquatic life, as certain species are more toxic than others and are more prevalent in waters of differing characteristics, such as hardness. Speciation can also help to explain results that first appear to be inconsistent with other studies. Further discussion of speciation effects on toxicity is discussed in section 5.2, "Aquatic chemistry and speciation."

3.0 PRODUCTION AND USES

3.1 Mining, milling, refining and conversion in Canada

Several areas in Canada, including parts of Saskatchewan and Ontario, contain naturally high concentrations of uranium ore deposits, which has led to past and present mining operations (Table 4) (Environment Canada and Health Canada 2003; Giancola 2003). In comparison with other countries, Canada has a high concentration of uranium resources (Clark et al. 1997). Production of uranium ore (as U_3O_8) in Canada burgeoned in the 1950s from 500 to 15 900 short tons,[1] which is approximately 450 to 14 420 metric tonnes (Bailar et al. 1973). In comparison, production from the four operating mines in Saskatchewan alone was 9001 metric tonnes in 2008 (CAMECO Corporation 2009; AREVA Resources Canada 2009).

Several processes are involved in the manufacture of nuclear fuel products from uranium ore, and potential release points into the environment include uranium mines and mills, uranium refining and conversion facilities and fuel fabrication facilities, power reactors and associated waste management facilities, and research reactors (fission and activation products) and stand-alone waste management facilities (Environment Canada and Health Canada 2003). Milling generally occurs near the mines, whereas refining and conversion occur off-site. Refining, conversion and fuel fabrication facilities are located in Ontario. A detailed review and description of these facilities is provided in a Priority Substances List (PSL) report (Environment Canada and Health Canada 2003).

Knowledge of the location of uranium deposits, where background levels of uranium would be naturally high, as well as the activities that release anthropogenic sources of uranium into the environment, are important in guideline derivation. This information provides a foundation for determination of natural and acceptable levels of uranium, especially in uranium-rich areas.

3.2 Uranium products and end uses

Fuel is the main use of mined uranium. In Canada, refining and conversion operations produce UO_2, which is used as a fuel in the CANDU (Canadian Deuterium Uranium) reactors; these reactors do not require enrichment of uranium. Refining and conversion operations in Canada also produce UF_6, which

[1] 1 short ton = 907.185 kg. To compare, 1 metric tonne = 1000 kg

is the form favoured for enrichment processes, for export. Enrichment does not occur in Canada. Major depleted uranium uses in the United States include use in armour-piercing ammunition, as a counterweight (e.g., in helicopter blades and airplane control surfaces), in military applications (e.g., ammunition manufacturing and in military shielding on army tanks), and in radiation shielding. The production of high-energy x-rays uses uranium metal as x-ray targets (ATSDR 1999; Bleise et al. 2003; Lide 2002). Small amounts of uranium are used in various other industries and household products[2] (ATSDR 1999; Lide 2002). Historically, uranium has been used in nuclear weapons (Whicker and Schultz 1982a). Uranium is also present as a contaminant in phosphate fertilizers (ATSDR 1999; Chou and Uthe 1995; Federal-Provincial-Territorial Committee on Drinking Water 2001).

3.3 Other contaminants and hazards from uranium operations

In some cases, uranium facilities release contaminants other than uranium into the environment; any resulting toxicity of the mixture may be only partially attributable to uranium itself. Other substances may include inorganic contaminants (including metals and radioactive compounds), organic contaminants, and salinity and heat stressors (Environment Canada and Health Canada 2003). See section 4.0, "Sources and Pathways into the Environment," for more information.

4.0 SOURCES AND PATHWAYS INTO THE ENVIRONMENT

Uranium mill tailings can be routes of exposure in both aquatic and terrestrial systems. Tailings deposited on dry land, if not capped with a clean cover and re-vegetated, can be lifted as dust particles, allowing for exposure through inhalation, and subsequently be deposited or washed out by precipitation into surface water bodies. They can also contaminate groundwater sources, linking into the aquatic ecosystem. Treated uranium mill effluent is another source of possible uranium contamination; however, the effluent and the stack emissions are highly monitored and regulated. Uranium is also released into the atmosphere from uranium refining and conversion plants in both soluble and insoluble forms. The effluent of these plants is discharged into the aquatic environment; however, it has not been identified as a source of uranium (Environment Canada and Health Canada 2003). The impact of these effluents on aquatic ecosystems is small; still, the consequence of a large-scale accidental discharge of uranium from an operating nuclear reactor or storage facility, though extremely unlikely, must be considered (Ahier and Tracy 1995.

[2] All specified uses of uranium are reported in documents from the United States; these uses may also occur in Canada. Uranium dioxide is used in incandescent bulb filaments used in photography and movie projectors; uranium nitrate is used in photography for toning; ammonium diuranate is used as colouring in glass and glaze; uranium carbide is used as a catalyst in the production of synthetic ammonia; unspecified uranium compounds are used as stains and dyes (leather and wood industries) and as mordants (silk and wood industries) (ATSDR 1999; Lide 2002).

4.1 Natural background

As a naturally occurring element, the presence of uranium in water does not necessarily indicate pollution where, by definition, pollution is a contamination that results in adverse biological effects. As a result of geochemical processes, some areas naturally contain elevated concentrations of uranium in underlying rock. Superimposed on the mineral composition of the environment are abiotic processes that are crucial in determining the spatial and temporal variability in natural background. These processes include weathering, climate, soil type, pH, redox potential and dilution (e.g., due to rainfall, snowmelt, other seasonal variations) (Natural Resources Canada 2004).

Anthropogenic activities, such as mining and related industries, can release uranium deposits within the earth to surface environments, resulting in concentrations of uranium that exceed natural background concentrations. In these cases, statistics-based methodologies and comparisons with non-impacted environments may be used to differentiate anthropogenic contributions of uranium from natural background. Using one common approach, based on 95th percentiles, background concentrations of uranium have been estimated to be 0.35 µg/L in northern Saskatchewan and 0.28 µg/L near Elliot Lake in Ontario (Environment Canada and Health Canada 2003), and these areas are naturally high in background levels of uranium.

As of January 2007, the Geological Survey of Canada, as part of the National Geochemical Reconnaissance (NGR) and its precursor Uranium Reconnaissance Programme (URP), has 36 years of data concerning uranium levels in lakes and stream water across Canada (2007 personal communication from RG Garrett, Geological Survey of Canada, Natural Resources Canada, Uranium in Canadian Fresh Waters; unreferenced). These data have been summarized in Table 5 and Figure 1, which indicate that lake waters have the highest uranium concentrations (> 10 µg U/L) in Saskatchewan, Manitoba, Nunavut and Ontario, where the areas are characterized by uraniferous rocks (RG Garrett, pers. comm. 2007). The lowest lake levels were found in British Columbia (BC), Yukon, northern Alberta, New Brunswick, and Newfoundland and Labrador, where uraniferous rocks are found less frequently (RG Garrett, pers. comm. 2007). Stream levels of uranium across Canada are highest (≥ 100 µg U/L) in BC and Yukon, and the rocks in the areas sampled are known to have high levels of uranium. The lowest stream levels were found in Ontario, New Brunswick, and Newfoundland and Labrador (RG Garrett, pers. comm. 2007). In Quebec, stream levels range from less than detection (< 0.0009 µg /L) to a maximum of 3.3 µg/L (2009 personal communication from I Guay, Ministère du Développement durable, de l'Environnement et des Parcs du Québec; unreferenced).

Of the water concentrations of uranium recorded across Canada, 60% of the lake data and 40% of the stream data were below the detection limit—most frequently 0.05 µg U/L, the highest detection limit used (RG Garrett, pers. comm. 2007). Natural background levels across the country range from < 0.05 µg U/L to > 100 µg U/L. About 75% of the data indicated concentrations lower than about 1 µg U/L, with concentrations higher than this occurring in areas with uraniferous geological conditions or possible anthropogenic contamination (RG Garrett, pers. comm. 2007).

The natural background concentration of naturally occurring substances is a very site-specific matter. High levels of such a substance, if toxic, or low levels, if essential, will lead to specific, locally adapted ecological communities, which may respond differently to anthropogenic releases of this substance when compared to non-adapted communities. This aspect cannot be incorporated into a nationally applicable guideline value. Therefore, in some situations, such as when the recommended national guideline value falls below (or outside) the natural background concentration, it may be necessary or

advantageous to derive a site-specific guideline (or objective). It should also be noted that natural background levels may vary seasonally, allowing for more than one value, or a range of values, for the concentration of uranium in water at any given site. Under the federal Canadian Environmental Sustainability Indicators (CESI) initiative, a framework for estimating natural background was developed (Stantec 2008; Tri-Star 2006). The framework includes a decision tree to help determine the appropriate statistical methods depending on data quality and quantity. Where reference data are available (e.g., historical, upstream or reference site) and these data are distributed normally, the 95th percentile was selected as the standard statistical measure to estimate the upper limit of the normal range of natural concentrations (Intrinsik 2010). More complicated methods (e.g., non-parametric statistics, rating curves, land-use gradients) are recommended for different types and quantity of data.

4.2 Occurrence in mixtures

As discussed in section 3.0, "Production and Uses," uranium facilities can release contaminants other than uranium into the environment. In particular, radioactive elements such as thorium, radon, radium and radioactive lead, can be released (Environment Canada and Health Canada 2003); the hazard of these compounds is fundamentally different from uranium, as toxicity occurs through alpha, beta and gamma radiation, possibly leading to genetic damage. Non-radioactive metals and metalloids such as cadmium, nickel, copper, arsenic and molybdenum may also be released at uranium mines and mills (Environment Canada and Health Canada 2003; Pyle et al. 2001), and metals (e.g., copper, zinc) may also be released at nuclear generating facilities and waste management facilities (Environment Canada and Health Canada 2003). Non-metal pollutants may also be released from nuclear generating stations (e.g., hydrazine) and waste management facilities (e.g., organic contaminants) (Environment Canada and Health Canada 2003). Physical stressors co-released with pollutants from nuclear facilities include heat and saline water (Environment Canada and Health Canada 2003). The guideline for uranium does not take the impacts or risks of mixtures into account as there is not enough information available.

4.3 Concentrations in surface waters (freshwater and marine)

In a federal assessment, the natural background of uranium in surface water has been estimated to be 0.35 µg/L in northern Saskatchewan and 0.28 µg/L near Elliot Lake in Ontario; both estimates use the 95th percentile approach (Environment Canada and Health Canada 2003). Median concentrations of uranium in both of these areas were below the detection limit of 0.05 µg/L. In cases where uranium ore deposits in northern Saskatchewan were near or under the lake sediments, the baseline concentration (based on 90th percentiles) was calculated as 0.85 µg/L, and the mean was 0.49 µg/L, indicating that natural uranium concentrations are relatively low (Environment Canada and Health Canada 2003). Field measurements by Waite et al. (1988)—who report a background uranium concentration of 0.2 µg/L in Lake Athabasca, Saskatchewan (SK), an area with naturally high levels of uranium—do not exceed these values, and so are in rough agreement with the former estimates from the assessment report. The monitoring of surface waters in Alberta between 1998 and 2008 has shown that uranium concentration in rivers ranged between 0.006 and 5.14 µg/L, with a mean of 0.704 µg/L. The concentration in lakes ranged between 0.0001 and 1.55 µg/L, with a mean of 0.37 µg/L. The surface water bodies sampled during this survey were believed to be weakly influenced by industrial or municipal development (Alberta Environment 2010). Due to the presence of uranium ore deposits in Canada, there is a great deal of information regarding concentrations of uranium in waters surrounding these areas and mining activities associated with these deposits.

Concentrations of uranium in surface waters at decommissioned and active mining sites are summarized in Table 6. In some cases, there is an overlap between the predicted background concentrations of uranium and the lower range of measured values at nuclear facilities. Some decommissioned sites may have higher measured uranium concentrations than active mine sites. Decommissioning a site is described as the actions taken to retire a licensed facility into a predetermined state. These actions will take into account health, safety, security and the protection of the environment (Canadian Nuclear Safety Commission 2000).

Contrary to the freshwater situation, there are very limited data for marine levels. In seawater of the Atlantic and Pacific oceans, the total (unfiltered) concentration of uranium was found to be 3.1 μg/L or, when normalized for salinity on a strictly weight basis, 3.238 μg/kg (Chen et al. 1986; Choppin and Stout 1989). The normalized data were collected from a variety of depths, from 10 m to 5740 m, from two locations in the Atlantic Ocean and one in the Pacific Ocean (Chen et al. 1986). The seawater value for natural background is approximately 9 to 11 times greater than those reported for freshwater; there is no apparent explanation for this difference.

In southeastern Manitoba, groundwater samples had a range of uranium concentrations from < 0.02 μg/L to 2020.0 μg/L with a median of 10.0 μg/L and a mean of 58.3 μg/L (Betcher et al. 1988).

4.4 Concentrations in biota

Data for tissue concentrations in biota are summarized in Table 7. Tissue concentrations in algae, macrophytes, invertebrates and fish are valuable because they can be used to estimate uranium exposure through the food web. Concentrations of contaminants in food may contribute substantially to the dose of uranium and can lead to long-term toxicity.[3]

Within the freshwater ecosystem, no studies or risk assessments were found that measured or estimated the uranium dose from food consumption relative to direct water column exposure. An assessment report (Environment Canada and Health Canada 2003), however, estimated the dose to muskrat and mink from freshwater macrophytes and fish, respectively. Based on risk quotients,[4] the report concluded that at some mining sites in Saskatchewan, consumption of uranium-containing macrophytes and fish (along with water and sediment) were important pathways for uranium exposure and potential toxicity to these terrestrial mammals (Environment Canada and Health Canada 2003). One study from the marine ecosystem showed that in a crab (*Pachygrapsus laevimanus*) and zebra winkle (*Austrocochlea constricta*) native to Australia, uptake of uranium was primarily from water exposure, as opposed to food exposure (Ahsanullah and Williams 1989).

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[3] Short-term studies are defined as having exposure periods of 96 hours or less and long-term studies are defined as ≥ 21 days for adult or juvenile fish, ≥ 7 days for fish eggs and larvae, ≥ 96 hours for short-lived invertebrates (e.g.., Ceriodaphnia dubia), ≥ 7 days for non-lethal endpoints of longer-lived invertebrates, ≥ 21 days for lethal endpoints of longer-lived invertebrates, all tests for *Lemna* sp., and ≥ 24 hours for all algae. All other studies are considered on a case-by-case basis (CCME 2007).

[4] A risk quotient is the quotient of the estimated exposure value divided by the estimated no-effects value, where the value is concentration or dose (Environment Canada and Health Canada 2003).

The concentration of uranium in biota is important to consider in guideline derivation due to the possibility of bioaccumulation and/or biomagnification.

5.0 ENVIRONMENTAL FATE AND BEHAVIOUR

As with many other metals of concern, the environmental fate and behaviour of uranium are dependent on abiotic conditions, such as pH, hardness, alkalinity and natural organic matter (NOM). These abiotic factors influence the bioavailability, toxicity and mobility of uranium by altering the speciation, or physical-chemical forms, of uranium in aquatic systems.

5.1 Partitioning within the aquatic ecosystem

Within the aquatic ecosystem, trends can be discerned from chemical properties and field studies. Fate within the aqueous phase (speciation) is influenced by abiotic features (see section 5.2, "Aquatic chemistry and speciation"), and through biotic uptake. Organisms themselves are an environmental compartment (i.e., through biotic uptake), although uranium does not move through the food web efficiently (Environment Canada and Health Canada 2003; Swanson 1985). Uranium tends to partition into sediments (ATSDR 1999), as evidenced by high K_d[5] values between 0.36 and 3.2 x 10^3 L/kg wet weight (ww) (Swanson 1985). K_d is governed by precipitation and sorption reactions (see section 5.3, "Precipitation and sorption of uranium"). Within sediments, biotic process (e.g., activity of microbes) can also alter environmental fate (see section 5.4, "Microbial transformation").

5.2 Aquatic chemistry and speciation

As mentioned above, several uranium compounds can be found in the aquatic environment. These compounds or species differ in their physical and chemical properties, as well as their toxicity. The speciation of uranium depends on abiotic conditions, such as pH and presence of complexing agents. Furthermore, in comparison with other metals, two unique features of uranium speciation are worth noting so that faulty generalizations are avoided.

(i) *Relative abundance of species changes with total uranium concentration*
Under a given set of abiotic conditions, uranium forms multi-hydroxide and multi-carbonate complexes, and the relative abundance of these species changes with the total uranium concentration in the system. For example, under moderate hardness, the uranium species $UO_2(CO_3)_2^{2-}$, often considered one of the species of concern in relation to aquatic toxicity, varies from 70.6% to 6.14% relative abundance depending on the total amount of uranium in the system, with a higher percentage of the species (70.6%) being seen at the lower ranges of total uranium concentrations (Barata et al. 1998). Numerous other uranium species are also predicted to be present in lesser amounts, and the relative abundance of these species is also predicted to change with total uranium. This indicates that relative

[5] K_d is the partition or distribution co-efficient, and is the ratio of contaminant in the sediment (mg/kg) to the contaminant in the aqueous phase (mg/L) (Whicker and Schultz 1982b). The units of K_d are typically L/kg ww.

abundance of species is not constant with total uranium. Large variations in speciation with total uranium are commonly predicted from speciation codes (Barata et al. 1998; Markich et al. 2000).

From a theoretical perspective, the distribution among aqueous species is related to the metal-to-ligand ratio. Under conditions in which the concentration of ligands is much greater than that of the metal, the relative distribution of metal speciation tends to remain stable. However, when the speciation of the ligand becomes significantly affected by the increase in metal concentration, then a significant shift is observed in the relative distribution. This is the case with uranium, because it has a tendency to form hydroxyl and carbonate complexes (Fortin et al. 2004).

(ii) *Occurrence of the uranyl ion (UO_2^{2+})*
For other metals of potential concern (e.g., zinc, copper, cadmium), the free ion in solution is the ionic form of the element (e.g., Zn^{2+}, Cu^{2+}, Cd^{2+}). Several studies have agreed that under oxidizing conditions, the uranyl ion UO_2^{2+}, as opposed to U^{6+}, is the dominant "free ion" in aquatic systems (Barata et al. 1998; Morse and Choppin 1991; Sylwester et al. 2000).

Although the speciation of uranium in water is complex, modelling results show that conditions that favour the formation of the free ion UO_2^{2+} include low pH and low concentrations of natural organic matter, and probably low alkalinity (Gilbin et al. 2003; Markich et al. 2000; Riethmuller et al. 2001). Details on speciation studies are further discussed in relation to toxicity in section 8.0, "Toxicity of Uranium to Aquatic Life."

5.3 Precipitation and sorption of uranium

In addition to the chemical reactions that occur in the aqueous phase, uranium (like other metals) also interacts with solid particles, such as those found in sediment, through sorption. Dissolved metals may also precipitate out of the water column, hence changing physical-chemical form from aqueous to solid. Although precipitation and sorption are distinctively different processes,[6] sorption influences the reactivity of surfaces, including rates of surface precipitation (Stumm and Morgan 1996). Both precipitation and sorption of uranium influence the portion of the metal in the bioavailable form by removing uranium from the aqueous phase.

In one modelling exercise, with parameters set to mimic seawater conditions, haiweeite ($(Ca(UO_2)_2Si_2O_5.5H_2O)$) was assumed to be the solid that limits solubility. This modelling exercise predicted the UO_2^{2+} concentration about an order of magnitude greater than the measured value; the authors suggest that overestimation of the model may be a result of ignoring sorption processes or problems with the constants used in the calculations (Choppin and Stout 1989). Sylwester et al. (2000) suggest that UO_2^{2+} can precipitate in the form of multinuclear hydroxides, or, in the presence of silicates or aluminates, co-precipitate as Si- or Al- compounds (e.g., soddyite, weeksite).

[6] Adsorption is the accumulation of matter at the solid-water interface. For metals, example processes include: surface complexation reactions (chemical bond formation, e.g., through hydrolysis) and electrostatic interactions (e.g., ion exchange) (Stumm and Morgan 1996). Bond formation is also referred to as inner-sphere complexation, and ion exchange as outer-sphere complexation. Precipitation is a change in state from the aqueous form to the solid form.

Most chemical reactions that occur in natural waters take place at phase interfaces (e.g., solid-solution) (Stumm and Morgan 1996), and hence sorption can play a major role in the environmental fate of metals.

Sediments have a cation exchange capacity, which allows reversible binding of trace elements at exchange sites on the surface (Manahan 1994). As hydrogen ions can compete for these same sites, sorption of cationic metals is generally inhibited at low pH; this trend is also well-documented for UO_2^{2+}. Sorption also tends to decrease at high pH values, although there are more inconsistencies with this trend. Adsorption experiments with hematite (Fe_2O_3) showed maximal adsorption between pH ~5 and 8.5, dropping off to zero adsorption at pH 3 and pH 10 (Lenhart and Honeyman 1999). Comparison of UO_2^{2+} adsorption at two pH values (~pH 3–4 and ~pH 6.4) on three different mineral surfaces (silica, alumina, montmorillonite[7]) confirmed the trend of maximum absorption at near-neutral pH, with few differences between the mineral type (Sylwester et al. 2000). While low adsorption of uranium at low pH values and high adsorption at near-neutral pH values was also observed in Zuyi et. al. (2000), adsorption continued to be high at pH values up to pH 10.

The scientific consensus is that the presence of organic ligands (typified in laboratory batch experiments with NOM) tends to increase adsorption of uranyl to hematite at low pH values and decrease uranyl adsorption at high pH values (Lenhart and Honeyman 1999 and references therein). Mechanistically, the differences in sorption related to organic ligands are explained through the formation of ternary complexes, which are at the junctions between the mineral surface, the organic ligand and the metal ion. The data of Lenhart and Honeyman (1999) show the well-defined low pH trend, with NOM (as humic acid) increasing sorption of UO_2^{2+} to hematite (Fe_2O_3) at pH ~3–5. However, this trend varies across mineral types, as when fulvic acid (for NOM) was used in conjunction with UO_2^{2+} sorption to Al_2O_3, Fe_2O_3 and SiO_2. Here, decreased sorption was highly noticeable with Fe_2O_3, yet only marginal with Al_2O_3 and SiO_2 (Zuyi et al. 2000). In both studies, the organic ligand only marginal reduced sorption at high pH values (Lenhart and Honeyman 1999; Zuyi et al. 2000).

Changes in ionic strength can alter sorption of UO_2^{2+} as cations compete for exchange sites on the mineral surface (Sylwester et al. 2000). For example, with increased ionic strength from 0.001 to 0.01 and 0.1, the sorption envelope with ferric oxide slightly decreased; that is, sorption was increased at low and high pH values with lowered ionic strength (Lenhart and Honeyman 1999). Similar effects were noted when ionic strength was altered with montmorillonite (Sylwester et al. 2000).

In contrast to the well-studied phenomena of sorption and precipitation, information on the release from contaminated sediments is scarce. One field study (Waite et al. 1989) has reported on the release of uranium and radionuclides from sediments at a contaminated site (Langley Bay) in Saskatchewan. Here, a uranium mine that operated from 1955 to 1964 released tailings that covered the entire bottom of the bay. Sample collection of flux showed that in 1986, movement of ^{210}Pb and ^{226}Ra from historically contaminated sediments into the aqueous phase could account for the elevated concentrations in the surface water of the bay. Flux of ^{228}Th and uranium from sediments was below detection limits; hence, for uranium, no conclusions could be drawn on the contribution of the contaminated sediment to elevated surface water concentrations at Langley Bay (Waite et al. 1989).

[7] Montmorillonite is a silicate clay, with chemical formula $Na_{0.2}Ca_{0.1}Al_2Si_4O_{10}(OH)_2$.

Uranium is also present in groundwater. It is generally found in higher concentration than in surface water due to interaction with underlying rock structures and weathering of surface rocks (WHO 2001). In groundwater, weak acidic conditions usually predominate. As mentioned above, these conditions may promote the adsorption of uranium, resulting in the formation of stable complexes with organic matter. Subsequently, deposition and accumulation of uranium in the matrix can occur. However, in some conditions, bounded uranium can be transported to surface water with dissolved organic matter (WHO 2001).

Sorption and precipitation of uranium are considerations in the development of a guideline for uranium, as they allow better characterization and understanding of how uranium interacts in the environment. How quickly and easily uranium species adsorb to other compounds, and what compounds those are, can affect the bioavailability of uranium in aquatic systems. Given more data, it is possible that an effect of pH on the toxicity of uranium could be seen, perhaps through adsorption to organic ligands, as described above. The more binding sites with strong affinities for uranium that are present, the greater the competition for entering biota as well, which could indicate a decrease in bioavailability of uranium under certain circumstances. More data are needed to determine the relationships that exist here.

5.4 Microbial transformation

Reduction of metals can take place by abiotic processes as well as via certain strains of anaerobic bacteria that can directly reduce U(VI), the soluble state, to U(IV), the insoluble state (Lovely et al. 1991). As U(IV) is less soluble than U(VI), this reduction can result in immobilization in aquatic sediments. In anaerobic sediments spiked with U(VI) as uranyl acetate, reduction to U(IV) proceeded much more rapidly in sediments that had live cell cultures, as opposed to sediments that had been sterilized via autoclaving (Lovely et al. 1991). This pathway is an enzymatic reduction of uranium using U(VI) as a terminal electron acceptor, and is quite unlike the abiological reduction as seen with sulphide, H_2, and organic compounds in sediments. These results from Lovely et al. (1991) show that microbial reduction of U(VI) to U(IV) can proceed much faster than abiotic reduction, and hence is an important pathway in uranium fate and behaviour.

5.5 Persistence and residence times

Persistence and residence times are difficult to interpret for metals, and are not related to toxicity and exposure in the same way that degradable contaminants are (e.g., degradation of contaminants affects exposure time, which affects toxicity). This is because metals do not degrade beyond their ionic forms.

6.0 BIOCENCENTRATION AND BIOACCUMULATION

While bioconcentration and bioaccumulation are not considered in the derivation of a Canadian water quality guideline, a discussion on this topic is included here for three reasons: (i) to highlight the active debate among experts on the use of bioaccumulation as an index of metal hazard in general, using uranium as a example where applicable; (ii) to briefly compile information on tissue concentrations of uranium reported in the literature; and (iii) to evaluate the uranium water quality guideline in relation to bioconcentration/bioaccumulation.

6.1 Bioaccumulation and bioconcentration: history and concepts

Bioaccumulation and bioconcentration (B), in conjunction with persistence (P) and inherent toxicity (iT), are important properties to consider in hazard identification, and can help inform the prioritizing of environmental contaminants (McGeer et al. 2003). Metrics based on these concepts (PBiT) were developed in the 1970s to systematically evaluate the growing number of organic contaminants that were causing non-target effects in the environment (Skeaff et al. 2002). The most common expressions of the process of bioaccumulation are the BAF (bioaccumulation factor) and BCF (bioconcentration factor), which are simply ratios of the internal concentration of a contaminant (within an organism) to the contaminant concentration in ambient water. Internal concentrations can be reported as whole-body concentrations, or as concentrations in specific parts of the body, such as gills, liver or muscles. BAFs are interpreted to include intake from food as well as ambient water, and so are generally applicable to field measurements, whereas BCFs capture uptake strictly from ambient water, and are therefore usually derived from laboratory data.[8] In Canada (and internationally), bioaccumulation as part of PBiT was subsequently written into legislative frameworks (Canada 1999) to aid risk assessors in categorization of existing[9] substances (Canada 2000; Environment Canada 2003; Schnabel et al. 2003).

The environmental fate and toxicology of metals is substantially different from that of organics, complicating the use of bioaccumulation as a criterion in metals assessment (Table 8). Bioaccumulation/bioconcentration criteria were originally developed for organic contaminants that were generally synthetic, exerted toxic effects through narcosis, and were neutral and lipophilic (McGeer et al. 2003). For organic lipophilic contaminants, bioaccumulation and bioconcentration can potentially indicate the risk of biomagnification and trophic transfer up the food chain; toxic effects may then be seen in organisms at the top of the food web (Government of Canada and Environment Canada 1995). In Canada, under the Toxic Management Substances Policy, bioaccumulation (and persistence) criteria were meant to be applied to substances that were also toxic and predominantly anthropogenic (Canada 1995). Metal contaminants, by comparison, are naturally occurring and have physical and chemical properties that can be different from those of organic contaminants (Table 8). BAFs of the organic contaminants for which they were originally derived are roughly independent of exposure concentration, and so are useful intrinsic or quasi-intrinsic properties in the hazard ranking of chemicals (Franke et al. 1994; Mackay et al. 2001). In contrast, BAFs for metals are not constant across exposure concentrations, possibly due to an ability by organisms to acclimate to higher exposures of metals and control, to some degree, bioaccumulation (McGeer et al. 2003). Critically, as BAFs for metals tend to be highest at low exposure concentrations (where they are expected to be less toxic), they are not reliable indicators of hazard. More specifically, this trend has also been observed for uranium in brook trout (*Salvelinus fontinalis*), where BCFs decreased with increasing uranium exposure (Parkhurst et al. 1984). Experts have also commented that BAFs do not reflect the essentiality of some metals and internal detoxification mechanisms (Brix and DeForest 2000; McGeer et al. 2003).

[8] Although in theory, BCFs and BAFs differ in the denominator used in the ratio, in practice, both can be calculated using the concentration of contaminant in water as the denominator (Environment Canada 19992000). For the purposes of this brief section, they will be treated as measures of the same phenomenon, and will not be differentiated.

[9] In the New Substances Program, bioaccumulation potential is assessed, but it is not used in the same categorization manner that is employed for existing substances (2004 personal communication from M Lewis, Ecological Assessment Division, Environment Canada; unreferenced).

Furthermore, the interpretation of quantities such as the bioconcentration factor is problematic for metals (McGeer et al. 2003).

Selected results from uranium tissue bioconcentration and/or bioaccumulation in aquatic biota are summarized in Table 7. Only one data point for marine species was found, and it is grouped with the abundance of freshwater data for completeness. Results from the PSL report group bioconcentration ratios (ratio between uranium in tissue and water, in L/kg wet weight or ww) of ^{238}U by general taxa of fish, algae and macrophytes (Environment Canada and Health Canada 2003). The geometric mean of the bioconcentration ratios were 89 (maximum of 158) for algae, 1.5 (max 38) for macrophytes, and 1.24 (max of 38) for fish (Environment Canada and Health Canada 2003).

Because of the problematic implementation of a bioaccumulation or bioconcentration criterion for metals and the scope of the CWQG, bioaccumulation is not considered to be part of the "Protocol for the Derivation of Canadian Water Quality Guidelines for the Protection of Aquatic Life," as this protocol deals with the concentration of the substance in the water column and the toxic effects resulting from direct exposure (CCME 2007). However, it is still taken into consideration, and can be used in more detail on a case-by-case basis.

6.2 Partitioning of uranium accumulation within the organism

Evidence from whitefish (*Coregonus clupeaformis* and *Prosopium cylindraceum*), rainbow trout (*Oncorhynchus mykiss*), lake trout (*Salvelinus namaycush*), and northern pike (*Esox lucius*) suggests that within the fish, uranium concentrations in the gut from food are generally higher than those in fish tissue (Clulow et al. 1998; Cooley and Klaverkamp 2000; Poston 1982;Waite et al. 1988; Waite et al. 1990). Lake trout and whitefish from lakes associated with uranium mining operations have been found to contain higher levels of uranium in the bones and gut contents than in muscle (Clulow et al. 1998). Within fish tissue itself, uranium tends to accumulate in mineralized tissue, such as bone and scales, and to a lesser extent in the kidney, with measurable accumulation in the liver, gills, skin and muscle (Cooley and Klaverkamp 2000; Waite et al. 1990). Under some exposure conditions,[10] high concentrations of uranium can accumulate in the gonads (Cooley and Klaverkamp 2000). Histopathological examinations of whitefish (*Coregonus clupeaformis*) suggest that although uranium tends to accumulate in mineralized tissue, the kidney is the primary site of action for food-borne long-term exposure; the liver is an important site of uranium-induced toxicity. This is reasonably similar to mammalian studies, which show that the kidney is the primary target organ (Environment Canada and Health Canada 2003; Federal-Provincial-Territorial Committee on Drinking Water 2001; Ribera et al. 1996).

No information on organ partitioning of uranium in marine invertebrates was found. For freshwater invertebrates, a few studies on the bivalve *Corbicula fluminea* were found. When *C. fluminea* was exposed to high levels of uranium (93 500 µg U/L), the gills and visceral mass accumulated more

[10] At the highest uranium concentration in food tested in this study (10 000 µg/g, ww), uranium concentration in the gonads and ovary maturation in females peaked at 30 days (when compared to 10 and 100 days). At 30 days, the concentration of uranium in the gonads exceeded uranium concentrations in bones and scales by approximately a factor of 6.6 and 4.0, respectively. The fish were 3.5 years old at the start of exposure (Cooley and Klaverkamp 2000).

uranium than the foot (Labrot et al. 1999). This was further substantiated by Simon and Garnier-Laplace (2004), who found that the foot had a weak and constant accumulation of 2.6 µg/g ww, and the visceral mass had the highest level of accumulation of any organ tested when exposed to 63 µg U/g ww, accounting for 82% of the total burden of uranium. This study also showed that the gills contained a higher percentage of the total accumulated uranium (49.1% and 38.4%) under high exposures (482 and 1477 µg U/L respectively), whereas the visceral mass was favoured under the lower exposure conditions. Simon and Garnier-Laplace (2005) later found that in the crayfish (*Orconectes limosus*), uranium was primarily accumulated in the stomach and particularly the digestive gland. Muscatello and Liber (2009) found that under exposure experiments with *Chironomus tentans*, exuvia had the highest uranium concentration, resulting in large uranium elimination during moulting.

In marine molluscs (*Mytilus edulis* and related species) and crustaceans (the crab *Liocarcinus puber* and related species), uranium accumulated mainly in the digestive gland, the gills and the exoskeleton (Chassard-Bouchaud 1996), similar to the freshwater crayfish as found by Simon and Garnier-Laplace (2005). There was also very little information on uranium partitioning within aquatic plants, although one field study showed higher accumulation in the roots, as compared to the shoots, of cattails (*Typha* sp.) (Waite et al. 1988).

Uranium accumulation in the Asiatic clam was dependent on sorption-absorption, excretion, and storage processes, and varied among organisms (Simon and Garnier-Laplace 2004). Simon and Garnier-Laplace (2004) found an accumulation level of 10 µg/g ww over 42 days of exposure of the Asiatic clam to 63 µg U/L, with an accumulation factor of 160 from a direct water exposure. After 21 days of exposure to 93 500 µg U/L, the same species of clam was seen to have a maximum accumulation of 27.0 µg U/L, with a BCF of 0.05567 (Labrot et al. 1999). As has been discussed, uranium can bioaccumulate in aquatic organisms, though it does not bioconcentrate or biomagnify. However, long-term exposures could reach levels that could be toxic to aquatic organisms when accumulation is taken into consideration. It is important to examine the levels of bioaccumulation of specific areas; however, with different ecosystems and conditions, such as different pH and hardness values, a conclusive, all-encompassing answer is not possible at this time, and it is beyond the scope of this document.

6.3 Biomagnification

Inorganic metals do not biomagnify in food webs (Brix and DeForest 2000), although in some cases, methylation of metals such as mercury can cause substantial and ecologically hazardous biomagnification (Schnabel et al. 2003). Organisms do accumulate uranium, but because it has a low assimilation efficiency, it does not biomagnify (Environment Canada and Health Canada 2003; Simon and Garnier-Laplace 2005; Swanson 1985). Trophic transfer rates of uranium (1-13%) have been found to be low, similar to that of cadmium (Simon and Garnier-Laplace 2005). Organisms lower on the food chain typically have higher levels of uranium than upper trophic level organisms (Environment Canada and Health Canada 2003).

6.4 Secondary poisoning

Secondary poisoning[11] in the aquatic system may be a concern despite lack of biomagnification, as concentrations in tissue are a potential route of exposure through the food web. While uranium is not expected to biomagnify in food webs, the process of bioaccumulation does contribute to uranium intake at higher trophic levels, which may in turn contribute to secondary poisoning. Under some circumstances, food-borne intake contributes to total daily uranium intake, which may pose a risk of adverse effects. As mentioned earlier, risk assessments have shown that fish and aquatic plants are a source of uranium exposure to terrestrial wildlife such as mink and muskrat respectively (Environment Canada and Health Canada 2003). At some of the sites evaluated, the total uranium intake results in a risk quotient that is greater than 1, showing the potential for adverse effects to occur. The range of risk quotients reported was from 0.14 for mink and 0.17 for muskrat under conditions of natural background (0.35 µg U/L) up to a high of 10 for mink and 21 for muskrat under high environmental conditions of 1061 µg U/L. Data gaps exist for uranium toxicity to birds, amphibians and reptiles (Environment Canada and Health Canada 2003), all of which may be susceptible to secondary poisoning through the aquatic ecosystem.

7.0 EXPOSURE AND ROUTE OF UPTAKE

7.1 Essentiality

Uranium is not known to be essential to mammals (Berlin and Rudell 1986). Similarly, there have been no reports of a metabolic function for uranium in aquatic organisms, although there has been a report of mild growth stimulation of the marine amphipod *Allorchestes compressa* at low concentrations (100 µg/L) of depleted uranium[12] (Ahsanullah and Williams 1986).

7.2 Mode of action and toxicokinetics

This document reflects the information concerning the chemical toxicity of uranium only, and does not include information concerning the radiological toxicity of uranium. For more details, see section 8.0, "Toxicity of Uranium to Aquatic Life."

Researchers have suggested that for short-term toxicity in fish, the site of action may be the gill, as is common for metal toxicants in general (Bywater et al. 1991). In the Asiatic clam, *Corbicula fluminea*, Simon and Garnier-Laplace (2004) suggested that the main site of toxicity of uranium was the digestive organ, which they also saw in the freshwater crayfish *Orconectes limnosus* (Simon and Garnier-

[11] Brix and DeForest (2000) provide good and distinctive definitions between some apparently overlapping terms (italics added): "*Secondary poisoning* results when toxicant concentrations in an organism reach a level that is toxic to the organisms that feed on it. Substances that *bioaccumulate* or *biomagnify* in food webs are often considered to have the greatest potential to cause *secondary poisoning*. *Biomagnification* is the process by which tissue concentrations of a *bioaccumulated* substance increase as it is passed up to the food web through at least two trophic levels."

[12] Although growth increased at the lowest concentration tested (0.1 mg/L), respiration rate decreased at the same concentration, showing that the stimulatory effect is endpoint-dependent.

Laplace 2005). They also found uranium to have a similar pathway in the crayfish as cadmium, mercury and copper (Simon and Garnier-Laplace 2005).

At a cellular level, toxicity may occur as a result of the binding of uranium to enzymes (Khangarot 1991), which would lead to subsequent inactivation or disabling of enzyme function. If UO_2^{2+} behaves as a calcium (Ca^{2+}) mimic, as is the case for a wide variety of other transitional metals (Foulkes 1990), uranium may exert toxicity at the cellular level by interfering with calcium homeostasis. Tran et al. (2005) reported that the multixenobiotic resistance protein (MXR) was induced when the Asiatic clam was exposed to uranium. The MXR protein is a membrane extrusion pump that mediates the efflux of numerous xenobiotics. It could play a role in uranium contamination, as many studies support the hypothesis that in addition to organic xenobiotics, MXR proteins might pump metals (Tran et al. 2005).

In mammals, the biokinetics (absorption, distribution, transformation and elimination) of uranium are well-studied, and the target organ has been identified as the kidney (ATSDR 1999; Federal-Provincial-Territorial Committee on Drinking Water 2001). In lake whitefish (*Coregonus clupeaformis*), one study suggests that food-borne uranium via long-term exposure causes kidney damage as the probable primary effect, and also liver damage (Cooley et al. 2000).

In biochemical studies, biomarkers or bioindicators are used to detect early cellular responses occurring due to a toxicant exposure. These molecular and cellular level experiments measure sub-lethal effects that could potentially affect growth, reproduction and survival of the organism. In general, biochemical studies in fish have not been successful in elucidating the mode of action or revealing sensitive bioindicators of damage. In lake trout (*Salvelinus namaycush*), muscle RNA/DNA ratio, whole-body triglycerides and total muscle protein were not predictive of sub-lethal effects from waterborne uranium exposure in an early life stage test (Liber et al. 2004a). Elevated levels of metallothionein were not indicative of long-term food-borne uranium exposure in lake whitefish (*Coregonus clupeaformis*); however, lipid peroxidation (when used in conjunction with tissue accumulation) may be a useful indicator of toxicity, as it was significantly elevated at the lowest treatment exposure of 10 µg U/g in food (Cooley et al. 2000).

7.3 Speciation and bioavailability[13]

In surface water, the bioavailability of uranium depends on its speciation. According to the free ion activity model (FIAM), metal toxicity in aquatic systems is better correlated with the concentration of free ion than with total metal concentration, although there are reports of apparent exceptions to this model (Campbell 1995). For uranium in particular, there are some studies that have examined the assumptions and potential exceptions to FIAM. As stated by Markich (2002), it seems that UO_2^{2+} and UO_2OH^+ are the major forms available to organisms. Their complexation with inorganic ligands or their adsorption to colloidal or particulate matter reduce the activity of UO_2^{2+} and UO_2OH^+ and thus the bioavailability of uranium (Markich 2002). Following a stepwise multiple linear regression on modelled uranium speciation, Markich et al. (2000) found that both UO_2^{2+} and UO_2OH^+ were significant predictors of sub-lethal short-term toxic response (in this case, valve movement in a

[13] For the reasons outlined in the Protocol for the Protection of Aquatic Life, the discussion of bioavailability is restricted to speciation.

freshwater bivalve over a 48-h period). Together, UO_2^{2+} and UO_2OH^+ explain 97.5% of the variability in toxic response, whereas individually, these species are poor predictors of toxic response (Markich et al. 2000). The authors conclude that these results provide evidence of an exception to the FIAM with uranium. This type of experiment has not been repeated on other species, and other speciation-based toxicity conclusions assume that UO_2^{2+} is the toxic species. Algal studies have focused on uptake, as opposed to toxicity, of uranium (Fortin et al. 2004, 2007). First, they observed that uranium uptake was influenced by water pH, where maximum uptake occurred at pH 7 compared to pH 5 (Fortin et al. 2007). They concluded that the simple proton-metal competition usually described by the biotic ligand model (BLM)[14] could not successfully explain uranium-algae interactions when pH was varied. At a constant pH and in the presence of three different ligands, the FIAM reliably predicted uranium uptake as free ion, suggesting that uranium complexes are not bioavailable (Fortin et al. 2004). Another study found that high CO_2 (276 µmol/L) decreased uranium bioavailability in the gills and the mantle of the Asiatic clam (Tran et al. 2004). The authors indicated that this was likely because uranium species bound to carbonates were not particularly bioavailable.

One clear advantage of the FIAM is that it partially takes into account the differences in metal toxicity that are observed under different water chemistry conditions. Regardless of the extent to which uranium does or does not conform to the FIAM, a survey of the observation-based effects of water chemistry on uranium toxicity enhances the ability to predict toxicity.

7.4 Exposure and route of uptake

In general, dietary exposure to metals can be toxicologically relevant. Metal uptake from dietary sources occurs independently from water uptake and thus, toxicity from dietary exposure cannot be predicted from water exposure toxicity (Chapman 2008). There has not been a lot of study comparing waterborne exposures and dietary exposures to uranium; however, Simon and Garnier-Laplace (2005) found that trophic transfer through the diet was minimal in the absence of waterborne exposures. Uranium has also been found to have a very low rate of uptake through the gut of many species (Environment Canada and Health Canada 2003), indicating that dietary exposure is likely not the primary route of uptake. Most uranium toxicity tests have attempted to isolate water as the main route of exposure; however, in long-term studies, animals would need to be fed, and hence some of the toxicity results that are interpreted as water-only exposures could be confounded with uranium-contaminated food supplies. In high-quality toxicity studies, inadvertent uranium intake via food is not expected to be a major issue, as uneaten food would be removed from the exposure system. In addition, in the one uranium-contaminated food study that is available, effects were only observed at relatively high concentrations of uranium (100-10 000 mg/kg) (Cooley et al. 2000).

Uranium is expected to partly partition into sediment. Accordingly, in the environment, sediment ingestion may be a route of exposure. Data on the toxicity of uranium in sediments are limited. A study in which *Hyalella azteca* was exposed to water overlying sediments spiked with uranium showed that uranium bioaccumulation and toxicity were due primarily to the dissolved phase rather than the sediment solid phase (Alves et al. 2008). The overlying water chemistry, mostly pH, influenced

[14] The biotic ligand model (BLM) is a bioavailability model that predicts site-specific metal toxicity by using local water chemistry information.

uranium desorption from sediment particles and thus uranium speciation in water (Alves et al. 2008). In a laboratory study using *Tubifex tubifex*, malformations and reduction of survival, biomass and burrowing activities were observed after a 12-day exposure to sediments containing more than 600 µg U/g dry weight (dw) (Lagauzère et al. 2009). Although natural uranium concentrations in freshwater sediments are usually below 10 µg U/g dw, maximum concentrations between 450 and 18 000 µg U/g dw have been measured near mining sites (Hart et al. 1986; Kurnaz et al. 2007; Lottermoser et al. 2005; Lozano et al. 2002; Neame et al. 1982) In a case study on a pond contaminated by past uranium mining activities (Cunha Baixa, Portugal), screening bioassays on the acute toxicity of the different compartments using algae, crustaceans and dipterans showed that, unlike superficial water, sediments were non-toxic (Antunes et al. 2007). One study from the marine ecosystem showed that in a crab (*Pachygrapsus laevimanus*) and zebra winkle (*Austrocochlea constricta*) native to Australia, uptake of uranium was primarily from water exposure, as opposed to food exposure (Ahsanullah and Williams 1989). In conclusion, water is expected to be the primary route of exposure for uranium.

8.0 TOXICITY OF URANIUM TO AQUATIC LIFE

This report only focuses on the chemical toxicity of uranium and does not include its radiation toxicity. Uranium is an alpha particle emitter. Alpha radiation has an extremely low penetrating power; therefore, the ionizing radiation from uranium would be attenuated in about 50 µm in water or tissue (Bleise et al. 2003; Kuhne et al. 2002; Whicker and Schultz 1982a). As a result of uranium's low penetrating power, and because it is a weak emitter (shown by the long half-life), the radiotoxicity of uranium from aqueous exposure is expected to be minimal. The risks from chemical toxicity of uranium in freshwater ecosystems are generally higher than the radiological toxicity risks (Mathews et al. 2009). In toxicity experiments on *Daphnia magna*, Zeman et al. (2008) confirmed that uranium chemical toxicity predominates over its radiotoxicity. Aquatic organisms that ingest sediment or food contaminated with uranium may be exposed to radiological hazard; however, some scientists (see Kuhne et al. 2002 and references therein) have stated that a large amount of uranium would have to be ingested for the radiological hazard to exceed the chemical hazard.

Uranium is added to water in experimental exposures in a variety of forms, including uranyl nitrate ($UO_2(NO_3)_2.6\ H_2O$), uranyl sulphate (UO_2SO_4), and uranyl acetate ($UO_2(C_2H_3O_2)_2.2H_2O$) to name a few. The concentration of uranyl ion in the water is often the target concentration to indicate toxicity, though speciation models are often used to determine the different uranium compounds in the exposure water. The speciation at the given conditions (e.g., hardness, pH, temperature) is more indicative of toxicity than the nominal concentration and type of compound used to get uranium into the water.

Toxic responses to uranium for aquatic organisms are primarily reported as effects on mortality (e.g., median lethal concentration or LC_{50}s), reproduction, growth, and weight, or as a lack of response, such as reduced valve movement and reduced swimming activity (Sheppard et al. 2005).

8.1 Toxicity-modifying factors

The water chemistry of surface waters in Canada is complex and diverse, and it is beyond the scope of this supporting document to survey background water chemistry conditions in Canada. However, water

chemistry survey data can play an important role in development and application of the CWQG, particularly for metals,[15] since metal toxicity can be modified by hardness, alkalinity, pH and dissolved organic carbon (see section 8.0, "Toxicity of uranium to aquatic life"). Because these water variables are important in the application of the CWQG, development of the guidelines should also be sensitive to and aware of common water chemistry conditions. As partial guidance towards this goal, one report has documented the average values of hardness, alkalinity, pH and dissolved organic carbon in Ontario lakes on the Canadian Shield (Bird and Schwartz 1997). The analyses show that lakes that would usually be considered low hardness (geometric mean of 16.6 mg $CaCO_3$/L, range of 8.49–84.47 mg $CaCO_3$/L) and low alkalinity (geometric mean of 4.56 mg $CaCO_3$/L, range of 0.01–82.88 mg $CaCO_3$/L) are common, that average pH is near neutral, and that average dissolved organic carbon is slightly less than 5 mg/L (Table 9).

Four main water chemistry variables (pH, hardness, alkalinity and NOM) that are known to affect the toxicity of metals are discussed below for the specific case of uranium. Where possible, these variables are considered along with chemical speciation. Temperature, a physical property, is also briefly discussed.

8.1.1 pH

Studies indicate that pH could influence uranium toxicity and uptake via two main mechanisms. First, increasing pH could enhance complexation of the uranyl ion by hydroxides and carbonates, resulting in a decreasing bioavailability. On the other hand, the same increase in pH could decrease uranyl ion competition with protons for the physiologically active sites on the organism membranes (Fortin et al. 2007). Current data on the effect of pH on the toxicity of uranium are limited and inconsistent. Some data suggest that at low pH, the freshwater clam *Velesunio angasi* and the freshwater bivalves *Chlamydomonas reinhardtii* and *Corbicula fluminea* may be more sensitive, presumably because the resultant changes in speciation favour high relative abundance of the toxic free ion UO_2^{2+} (and UO_2OH^+ in one case) (Gilbin et al. 2003; Markich et al. 2000; Simon and Garnier-Laplace 2004). A separate study showed lower toxicity of *Chlorella* sp. at low pH, presumably because H^+ competes with UO_2^{2+} at the site of uptake, and thus produces a protective effect (Franklin et al. 2000).

Decision: There is not enough information on the effects of pH on uranium toxicity to reliably adjust or normalize toxicity data for this variable.

8.1.2 Hardness

Hardness is usually defined as the sum of calcium (Ca^{2+}) and magnesium (Mg^{2+}) cations in solution, although the original definition of hardness focused on the ability of water to precipitate soap (APHA et al. 2005). Alkalinity is defined as the capacity of water to neutralize acid; in many surface waters, alkalinity is primarily due to carbonate concentrations (Environment Canada and Health Canada 2003).

[15] The assessment of the ability of water chemistry parameters to modify the toxicity of uranium specifically is equivocal; for example, for some species in some experiments, variations in hardness and pH show dramatic effects on toxicity. In terms of guideline development, the data for uranium were not normalized for any water chemistry parameters.

In the environment, one main source of both hardness and alkalinity is dissolved limestone ($CaCO_3$), which creates conditions in which hardness and alkalinity can co-vary. However, conceptually, hardness and alkalinity alter toxicity through different mechanisms. While both hardness and alkalinity reduce the concentration of the metal at the biological receptor, Ca^{2+} generally reduces toxicity through competition at the biological receptor, whereas CO_3^{2-} and HCO_3^- form complexes with the metal that generally do not elicit a toxic response. Numerous studies have reported that water with "high hardness" can ameliorate short-term and long-term toxicity of uranium (Table 10); however, in many of these studies, true hardness is confounded with alkalinity because Ca^{2+} was added as $CaCO_3$ (Charles et al. 2002). Results from Riethmuller et al. (2001) suggest that water hardness is more important than alkalinity in reducing uranium toxicity.

In recent studies that have isolated for true hardness (e.g., by adding Ca^{2+} in the form of $CaSO_4$), increases in true hardness have been shown to decrease long-term toxicity to algae and macrophyte *Lemna minor* (Charles et al. 2002; Vizon SciTec Inc. 2004). Algae *Chlorella* sp. showed a more consistent effect of hardness than *Selenastrum capricornutum* (now *Pseudokirchneriella subcapitata*) (Charles et al. 2002; Vizon SciTec Inc. 2004). For *Chlorella* sp., a 50-fold (mg/L) increase in hardness resulted in a 4.8-fold (mg/L) decrease in toxicity (Table 10) (Charles et al. 2002). This reduction in toxicity was most likely due to the competition between uranium and Ca^{2+} and/or Mg^{2+} for the binding sites, since Charles et al. (2002) have shown that the predicted speciation of uranium did not significantly change with increasing hardness.

For invertebrates, one test that isolated for true hardness showed a dramatic modifying effect of long-term uranium toxicity to *Hyalella azteca*; a 16-fold increase in hardness (mg/L) decreased uranium toxicity 12 fold (mg $CaCO_3$/L) in a 14-day growth and survival test (Table 10) (Vizon SciTec Inc. 2004). Other studies in which hardness was increased concurrently with alkalinity have reported reduced short-term toxicity to *Daphnia magna* (Barata et al. 1999; Poston et al. 1984), and reduced long-term toxicity to green hydra, *Hydra viridissima* (Riethmuller et al. 2001).

For fish, current data are equivocal on the effect of hardness on uranium toxicity. Some evidence from older studies suggests that increased hardness (with concurrent increase in alkalinity) does modify short-term toxicity to fish (e.g., Parkhurst et al. 1984). More recent studies that tested with true hardness (no concurrent increase in alkalinity) suggest no effect of increasing hardness on short-term uranium toxicity, and no effect or a moderate effect of hardness on uranium toxicity (Table 10) (Vizon SciTec Inc. 2004).

Sheppard et al. (2005) derived predicted no-effect concentration (PNEC) values for uranium toxicity to various groups of organisms. They also derived an equation to predict PNEC values for uranium for freshwater fish from water hardness, as follows:

$$\text{(effect concentration, mg U/L)} = 0.26*\text{(hardness, mg } CaCO_3\text{/L)}$$

Decision: From these results, it seems clear that changes in hardness alone or in co-variation with alkalinity can affect uranium toxicity toward aquatic organisms. However, at this time, the data regarding hardness effects on uranium toxicity are not consistent, and no constant effect of hardness on toxicity could be determined when all relevant data were examined. From the available information, it cannot be concluded that the protection effect of hardness is similar and generalized among species. Thus, there is not enough information on the effects of hardness on uranium toxicity to reliably adjust

or normalize toxicity data for this variable with a generic correction based on a quantitative relationship between hardness and uranium toxicity.

8.1.3 Alkalinity

Current data on the effect of alkalinity on the toxicity of uranium are limited. There are a number of studies that suggest that modification of toxicity occurs through formation of carbonate or hydroxide-carbonate complexes under conditions that favour high alkalinity (e.g., higher pH values); however, this evidence is indirect, as other water chemistry parameters were not held constant and the toxicity ameliorating mechanism is inferred from modelling results. Only one toxicity study was found that partially manipulated alkalinity at a constant hardness. Although alkalinity changed from 4.0 to 102 mg $CaCO_3$/L, toxicity to the freshwater polyp *Hydra viridissima* did not change when comparing median effective concentration (EC_{50}) values[16] (Riethmuller et al. 2001). The authors reason that no amelioration of toxicity was observed because modelling exercises show that the concentration of the putative toxic species, UO_2^{2+}, changed minimally with the 26-fold increase in alkalinity (from 6% relative abundance at low alkalinity to 1% relative abundance at high alkalinity). Other studies do suggest that under conditions that favour high alkalinity (e.g., mid range or high pH values), carbonate or hydroxide-carbonate complexes do occur in high abundance (Barata et al. 1998; Markich et al. 2000), but as these experiments varied other water chemistry parameters (pH, fulvic acid, hardness with alkalinity), the direct effects of alkalinity on toxicity cannot be resolved.

Decision: The one reliable example on alkalinity effect on uranium toxicity showed that this factor did not modify the toxicity. However, this information on the effects of alkalinity on uranium toxicity is not enough to reliably adjust or normalize toxicity data for this variable.

8.1.4 Natural organic matter

Current data on the effect of NOM on uranium toxicity are limited. Similar to the ameliorating effects of alkalinity, NOM has the potential to bind the toxic (free ion) forms of a metal and hence reduce toxicity.[17] However, unlike alkalinity, NOM is composed of an extraordinarily heterogeneous class of organic molecules with differing physical and chemical properties, including binding affinity for metals (Aiken et al. 1985; Markich and Brown 1999). One research group has established that an increase in fulvic acid (a type of NOM) from 0 mg/L to 7.91 mg/L decreased uranium toxicity to a freshwater bivalve by a factor of 2–3 (Markich et al. 2000). A separate study conducted by Hogan et al. (2005) indicated that increasing dissolved organic matter (DOM) may decrease the toxicity of uranium, as was seen in *Chlorella* sp.

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16 There was a notable difference in toxicity endpoints amongst the minimum detectable effect concentrations (MDEC); these results show higher sensitivity to uranium at the high alkalinity value. However, the authors conclude that "Differences in the slopes of the concentration-response curves of the two alkalinity treatments precluded a reasonable comparison of the MDEC." Hence, only the EC50 values are discussed here.

17 In some cases, complexation of trace metals by humic substances may increase the availability of metals. For example, humic substances may increase the bioavailability of iron, which is otherwise relatively insoluble under typical lake conditions (Aiken et al. 1985).

Decision: There is not enough information on the effects of NOM on uranium toxicity to reliably adjust or normalize toxicity data for this variable. Moreover, the study that investigated the effect of fulvic acid on a freshwater bivalve (Markich et al. 2000) is not applicable for guideline derivation because the study was performed on a non-native species at temperatures higher than Canadian waters. Therefore at this time, the data will not be adjusted for NOM.

8.1.5 Temperature

Uranium toxicity may also theoretically depend on water temperature, via changes in solubility, speciation or kinetics, or in the metabolic rate of the organism; however, there currently are no studies documenting the effect of temperature on uranium speciation or toxicity. Many recent high-quality toxicity tests on uranium have been conducted in Australia, where routine temperatures for testing fish, invertebrate and algae studies are about 27^{o}C. Extreme and elevated temperatures could increase metal toxicity potentially due to elevated energy costs and increased metabolic costs from metal detoxification (Chapman 2008). Suter (1993) suggests that chemical toxicity increases 2–4 times for a 10^{o}C rise in temperature. Also, the Australian and New Zealand Environment and Conservation Council, and Resource Management Council of Australia and New Zealand (ANZECC and ARMCANZ 2000), state that the toxicity of chemicals generally increases with increasing temperature.

Decision: There is not enough information on the effects of temperature on uranium toxicity to reliably adjust or normalize toxicity data for this variable.

8.2 Toxic interactions with other substances and metals

There is one report that investigates the toxic interaction of uranium with other substances, but it focused on human health. This report summarized the joint action of uranium in a mixture with fluoride, nitrate and cyanide, with information summarized from *in vitro* and *in vivo* studies that investigate health-based endpoints in humans and animals, and with physiologically-based pharmacokinetic models (ATSDR 2002). There were no data that examined the toxicity of all four contaminants in one mixture, and furthermore, information on joint action of pairs of these contaminants were lacking. No toxic interactions (either greater-than-additive, or less-than-additive) were reported for the chemical toxicity of uranium; however, weight-of-evidence analysis suggests that cyanide may have a less-than-additive effect on the radiation toxicity of uranium (ATSDR 2002).

One study has documented the toxic interactions of uranium in a metal mixture (Hamilton and Buhl 1997). In this experiment, the formulated metal mixtures were based on environmental concentrations in the San Juan River (New Mexico and Colorado, US); the exposures were short-term, and the test organism was the indigenous flannelmouth sucker, *Catostomus latipinnis*. In three separate mixtures containing varying amounts of copper, zinc, selenium, boron, vanadium, uranium and arsenic, all results exhibited an additive type of joint toxicity, i.e., no synergism or antagonism was observed (Hamilton and Buhl 1997).

A study on the toxic effects of uranium mine-receiving water on fathead minnows (*Pimephales promelas*) investigated metal mixture toxicity as well; however, the focus was on the multivariate technique of principal components analysis, as opposed to toxic interactions of metals (Pyle et al. 2001). Fish were exposed *in situ* for 7 days in Saskatchewan lakes receiving

molybdenum-contaminated mill effluent and nickel-contaminated mine-dewatering effluent; uranium (among other contaminants) was also present in the contaminated lakes, with mean total concentrations ranging from 0.58 to 8.95 µg/L. The toxicity endpoints measured were mortality and growth, and mortality was used in the principal components analysis. Because of the study design, the key conclusions were not related to metal mixture toxicity, but rather co-variation that may be indicative of the contaminants driving the observed differences in fish mortality. The key conclusions were that fish in the molybdenum-contaminated lake showed higher mortality than the nickel-contaminated lake, and that high dietary selenium intake may also be contributing to mortality (Pyle et al. 2001).

8.3 Toxicity to freshwater fish

Notably, for many fish studies, a steep dose-response curve was observed in both short-term and long-term exposures, and in some cases, no partial mortalities were observed (Table 11). The toxicity of uranium to aquatic life has been noted in a number of cases to be dependent on a variety of water quality characteristics, most notably pH and hardness. As previously discussed, the data were not normalized for these, or any other factors, and so it is important to note that differences among studies of similar design and execution may be attributed to water chemistry.

The following information is expanded upon in Table 11, and includes the water quality parameters for each study, as originally reported. For some cases and when raw data were available and were suitable for calculations, no-effect endpoints were calculated to obtain more preferred endpoints for use in the long-term SSD curve, including EC/LC/IC_{10} and maximum acceptable toxicant concentration (MATC).

8.3.1 Short-term toxicity to freshwater fish

As reported in the "Protocol for the Derivation of Canadian Water Quality Guidelines for the Protection of Aquatic Life" (CCME 2007), studies with exposure periods of 96 hours or less for fish, invertebrates and amphibians were considered appropriate for the derivation of a short-term guideline. Most commonly in short-term studies on freshwater fish, 96-h LC_{50} values are reported in the uranium toxicity literature for a wide variety of fish species. At a water hardness of 144 mg $CaCO_3$/L, Hamilton and Buhl (1997) reported a 96-h LC_{50} of 43 500 µg U/L for the flannelmouth sucker (*Catostomus latipinnis*) that was not seen to change over time (from 24 to 96 hours). In brook trout (*Salvelinus fontinalis*), the 96-h LC_{50} has been noted as changing with hardness (Parkhurst et al. 1984), but at similar hardness (32 and 30.8 mg $CaCO_3$/L), values of 5500 µg U/L (Parkhurst et al. 1984) and 8000 µg U/L (Davies 1980), respectively, have been reported.

Rainbow trout (*Oncorhynchus mykiss*) was found to have a 96-h LC_{50} of 6200 µg U/L (Davies 1980), which is similar to the range of 3800–4200 µg U/L found by Vizon Scitech Inc. (2004) at differing hardness values. Bluegill (*Lepomis macrochirus*) were found to be particularly sensitive to uranium in very soft water, with a 96-h LC_{50} of 1670 µg U/L (Trapp 1986). In a study concerning three species of fish, the Colorado squawfish (*Ptychochelius lucius*), the razorback sucker (*Xyrauchen texanus*), and the bonytail (*Gila elegans*), no differences were observed in 96-h LC_{50}s for uranium toxicity when comparing three life stages: swim-up and two sizes of juveniles (Hamilton 1995).

8.3.2 Long-term toxicity to freshwater fish

Exposure periods involving juveniles or adult stages of duration of at least 21 days, or periods involving eggs and larvae of at least 7 days, were considered long-term (CCME 2007). Long-term toxicity tests for uranium are relatively abundant, as seen in Table 11. White sucker (*Catostomus commersoni*) fry, when exposed to uranium for 30 days at a hardness of 72 mg $CaCO_3$/L, were seen to have sublethal effects in growth as indicated by length and dry weight (dw), with a no-observed-effect concentration (NOEC) of 7330 µg U/L, and a lowest-observed-effect concentration (LOEC) of 27 860 µg U/L (Liber et al. 2004b). From those reported endpoints, a MATC of 14 300 µg U/L was calculated. Even at the highest levels of uranium exposure (27 860 µg U/L) mortality was not significantly different from controls.

For survival, hatching success, and time to hatch, lake trout (*Salvelinus namaycush*) had a NOEC of 6050 µg U/L and a LOEC of 29 780 µg U/L after 141 days of exposure, resulting in a calculated MATC of 13 400 µg U/L (Liber et al. 2004a). There was no significant difference in length or dry weight that was related to uranium exposure. Rainbow trout embryos and alevin were more sensitive, with a survival LOEC of 280 µg U/L and EC_{25} of 340 µg U/L after 31 days of exposure (Vizon SciTec Inc. 2004), indicating a marked difference in sensitivity to uranium in different trout species. An EC_{10} of 260 µg U/L was also obtained from the slope of the toxicity relationship.

Fathead minnow (*Pimephales promelas*) embryos exposed to uranium for 7 days have reported NOECs of 810–1200 µg U/L and LOECs of 1300–2000 µg U/L for survival, depending on hardness (Vizon SciTec Inc. 2004). The slopes of the toxicity relationships could be determined from reported LC_{25}s and LC_{50}s. LC_{10}s ranging from 760 to 1300 µg U/L were thus derived using these slopes. From the same studies, fatheads have also been reported to have IC_{25}s (inhibitory concentration) for growth ranging from 1300 to > 2000 µg U/L (Vizon SciTec Inc. 2004).

Liber et al. (2005) have reported that the growth of *Esox lucius* embryos was affected following a 65-day exposure to 4320 µg/L (NOEC). Using the reported NOEC of 1510 µg/L, a MATC of 2550 µg/L could be calculated.

Many of these studies are consistently lower when compared to similar endpoints for short-term studies with the same species. In the case of fathead minnows, there is minimal difference between 96-h and 7-day LC_{50}s (Vizon SciTec Inc. 2004).

8.4 Toxicity to freshwater invertebrates

8.4.1 Short-term toxicity to freshwater invertebrates

Uranium can affect invertebrates in a myriad of ways; effects such as valve closure in bivalves and effects on reproduction have been reported, as well as mortality (Barata et al. 1998; Fournier et al. 2004; Pickett et al. 1993; Poston et al. 1984; Trapp 1986). In the Asiatic clam (*Corbicula fluminea*), valve closure has been reported, with an EC_{50} of 12 µg U/L at a pH of 5.5 and 31 µg U/L at a pH of 6.5 (Fournier et al. 2004). Valve closure could potentially be an ecologically significant endpoint by affecting food intake and survival. In this study, the duration of closure was not reported, so the

relevance as an indicator of toxicity is still in question and it was not classified as an acceptable endpoint.

Toxicity of uranium to *Daphnia magna*, as indicated by 48-h LC_{50}s, varied. Two studies at moderate hardness (66.6–72.9 and 90.7 mg $CaCO_3$/L) resulted in similar 48-h LC_{50}s of 6320 and 6530 µg U/L, respectively (Poston et al. 1984; Barata et al. 1998). *Ceriodaphnia dubia* had reported 48-h LC_{50}s of 60 and 89 µg U/L (Pickett et al. 1993), indicating that it is the most sensitive species studied to date. For depleted uranium desorbed from soil, Kuhne et al. (2002) found a much higher 96-h LC_{50} of 10 500 µg U/L. *Daphnia pulex* had a 48-h LC_{50} of 220 µg U/L (Trapp 1986). While *Daphnia magna* results are more similar to those seen in fish, the results of the other invertebrates indicate that there are more sensitive species that the water quality guideline must be concerned with protecting.

8.4.2 Long-term toxicity to freshwater invertebrates

Studies with test duration longer than 96 hours for short-lived invertebrates and 7 days for longer-lived invertebrates were considered to be long-term tests (CCME 2007). Invertebrates used in long-term studies are often the same as those used in short-term studies, only with longer exposure times. *Hyalella azteca* was found to have an LC_{50} of 21 µg U/L in soft water after 7 days of exposure (Borgmann et al. 2005). When exposed to water hardness ranging from 61 to 238 mg $CaCO_3$/L, the LC_{50}s for *H. azteca* were from 140 to 340 µg U/L and the LC_{25}s ranged from 100 to 130 µg U/L. Calculated LC_{10}s of 55–88 µg U/L were obtained from those endpoints. In another study, Liber et al. (2007) exposed *H. azteca* to uranium for 28 days in water with a hardness of 73 mg $CaCO_3$/L. They obtained an EC_{50} and an EC_{25} of 67 and 27 µg/L, respectively. From these, an EC_{10} of 12 µg/L was calculated.

Daphnia magna has reported LOECs based on reproduction and 21 days of uranium exposure of 520–2250 µg U/L (Poston et al. 1984). Moreover, the reported raw data of the toxicity experiments were used to calculate EC_{10}s of between 123 and 1360 µg U/L. In similar conditions, Liber et al. (2007) obtained an EC_{50} and an EC_{25} of 1250 and 830 µg/L, respectively. The resulting calculated EC_{10} was 570 µg/L.

The midge *Chironomus tentans* was reported to have a 10-day LC_{50} of 6400 µg U/L, with a LOEC of 1519 µg U/L and a NOEC of 421 µg U/L (resulting in a calculated MATC of 800 µg U/L) where the effect was mortality (Burnett and Liber 2006). The same study also found that survival was a more sensitive endpoint for this species than growth, as they found an IC_{50} of 10 200 µg/L for growth. From this study, when compared to the other species reported (Table 11), *C. tentans* were much more tolerant of uranium than other invertebrate species. However, in a more recent study by Liber et al. (2007), a lower 28-day EC_{10} of 930 µg/L for growth was calculated from the reported EC_{25} and EC_{50} (1930 and 4320 µg/L, respectively).

The literature presented in Table 11 shows that there are a wide variety of endpoints for uranium toxicity to *Ceriodaphnia dubia* in particular. In 7-day tests, the LC_{25} was found to be 54-150 µg U/L, depending on the hardness, which ranged from 5 to 252 mg $CaCO_3$/L (Vizon SciTec Inc. 2004). They also reported 7-day NOEC (1970 µg U/L) and LOEC (3910 µg U/L) values, based on neonate production, and IC_{25}s for reproduction ranging from 33 to 79 µg U/L. The use of the toxicity slopes (determined from reported IC_{25}s and IC_{50}s) resulted in the calculation of IC_{10}s of 22–59 µg U/L, depending on water hardness. For waterborne exposures, Pickett et al. (1993) reported a NOEC and a

LOEC for *C. dubia*, values of 1.5 and 2.7 µg U/L respectively, based on reproduction over 7 days. In contrast to these results, Liber et al. (2007) reported an EC_{25} of 2700 µg/L and an EC_{50} of 3970 µg/L when neonate *C. dubia* were exposed to uranium for 7 days. From these results, an EC_{10} of 1900 was calculated. Differences in dilution water hardness can not explain the observed discrepancy since Liber et al. (2007) used a water hardness of 76 mg $CaCO_3$/L, which was in the range of water hardness used in Vizon SciTec Inc. 2004 experiments.

8.5 Toxicity to freshwater aquatic plants

8.5.1 Long-term toxicity to freshwater plants

While there have been a number of studies on the effects of uranium on the tropical freshwater algae *Chlorella* sp. (Table 11), those studies are not able to be used in the derivation of a Canadian water quality guideline as the tests were run at temperatures higher than those found in Canadian waters (>25°C) and there are no comparable *Chlorella* sp. studies in more representative waters, and so these studies are not considered indicative of potential toxicity in Canada.

Studies reported in this document pertaining to the toxicity of uranium to algae and/or aquatic plants were classified as long-term studies, as per the CWQG protocol (CCME 2007).

There are other studies present, however, concerning the macrophyte *Lemna minor* and *Pseudokirchneriella subcapitata* (reported elsewhere as *Selenastrum capricornutum*). Toxicity of *Lemna minor* is reported as 7-day IC_{25}s for frond number (4700–12 300 µg U/L, depending on hardness) and for dry weight (6400–13 300 µg U/L, depending on hardness) (Vizon SciTec Inc. 2004). The calculated IC_{10}s for frond number and dry weight were 3400 µg U/L and 3100 µg U/L, respectively. *Pseudokirchneriella subcapitata* has much lower toxicity thresholds, based on inhibition of growth. The IC_{25}s varied from 27 to 150 µg U/L, depending on hardness, and the NOEC (14–220 µg U/L) and LOEC (29–430 µg U/L) values were also reported (Vizon SciTec Inc. 2004). IC_{10}s ranging from 5.4 to 120 µg U/L were calculated from the slope of the toxicity relationship.

8.6 Toxicity to marine life

Only one study on uranium toxicity to a marine organism was found. *Allorchestes compressa*, a marine amphipod tested in Australia, was exposed to depleted uranyl nitrate ($UO_2(NO_3)_2.6\ H_2O$) for approximately 10 weeks (Ahsanullah and Williams 1986). Respiration rate (in males only), growth, reproduction, survival and male/female ratio were the measured endpoints. For the limited concentration range tested (control, 100 µg/L, 1000 µg/L and 2000 µg/L), respiration seemed to be the most sensitive endpoint, although response was not entirely consistent with dose. At 100 µg/L, the respiration rate (averaged over 3 generations) decreased by about 41% when compared with control values (Ahsanullah and Williams 1986).

9.0 DERIVATION OF CANADIAN WATER QUALITY GUIDELINES

9.1 Existing water quality guidelines to protect aquatic life

The recommended water quality objective for uranium in Canada, established by the Inland Waters Directorate, Water Quality Branch, in relation to aquatic life and wildlife, was set at 300 µg U/L in 1983 (Environment Canada 1983). Provincial and international guidelines for uranium range from 0.5 to 100 µg U/L (Table 12).

Sheppard et al. (2005) derived predicted no-effect concentration (PNEC) values for uranium toxicity from published literature. The values obtained were not adjusted using a safety factor, but derived directly from the literature and are not official guidelines. The values they obtained are as follows, all for freshwater organisms:

Benthos: 100 000 µg U/kg dry sediment
Plants: 5 µg U/L water
Invertebrates: 5 µg U/L water
Fish at water hardness of: < 10 mg $CaCO_3$/L (very soft water) – 100 µg U/L water
10–100 mg $CaCO_3$/L (soft water) – 2800 µg U/L water
> 100 mg $CaCO_3$/L (hard water) – 23 000 µg U/L water

9.2 Adjusting toxicity endpoints for water chemistry conditions

The water chemistry of uranium is very complex, and the specific forms and concentrations of the various uranium species is strongly determined by water characteristics such as pH, alkalinity, temperature, NOM and hardness. While it is suspected that uranium speciation will affect its toxicity, at this time there is insufficient information available to quantitatively evaluate the influence of these toxicity-modifying factors, and consequently, they were not taken into account during guideline derivation.

9.3 Derivation of guidelines

A CWQG for uranium is required to address its use in Canada and potential impacts to aquatic systems. A CWQG is required to provide guidance to risk assessors and risk managers in Canada on the level of uranium in an aquatic system below which no adverse toxic effects are expected on aquatic plants and animals.

There are currently three options for developing a CWQG (CCME 2007):

1. Statistical approach (Type A or SSD approach) using primary and/or secondary data
2. Lowest endpoint approach using only primary data (Type B1);
3. Lowest endpoint approach using primary and/or secondary data (Type B2)

The minimum data requirements for each of the three methods are presented in Tables 13 and 14 for short-term and long-term exposure guidelines, respectively. An SSD is a statistical distribution that captures the variation in toxicological sensitivity to a contaminant among a given set of species. The species sensitivity distribution, often expressed as a cumulative distribution function (CDF), is

composed of effect concentrations obtained during toxicity testing (e.g., LC_{50}, EC_{50}, LOEL or NOEL) on the x-axis and cumulative probability on the y-axis (Posthuma et al. 2002). The number of data points used to construct the curve depends on the number of species tested for the endpoint of interest. Emphasis is placed on organism-level effects (e.g., survival, growth, reproduction) that can be more confidently used to predict ecologically significant consequences at the population level (Meador 2000; Forbes and Calow 1999; Suter et al. 2005). With the SSD method, the concentration of a substance in water that will be protective of at least 95% of aquatic biota is estimated.

If insufficient data are available for deriving a CWQG using the statistical approach, the CWQG can be developed using the lowest endpoint approach. Depending on the quantity and quality of data a Type B1 or Type B2 approach is used. The Type B1 approach uses only acceptable primary toxicity data to derive the guideline, while the Type B2 approach can use acceptable primary and/or secondary data. In every case, a CWQG must be developed using the most advanced method that the data allow.

The following sections describe the derivation of a CWQG for the protection of freshwater life in surface water for uranium. The derived CWQG is national in scope and does not take into account watershed-specific conditions.

Aquatic toxicity studies meeting the requirements of primary or secondary classification based on the CCME (2007) protocol are presented in Table 11. These studies represent data available for CWQG derivation.

A CWQG consists of guidance for both short- and long-term exposure. The short-term guidance offered by the CWQG is not intended to protect all components of aquatic ecosystem function. The purpose of the short-term exposure value of the CWQG is to protect most species against lethality during severe but transient events such as inappropriate application or disposal of the substance in question. This may include application under worst-case conditions and/or through improper use of label instructions (e.g., heavy precipitation/wind events), and spill events. The long-term exposure value of the CWQG is intended to protect against negative effects to aquatic organisms during indefinite exposures.

9.4 Short-term CWQG

To be considered for inclusion in CWQG development, the aquatic toxicity studies must meet minimum data quality requirements as specified in the water quality protocol (CCME 2007; Table 13). Both primary and secondary data as described in the protocol (CCME 2007) were considered acceptable for deriving the generic SSD for uranium.

Several of the studies reported in Table 11 are for the same species, effect, endpoint or life stage, though the LC_{50}s are different. This variation may be the result of differences in experimental conditions, species strain, and/or bioassay protocol. Multiple bioassay results for the same species should not be used in an SSD regression analysis. This is particularly important when there is a large amount of data available for very few test species. There are numerous methods that can be applied to account for multiple results for a single species (Duboudin et al. 2004). For the derivation of an SSD for uranium, intra-species variability was accounted for by taking the geometric mean of the studies considered to represent the most sensitive lifestage and endpoint. Geometric mean values were calculated for *Ceriodaphnia dubia, Daphnia magna, Pimephales promelas, Oncorhynchus mykiss* and

Salvelinus fontinalis. For *C. dubia*, *D. magna* and *S. fontinalis*, geometric means for similar endpoints were calculated for different water hardnesses, and the lowest geomean for each species was selected. For *P. promelas* and *O. mykiss*, geomeans of similar endpoints were calculated over a range of water hardnesses because no effect on toxicity was apparent (Table 10). Table 15 presents the final dataset that was used to generate the fitted SSD for uranium.

The values reported in Table 15 range from a 48-h LC_{50} of 72 µg/L for the water flea *Ceriodaphnia dubia*, to a 96-h LC_{50} of 46 000 µg/L for *Gila elegans*, *Ptychocheilus lucius* and *Xyrauchen texanus*.

The short-term SSD is preferentially derived from LC/EC_{50} data for short-term effects (Table 13). The final CWQG value for uranium was the 5th percentile of the short-term SSD.

Each species for which appropriate short-term toxicity data were available was ranked according to sensitivity, and its centralized position on the SSD was determined using the following Hazen standard equation (Aldenberg et al. 2002; Newman et al. 2002):

$$\frac{i - 0.5}{N}$$

where

i = the species rank based on ascending EC_{50}s and LC_{50}s
N = the total number of species included in the SSD derivation

These positional rankings, along with their corresponding EC_{50} and LC_{50}s, were used to derive the SSD. Several cumulative distribution functions (CDFs) (normal, logistic, Gompertz, Weibull, and Fisher-Tippett) were fit to the data (both in arithmetic space and log space) using regression methods. Model fit was assessed using statistical and graphical techniques. The best model was selected based on consideration of goodness-of-fit and model feasibility. Model assumptions were verified graphically and with statistical tests.

The log-Gompertz model provided the best fit of the ten models tested based on the Anderson-Darling Statistic ($A^2 = 0.437$); fiducial limits, and visual inspection among other factors (e.g., residuals). The equation of the Gompertz model is of the form:

$$f(x) = 1 - e^{-e^{\frac{(x-\mu)}{s}}}$$

For the fitted model x = log (concentration) of uranium (µg/L), $\mu = 4.15$ and $s = 0.88$.
Summary statistics for the short-term SSD are presented in Table 16. The 5th percentile on the short-term SSD is 33 µg $U \cdot L^{-1}$. The lower fiducial limit (5%) on the 5th percentile is 9 µg $U \cdot L^{-1}$, and the upper fiducial limit (95%) on the 5th percentile is 130 µg $U \cdot L^{-1}$. The concentration of 33 µg $U \cdot L^{-1}$, is beyond the range of the data (to which the model was fit). Therefore, the 5th percentile and its confidence limits are extrapolations.

9.5 Long-term CWQG

To be considered for inclusion in CWQG development, the aquatic toxicity studies must meet minimum data quality requirements as specified in the water quality protocol (CCME 2007; Table 14). Both primary and secondary data as described in the protocol (CCME 2007) were considered acceptable for deriving the generic SSD for uranium.

Several of the studies reported in Table 11 are for the same species, effect, endpoint or life stage, though the endpoints are different. This variation may be the result of differences in experimental conditions, species strain, and/or bioassay protocol. Multiple bioassay results for the same species should not be used in an SSD regression analysis. This is particularly important when there is a large amount of data available for very few test species. There are numerous methods that can be applied to account for multiple results for a single species (Duboudin et al. 2004). For the derivation of an SSD for uranium, intra-species variability was accounted for by taking the geometric mean of the studies considered to represent the most sensitive lifestage and endpoint. Geometric mean values were calculated for *Ceriodaphnia dubia*, *Daphnia magna*, *Oncorhynchus mykiss*, *Pimephales promelas* and *Pseudokirchneriella subcapitata*. Effect concentrations reported for *Hyalella azteca*, *Cryptomonas erosa*, *Simocephalus serrulatus*, *Chironomus tentans*, *Esox lucius*, *Lemna minor*, *Salvelinus namaycush* and *Catostomus commersoni* were taken from single studies. Table 17 presents the final dataset that was used to generate the fitted SSD for uranium.

The values reported in Table 17 range from a 28-day EC_{10} of 12 µg/L for the amphipod *A. azteca*, to a 30-day MATC of 14 300 µg/L for the white sucker, *C. commersoni*.

The long-term SSD is preferentially derived from no-effects data for long-term effects (Table 14). The preferred endpoints for use in the SSD are as in the following data points hierarchy:

EC_x/IC_x representing a no-effects threshold > EC_{10}/IC_{10} > MATC > NOEC > $EC_{11\text{-}25}/IC_{11\text{-}25}$ > LOEC > $EC_{26\text{-}49}/IC_{26\text{-}49}$ > nonlethal EC_{50}/IC_{50}.

where the four first endpoints are considered as “no-effect” endpoints. The others are classified as “low-effect” endpoints. The SSD for long-term toxicity of uranium only contains “no-effect” endpoints. The final CWQG value for uranium was the 5th percentile of the long-term SSD.

Each species for which appropriate long-term toxicity data were available was ranked according to sensitivity, and its centralized position on the SSD was determined using the following Hazen standard equation (Aldenberg et al. 2002; Newman et al. 2002):

$$\frac{i - 0.5}{N}$$

where

i = the species rank based on ascending no-effect endpoints
N = the total number of species included in the SSD derivation

These positional rankings, along with their corresponding no-effect endpoints were used to derive the SSD. Several cumulative distribution functions (CDFs) (normal, logistic, Gompertz, Weibull and Fisher-Tippett) were fit to the data (both in arithmetic space and log space) using regression methods. Model fit was assessed using statistical and graphical techniques. The best model was selected based on consideration of goodness-of-fit and model feasibility. Model assumptions were verified graphically and with statistical tests.

The logistic model provided the best fit of the ten models tested considering the Anderson-Darling Statistic ($A^2 = 0.145$), fiducial limits, and visual inspection among other factors (e.g., residuals). The equation of the fitted logistic model is of the form:

$$f(x) = \frac{1}{1 + e(-\frac{x - \mu}{\sigma})}$$

Where, in the case of the fitted model, x = log (concentration) of uranium (µg/L), $\mu = 2.780$, and $\sigma = 0.548$.

Summary statistics for the long-term SSD are presented in Table 18. The 5th percentile on the long-term SSD is 15 µg $U \cdot L^{-1}$. The lower fiducial limit (5%) on the 5th percentile is 8.5 µg U/L, and the upper fiducial limit (95%) on the 5th percentile is 25 µg U/L.

9.6 Use of the guideline in water quality management

The CWQG for uranium is set to provide protection for short- and long-term exposure periods. They are based on generic environmental fate and behaviour and toxicity data. The guideline is a conservative value below which all forms of aquatic life, during all life stages and in all Canadian aquatic systems, should be protected. Because the guideline is not corrected for any toxicity modifying factors, it is a generic value that does not take into account any site-specific factors. Moreover, since it is mostly based on toxicity tests using naïve (i.e., non-tolerant) laboratory organisms, the guideline may not be relevant for areas with naturally elevated concentration of uranium with adapted ecological community (CCME 2007). Thus, if an exceedence of the guideline is observed (due to anthropogenically enriched water or because of elevated natural background concentrations), it does not necessarily suggest that toxic effects will be observed, but rather indicates the need to determine whether or not there is a potential for adverse environmental effects. In some situations, such as where an exceedence is observed, it may be necessary or advantageous to derive a site-specific guideline that take into account local conditions (water chemistry, natural background concentration, genetically adapted organisms, community structure) (CCME 2007).

The guideline should be used as a screening and management tool to ensure that uranium does not lead to the degradation of the aquatic environment. The CWQG for uranium could, for example, be the basis for the derivation of site-specific guidelines and objectives (derived with site-specific data as well as consideration of technological, site-specific, socioeconomic or management factors) (CCME 2007).

9.7 Comparisons to other guidelines and protective concentrations

The PNEC values derived by Sheppard et al. (2005) for freshwater fish are higher than the guideline values; this is due primarily to different criteria and derivation processes, a more specific by-taxa grouping, as well as an increase in the amount of data available. Many of the studies used to calculate those values were non-resident species under unacceptable conditions for guideline derivation. The PNEC for freshwater plants was derived using a safety factor on the geometric mean of the EC_{25}s, and more values were found for national guideline derivation. Where freshwater fish are concerned, as stated, hardness was not found to be a reliable modifying factor at this time. There were also other studies included in the guideline derivation that had values lower than those reported in Sheppard et al. (2005), and many of those used to derive PNEC values were unacceptable for CWQG derivation.

As the CWQG is meant to protect all aquatic life, the value is determined using more tolerant species, such as fish species, as well as very sensitive species, such as algae and invertebrates. All of these values are taken into account to provide one value that is protective for all species; this value is lower, but still within the ranges of PNEC values determined by Sheppard et al. (2005). The long-term CWQG value of 15 μg U/L is similar to water quality guidelines from other jurisdictions (Table 12), except for Australia and New Zealand; however, that guideline is one of "low reliability" (ANZECC and ARMCANZ 2000).

10.0 GUIDELINE SUMMARY

The short-term data met the toxicological and statistical requirements for the Type A guideline derivation method. The Gompertz model was used for short-term guideline derivation. As seen in Table 15, the data requirements for the SSD were surpassed, and a total of 11 data points from 11 species were used in the derivation of the guideline. LC_{50} values from each species were used in derivation.

The long-term data met the toxicological and statistical requirements for the Type A guideline derivation method. The logistic model was used for long-term guideline derivation. As seen in Table 17, the data requirements for the SSD were surpassed, and a total of 13 data points from 13 species were used in the derivation of the guideline.

There were not enough data to produce CWQG values for long-term or short-term exposures in marine settings.

The following Canadian water quality guidelines (CWQGs) are recommended to protect aquatic biota from harmful exposure to uranium compounds in water.

Canadian Water Quality Guidelines for the Protection of Aquatic Life

	Long-term exposure guideline (µg U/L)	Short-term exposure guideline (µg U/L)
Freshwater	15	33
Marine	NRG	NRG

NRG = no recommended guideline

TABLES

Table 1. Description of the radioactive properties of the isotopes of uranium

Isotope	Relative abundance[1]	Half-life[1]	Relative radioactivity	Decay series includes[2]
^{238}U	99.274%	4.47×10^{9}	48.9%	^{234}U, radon ^{222}Rn, lead ^{206}Pb
^{235}U	0.7202%	7.08×10^{8}	2.2%	radon ^{219}Rn, lead ^{207}Pb
^{234}U	0.0057%	2.45×10^{5}	48.9%	radon ^{222}Rn, lead ^{206}Pb

[1] Clark et al. (1997)
[2] ATSDR (1999). Full list of uranium decay series found here.

Table 2. Selected physical and chemical properties of uranium compounds

Property	Elemental uranium	Uranium (IV) dioxide	Uranium (IV) trioxide	Uranium (V,VI) oxide[1] (triuranium octaoxide)	Uranyl sulphate trihydrate	Uranyl nitrate hexahydrate	Uranyl acetate, dihydrate
CAS RN	7440-61-1	13344-57-6	1344-58-7	1344-59-8	20910-28-5	13520-83-7	541-09-3
Molecular formula	U	UO_2	UO_3	U_3O_8	$UO_2SO_4.3H_2O$	$UO_2(NO_3)_2.6H_2O$	$UO_2(CH_3OO)_2.2H_2O$
Molecular weight	238.029	270.028	286.027	842.082	420.138	502.129	424.15
Solubility in water (g/100 g H_2O)	Insoluble	Insoluble	Insoluble	—	Soluble	Soluble 127	Soluble 7.7 g/100 ml at 15°C
Solubility in acid	Soluble	Insoluble in dilute acid, Soluble in concentrated acid	Soluble in acid	—	152	—	—-
Boiling point (°C)	4131	—		—	—	118[3]	Decomposes at 275
Melting point (°C)	1135	2827		1300[3]	—	60	Loses 2 H_2O at 110
Density (g/cm^3)	19.1[2]	10.97	7.3	8.38	3.28	2.81	2.893 at 15°C

Summarized from various sources (ATSDR 1999; Budavari et al. 1996; Lide 2002)

— indicates no information was stated.

[1] This is the extracted and processed uranium termed (with other oxides) as yellowcake (Environment Canada and Health Canada 2003).

[2] Highest density of any naturally occurring element

[3] Decomposes at this temperature

Table 3. Most common uranium detection methods with detection limits

Method	Reported detection limit	Comments
Spectrophotometric (e.g., atomic absorption, atomic emission, fluorometric, phosphorescence)	0.01–20 μg/L, with one report of up to 10 500 μg/L	Solid fluorometry has been cited as the most common detection method.
Inductively coupled plasma with mass spectrometry	0.0003–0.1 μg/L	Isotope distinction possible. Method recommended in *Standard Methods for the Examination of Water and Wastewater* (APHA et al. 1981, 1985, 2005)
Gross alpha (α) or alpha spectrometry[1]	0.009–1 pCi/L	Isotope distinction possible. Recovery can be highly variable. Analyses based on total alpha activity or alpha spectrometry is recommended in *Standard Methods for the Examination of Water and Wastewater* (APHA et al. 1981, 1985, 2005)
Neutron activation analysis or instrumental neutron activation analysis	0.4 μg/L, 3 μg/L	

Analysis is for total uranium only.
Summarized from various sources (APHA et al. 2005; ATSDR 1999; Federal-Provincial-Territorial Committee on Drinking Water 2001).
[1] This method cannot be used for comparison with the guidelines, as is quantifies radioactivity and not chemical concentration.

Table 4. Selected uranium mining operations in Canada

Mine	Location	Status (as of 2008)	Production in million pounds of U_3O_8 from 2008	Company[1]
Key Lake and McArthur River[2] operation	Northern Saskatchewan	Producing Operating for about 20 years	16.6	Cameco Corporation
McClean Lake mine	Northern Saskatchewan	Producing Operating since 1999–2000	3.2	AREVA Resources Canada
Rabbit Lake operation	Northern Saskatchewan	Producing Operating for about 20 years	3.6	Cameco Corporation
Cigar Lake	Northern Saskatchewan	Approved for opening Permission to proceed with U mining operations granted in 1998 Construction began in 2005	Expected: 18 x 10^{66} lbs. U_3O_8 per year	Cameco Corporation
Midwest	Northern Saskatchewan	Approved for opening Permission to proceed with U mining operations granted in 1998	n/a	AREVA Resources Canada
Cluff Lake	Northern Saskatchewan	Closed since 2002 Operated for about 20 years Decommissioning licence was renewed by the Canadian Nuclear Safety Commission until 2019	n/a	AREVA Resources Canada
Elliot Lake	Ontario	Closed Operated between 1955 and 1996 Request for decommission granted Management and containment of tailings underway	n/a	Rio Algom and Denison Mines

Summarized from various sources (CAMECO Corporation 2009; CEAA 2004; AREVA Resources Canada 2009; Environment Canada and Health Canada 2003; Giancola 2003).

[1] In many cases, Cameco, AREVA and others partially own the mining operations. Majority owner is specified here.

[2] Ore from McArthur River is trucked to Key Lake where it is mixed with lower-grade ore that has been stockpiled from the Key Lake mine. This "blended" ore is then fed into yellowcake.

Table 5. Background concentrations of uranium across Canada[1]

Location	Number of data (N)	Minimum[4] (µg U/L)	Maximum (µg U/L)	Median (µg U/L)
Lakes				
Canada	68 303	< 0.05	1350	< 0.05
Alberta	1160	< 0.05	1.2	< 0.05
British Columbia	692	< 0.05	4.4	< 0.05
Manitoba	13 969	< 0.05	170	< 0.05
New Brunswick	336	< 0.05	2.02	< 0.05
Newfoundland and Labrador	17 665	< 0.05	4.62	< 0.05
Nunavut	7848	< 0.05	18.6	0.11
Ontario	17 098	< 0.05	30	< 0.05
Saskatchewan	9331	< 0.05	1350	< 0.05
Yukon	204	< 0.05	0.99	0.12
Streams				
Canada	75 471	< 0.05	255	0.06
Alberta[3]	1643	0.006	0.704	0.54
British Columbia	38 666	< 0.05	69	0.05
New Brunswick	7261	< 0.05	7.5	< 0.05
Newfoundland and Labrador	1365	< 0.05	5	< 0.05
Northwest Territories	645	< 0.05	6.4	0.25
Nunavut	415	< 0.05	8.5	0.15
Ontario	198	< 0.05	2.78	< 0.05
Quebec[2]	375	< 0.0009	3.3	0.09
Yukon	26 921	< 0.05	255	0.15

[1] Data supplied by RG Garrett (pers. comm. 2007), summarized from NGR and URP data from the Geological Survey of Canada except for Quebec.
[2] Data (2004–2008) supplied by Quebec's Ministère du Développement Durable, de l'Environnement et des Parcs (Isabelle Guay, pers. comm.).
[3] Data (1998–2008) supplied by Alberta Environment (Kim Westcott, pers. comm.).
[4] The minimum value was often found to be at or below the detection limit. In these cases, the value represented in the table is the detection limit, though the actual value is likely lower.

Table 6. Uranium concentrations in surface waters[1] of mining sites, refineries and waste management facilities

Location	Status of facility	Water bodies	Range of U in surface water (µg/L)	Reference
Mines				
Rabbit Lake (SK)	Active	Upper Link Lake, Lower Link Lake, Pow Bay, Hidden Bay, Horseshoe Lake, Collins Bay Eagle Point	0.52–1061	(Environment Canada and Health Canada 2003)
Key Lake (SK)	Active	Horsefly Lake, McDonald Lake, Little McDonald Lake, Delta Lake, David Creek	2–38	(Environment Canada and Health Canada 2003)
McClean Lake (SK)[2]	Active	Vulture Lake, McClean Lake, Kewen Lake	0.11–2.23	(Environment Canada and Health Canada 2003)
McArthur River (SK)[2]	Active	Boomerang Lake, Lucy Lake, Little Yalowega Lake, Yalowega Lake	1.36–4.16	(Environment Canada and Health Canada 2003)
McClean Lake (SK)	New mine	Vulture Lake	2.23	(Environment Canada and Health Canada 2003)
McArthur River (SK)	New mine	Boomerang Lake	4.16	(Environment Canada and Health Canada 2003)
Cigar Lake (SK)	Approved for opening	No info	No info	n/a
Midwest (SK)	Approved for opening	No info	No info	n/a
Cluff Lake (SK)	Closed	Island Lake, Snake Lake, Cluff Lake, Agnes Lake	1–248	(Environment Canada and Health Canada 2003)

Location	Status of facility	Water bodies	Range of U in surface water (µg/L)	Reference
Langley Bay (SK)	Closed	Lake Athabaska	2.3–101.7	(Waite et al. 1988)
Beaverlodge (SK)	Decommissioned	Beaverlodge Lake, Dubyna Lake, Ace Creek, Greer Lake, Marie Lake	59–649	(Environment Canada and Health Canada 2003)
Beaverlodge (SK)[3]	Decommissioned	-	193–600	(Swanson 1985)
Serpent River Watershed/Elliot Lake (ON)	Decommissioned	Hough Lake, Ten Mile Lake, Dunlop Lake, McCabe Lake, Quirke Lake, Kindle Lake, Elliot Lake, Whiskey Lake, Nordic Lake, Pecors Lake, McCarthy Lake	0.5–15.3	(Environment Canada and Health Canada 2003)
Refineries				
Blind River (ON)	Operating	Blind River	0.6	(Environment Canada and Health Canada 2003)
Port Hope (ON)	Operating	Port Hope Harbour	10	(Environment Canada and Health Canada 2003)
Waste Management Facilities				
Port Granby (ON)[3]	Operating	Port Gramby	10–900	(Environment Canada and Health Canada 2003)
Welcome (ON)[3]	Operating	Welcome	18–94	(Environment Canada and Health Canada 2003)

[1] Lakes and creeks have been grouped together for this table.
[2] Concentration ranges reported here are not based on average lake concentrations, but on predicted peak concentrations based on measured treated effluent quality.
[3] Subsamples at different locations of the same water body were treated as separate water bodies.

Table 7. BAFs and tissue concentrations of uranium in freshwater organisms

Species	U in water (µg U/L)	U in organism (µg U/kg)	BAF	Other notes	Reference
Salvelinus namaycush (lake trout)	1–35 (measured) at five different lakes	Bone: 180–4400 Muscle: < 50 Gut contents: 610–1700 Dry weight	--	Calculated from field, so U tissue accumulation likely includes U from food. Only dry weight measurements reported.	(Clulow et al. 1998)
Coregonus clupeaformis and *Prosopium cylindraceum* (whitefish)	1–35 (measured) at five different lakes	Bone: 1280–14 600 Muscle: <50–120 Gut contents: 820–10 290 Dry weight	--	Calculated from field, so U tissue accumulation likely includes U from food.	(Clulow et al. 1998)
Salmo trutta (brown trout)	1–39 at two different field sites	0.15–89 muscle-skin Wet weight	0.08–5.9	Calculated from field, so U tissue accumulation likely includes U from food. BAFs varied greatly between two sites tested	(Parkhurst et al. 1984)
Salvelinus fontinalis (brook trout)	230–9080 (measured) in lab dose regime	1100–18 000 Whole body Wet weight	1.94–4.28	Calculated after 60-day early life stage laboratory test. BCFs significantly decreased with increasing U exposure.	(Parkhurst et al. 1984)
Brachydanio rerio (zebra danio)	151	150–1040 Whole body Wet weight	0.00887	Fish exposed for 28 days. Maximum accumulation reported at day 28. BCF calculated using max tissue accumulation.	(Labrot et al. 1999)
Oncorhynchus mykiss (rainbow trout)	0.0776 and 963 (measured)	2.26 and 10 310 dry weight	37.2 and 19.8 based on dry weight accumulation[1]	Only dry weight accumulations reported. Author states that absorption through gut is the more likely exposure route (as compared with gill absorption). Author uses *Salmo gairdneri* as species name.	(Poston 1982)

Species	U in water (µg U/L)	U in organism (µg U/kg)	BAF	Other notes	Reference
Coregonus clupeaformis (whitefish)	-- Only concentration in food reported 85500 µg U/kg 982000 µg U/kg 9892000 µg U/kg all measured	@ 9 852 000 µg U/kg exposure, scales: 75 000 bone: 11 3000 gonads: 123 000 Wet weight	--	Tissue accumulation monitored at 10, 30 and 100 days. Highest accumulation in scales, bones and gonads; highest values reported.	(Cooley and Klaverkamp 2000)
Coregonus clupeaformis (whitefish)	2.3	Most samples 50–1700 Gut content: 7500 Wet weight	--	Calculated from field, so U tissue accumulation likely includes U from food. U in water averaged over 3 days. U concentrations in different tissues measured.	(Waite et al. 1988)
Esox lucius (northern pike)	2.3	Most samples below detection limit (< 600) Gut content: 2200 Wet weight	--	Calculated from field, so U tissue accumulation likely includes U from food. U in water averaged over 3 days. U concentrations in different tissues measured.	(Waite et al. 1988)
Corbicula sp. (mollusc; clam)	93 500	DL[2]–26 970 Whole body Wet weight	0.05567	Clam exposed for 28 days. Maximum accumulation reported at Day 19. Tissue distribution among foot, visceral mass, gills and remaining tissue also reported. BCF calculated using max tissue accumulation.	(Labrot et al. 1999)
Corbicula fluminea (Asiatic clam)	63	10 whole body Wet weight	160 (42-day exposure, not including day 2)	Toxicity of uranium potentially occurs in the digestive gland. Gills accumulated more U in high exposures; visceral mass accumulated more U in lower, environmentally relevant exposure concentrations. pH = 7.0 for reported values. Other concentrations, pHs, and durations tested, but U in tissues not reported as definitive numbers; shown in graphs.	(Simon and Garnier-Laplace 2004)

Species	U in water (µg U/L)	U in organism (µg U/kg)	BAF	Other notes	Reference
Homarus americanus (lobster)[3]	--	10.34 in controls 14.52–95.03 in contaminated water Wet weight	--	Lobster samples collected at a control site and an impacted site (a harbour in New Brunswick that has a lead smelter, fertilizer plant and coal-fired power station).	(Chou and Uthe 1995)
Potamogeton sp. (macrophyte) with control site	7.8 Control: 0.2	2600 Control: 1400 Wet weight	330 Control: 7000		(Waite et al. 1988)
Myriophyllum sp. (macrophyte) with control site	7.8 Control: 0.2	3400 Control: 3800 Wet weight	440 Control: 19 000		(Waite et al. 1988)
Typha sp, roots and stems (macrophyte)	--	100 (stem) 1700 (roots) Wet weight	--	Location of *Typha* collection not specified, therefore do not have measured concentration of U in water.	(Waite et al. 1988)

[1] Original article reports "concentration ratios," not BCFs. No calculation formula is given, but these metrics are assumed to be the same. Concentration ratios are based on highest observed tissue concentration.
[2] DL = detection limit. Not stated in the original report.
[3] Marine species.

Table 8. Generalized comparison of bioaccumulation of organic contaminants (produced and released in the 1970s) and metals

Criteria	Organic contaminants (circa 1970)	Metals
Physical-chemical characteristics	Generally lipophilic and neutral Toxicity not as widely affected by water chemistry	Generally ionic. Toxicity typically affected by water chemistry.
Origin	Mostly anthropogenic	All naturally occurring, with anthropogenic redistribution.
Biological uptake	By diffusion, driven by degree of lipophilicity	Usually active uptake and regulation, typically through endogenous transporters. Within the organism, metals may be stored in detoxified forms, so bioaccumulation may be a poor predictor of toxicity.
Essentiality to life	Not essential	Some metals are nutrients at low concentrations, and are therefore essential to life. However, all aquatic biota accumulate trace metals in their tissues, regardless of essentiality or non-essentiality.
BAFs dependence on exposure concentration	Independent. Because BAFs are intrinsic properties of the contaminant, BAFs can be used in hazard identification. High BAFs generally indicates high hazard.	Dependent. Because metals are actively regulated, BAFs are not constant across exposure concentrations. BAFs generally decrease with increasing exposure concentrations, so high BAFs do not necessarily indicate high hazard.

Summarized from various sources (Brix and DeForest 2000, 2003; McGeer et al. 2003; Rainbow 2002).

Table 9. Selected water chemistry characteristics of Ontario Lakes on the Canadian Shield

Water chemistry parameter	Geometric mean	Geometric standard deviation	Number of lakes used
Calcium (mg/L)	4.73	2.46	3841
Magnesium (mg/L)	1.17	2.27	3738
Hardness[1] (mg/L as $CaCO_3$)	16.63[1]	--	
Alkalinity (mg/L as $CaCO_3$)	4.56	16.12	6023
pH	6.64	1.14	6089
Dissolved organic carbon (mg/L)	4.71	2.26	2713

All data from a report from the Atomic Energy of Canada Limited (Bird and Schwartz 1997)
[1]Calculated based on the formula: 2.497 [Ca^{2+} mg/L] + 4.118 [Mg^{2+} mg/L] (APHA et al. 2005)

Table 10. Summary of studies that investigated hardness as a toxicity-modifying factor

Taxa/organism	Short-term or long-term	Toxicity Endpoint	Effective concentration (µg/L U)	Hardness (as mg/L $CaCO_3$)	Effect of hardness on toxicity[1]	Comments	Reference
Fish (short-term and long-term)							
Fathead minnow *Pimephales promelas*	Short-term (96 h)	LC_{50}	2800	20	Substantial effect of hardness on toxicity. A 20-fold increase in hardness results in a 4.8-fold decrease in toxicity.	This study may not be reliable, and true hardness may be confounded with alkalinity and pH.	(Tarzwell and Henderson 1960)
		LC_{50}	135 000	400			
Fathead minnow *Pimephales promelas*	Short-term (96 h)	LC_{50}	2000	23	No apparent effect of hardness on toxicity.	True hardness isolated from alkalinity and pH.	(Vizon SciTec Inc. 2004)
		LC_{50}	2000	72			
		LC_{50}	2100	131			
		LC_{50}	1800	244			
Brook trout *Salvelinus fontinalis*	Short-term (96 h)	LC_{50}	5500	32	Substantial effect of hardness on toxicity. A 6.5-fold increase in hardness results in a 4.2-fold decrease in toxicity.	True hardness is confounded with alkalinity and pH.	(Parkhurst et al. 1984)
		LC_{50}	23 000	210			
Rainbow trout *Oncorhynchus mykiss*	Short-term (96 h)	LC_{50}	4200	20	No apparent effect of hardness on toxicity.	True hardness isolated from alkalinity and pH.	(Vizon SciTec Inc. 2004)
		LC_{50}	3900	68			
		LC_{50}	4000	126			
		LC_{50}	3800	243			
Fathead minnow *Pimephales promelas*	Long-term (early life stage, 7 days)	LC_{50}	1600	23	No apparent effect of hardness on toxicity.	True hardness isolated from alkalinity and pH.	(Vizon SciTec Inc. 2004)
		LC_{50}	2100	72			
		LC_{50}	2000	131			
		LC_{50}	1500	244			
Rainbow trout *Oncorhynchus mykiss*	Long-term (early life stage, 30/31 days)	EC_{50}	460	6	Minor effect of hardness on toxicity. A 12-fold increase in hardness resulted in a 2.2-fold decrease in toxicity.	True hardness isolated from alkalinity and pH.	(Vizon SciTec Inc. 2004)
		EC_{50}	640	61			
Invertebrates (short-term and long-term)							
Water flea	Short-	LC_{50}	6530†	90.7	Substantial effect of hardness on	True hardness is confounded	(Barata et al.

Taxa/organism	Short-term or long-term	Toxicity Endpoint	Effective concentration (µg/L U)	Hardness (as mg/L $CaCO_3$)	Effect of hardness on toxicity[1]	Comments	Reference
Daphnia magna	term (48 h)	LC_{50}	18500[†]	179	toxicity. A 2.0-fold increase in hardness resulted in a 2.7- to 3.0-fold decrease in toxicity.	with alkalinity and pH.	1998)
Water flea *Daphnia magna*	Short-term (48 h)	LC_{50}	6320[‡]	66–73	Substantial effect of hardness on toxicity. A ~3.1-fold increase in hardness results in a ~13.9- fold decrease in toxicity.	Variation in results is due to 2–3 separate replicate experiments. True hardness is confounded with alkalinity and pH.	(Poston et al. 1984)
		LC_{50}	36 830[‡]	126–140			
		LC_{50}	46 870[‡]	188–205			
Amphipod *Hyalella azteca*	Long-term (water only exposure, 14 days)	LC_{50}	140	61	Substantial effect of hardness on toxicity. A 16-fold increase in hardness results in an 11.5- fold decrease in toxicity.	This analysis includes hardness=15, but this endpoint was dropped, as control survival was low (only in this treatment). True hardness isolated from alkalinity and pH.	(Vizon SciTec Inc. 2004)
		LC_{50}	200	123			
		LC_{50}	340	238			
Green hydra *Hydra viridissima*	Long-term (96 h)	EC_{50}	114	6.6	Effect of hardness unclear. Inconsistent effect of hardness when comparing minimum-detectable-effect concentration (MDEC) values. Minor effect of hardness when EC_{50} values are compared; a 50-fold increase in hardness results in a 1.9-fold decrease in toxicity.	This study was not considered in the CWQG, since its status as a resident species is unclear. True hardness isolated from alkalinity and pH (study also reports the effect of alkalinity on toxicity).	(Riethmuller et al. 2001)
		EC_{50}	177	165			
		EC_{50}	219	330			
Plants, including algae (all long-term)							
Freshwater algae *Chlorella* sp.	Long-term (72 h)	EC_{50}	56	8	Minor effect of hardness on toxicity. A 50-fold increase in hardness results in a 4.8-fold decrease in toxicity.	True hardness isolated from alkalinity and pH.	(Charles et al. 2002)
		EC_{50}	72	40			
		EC_{50}	150	100			
		EC_{50}	270	400			
Freshwater algae *Pseudokirchneriella subcapitata*	Long-term (72 h)	IC_{25}	27	5	Minor effect of hardness when comparing IC_{25} values; a 48-fold increase in hardness results in a 7-fold decrease in toxicity. Minor effect of hardness when IC_{50}	True hardness isolated from alkalinity and pH.	(Vizon SciTec Inc. 2004)
		IC_{25}	94	15			
		IC_{25}	60	64			
		IC_{25}	100	122			

Taxa/organism	Short-term or long-term	Toxicity Endpoint	Effective concentration (µg/L U)	Hardness (as mg/L $CaCO_3$)	Effect of hardness on toxicity[1]	Comments	Reference
		IC_{25}	150	228	values are compared; a 48-fold increase in hardness results in a 1.4-fold decrease in toxicity.		
Macrophyte *Lemna minor*	Long-term (7 days)	IC_{25} dry weight	4700	35	Substantial effect of hardness on toxicity. For both IC_{25} and IC_{50}, a 4-fold increase in hardness results in a 1.6-fold decrease in toxicity.	True hardness isolated from alkalinity and pH. Precipitation noted at the highest test concentration, indicating uranium levels not constant, and lower toxicity estimates could be possible.	(Vizon SciTec Inc. 2004)
		IC_{25} dry weight	12 300	137			
		IC_{25} frond number	6400	35			
		IC_{25} frond number	13 300	137			

All studies that have a * in column one of Table 11 are summarized here.

† Geomean of two clones.

‡ Geomean of separate tests conducted under identical conditions in the same study.

[1] For the purposes of a simple trend analysis, results were compared on a mg/L basis; however, a molar comparison would be more appropriate, since hardness is believed to ameliorate toxicity through competition at the site of uptake. The qualitative terms of "no apparent effect," "minor effect" and "substantial effect" are subjectively assigned, but consistent among studies. "No apparent effect" was assigned if there was no consistent decrease in toxicity with increasing hardness. "Substantial effect" was assigned if the ratio of decrease in toxicity to increase in hardness was greater than or equal to 0.21. For example, in the first case (fathead minnow), this ratio is 4.8/20 = 0.24; hence, this would be classified as a substantial effect. The 0.21 cut-off is derived from the subjective estimate of the reasonable extremes of water hardness values (5 mg/L to 240 mg/L as $CaCO_3$, or 48-fold), and an arbitrary decrease in toxicity (10-fold decrease, a common safety factor used). Hence, 10-fold/48-fold = 0.21. "Minor effect" was assigned if the ratio was less than 0.21.

Table 11. Summary of freshwater short-term and long-term toxicity studies

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^oC)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Fish – Short-term studies all ranks (primary, secondary, unacceptable)																		
Reticulated perchlet	*Ambassis macleayi*	LC_{50} LC_1	800 73	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	96 h SR 2 reps	Juvenile	Yes; No control mortalities	27	-- Contin-uous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported	Unacceptable Non-resident	(Bywater et al. 1991)
Zebra fish	*Brachydanio rerio*	LC_{50}	3050	Uranyl acetate $UO_2(OCOCH_3)_2.2H_2O$	No	--	--	96 h S 1 rep	--	Yes; but control mortality not reported.	Mea-sured, but not stated	63% (not oxygen-ated)	Level of $CaCO_3$: 178 "mg/mL"?	--	7.86	Also noted bioaccumulation and depuration of U BCFs of U = 8.87 x 10^{-3} Depuration after 28 days exposure and 32 days depuration: about 52% of levels of U at day 28	Unacceptable non-resident Not suitable surrogate	(Labrot et al. 1999)
Flannelmouth sucker	*Catostomus latipinnis*	LC_{50}	At all 4 time points, **43 500**	Uranyl nitrate $UO_2(NO_3)_2$	No	--	--	24, 48, 72, 96 h S --	12–13 days old (larvae)	Yes; No control mortality	25.0	72%	144	103	7.93	Toxicity didn't increase with increasing time periods. Also looked at toxicity of inorganic mixtures (i.e., in combination with other metals in environmental samples) pH at the end of the test ranged from 6.7 to 8.9. Static tests were conducted in accordance with ASTM standards.	Secondary Acceptable non-resident	(Hamilton and Buhl 1997)
Mariana's hardyhead	*Cratero-cephalus marianae*	LC_{50} LC_1	1220 260	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	96 h SR 2 reps	Juvenile	Yes; 5% mortality	27	-- Contin-uous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported	Unacceptable Non-resident	(Bywater et al. 1991)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Eastern mosquitofish	*Gambusia holbrooki* Girard 1859	TTD at 4 mg/L U (nominal)	For U-naïve fish, LC_{50} ~ 114 h[4] with exposure of 2570 µg/L and @ 96 h ~ LC_{25-30}[4]	$UO_2(NO_3)_2$	No and Me	--	ICP DL --	7 days SR 2 reps	Not specified	Yes; but control conditions were different than test conditions Control mortality = 7%	19.0–19.7	9.0 or 9.1, and n/a in controls	5.8 and 6.1 in test batches, 8.3 and 9.5 in controls (calc from the Ca and Mg conc)	3.1–3.8 in test batches, 6.0 or 6.6 in controls	6.87–6.92 in test batch-es, 7.21 or 7.24 in con-trols	Tested tolerance (genetically based) of previous U-exposed population with that of a U-naïve population. Allele frequency and other genetic testing included. Control tanks and testing tanks were different sizes with different numbers of fish; control 5 L with 14 fish, test 25 L and 60 fish. After 7 days, 97% (average) of U-naïve fish had died, and 41% (average) of U-tolerant fish had died.	Unacceptable Species is acceptable non-resident, but non-standard endpoint and poor water chemistry of controls	(Keklak et al. 1994)
Bonytail	*Gila elegans*	LC_{50} for swim-up fry, small juvenile, large juvenile	**46 000** for all life stages	Uranyl nitrate $UO_2(NO_3)_2$	No	--	--	96 h S --	Swim-up fry, small juvenile, large juvenile	Yes; No control mortality	25	40%	196	107	7.8	No partial mortalities (i.e., test concentrations produced 0% or 100% mortality). Test concentrations were 13 000, 21 600, 36 000, 60 000, 100 000, 170 000, 280 000 and 470 000 µg/L	Secondary Acceptable non-resident	(Hamilton 1995)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Bluegill	*Lepomis macrochirus*	LC_{50}	**1670**	$UO_2(NO_3)_2 \cdot 6H_2O$	Me	--	Fluorometric --	96 h S 2 reps	Mean weight: 2.7 g Mean length: 5.61 cm	Yes; Control mortality was 0%	19	5.9–8.7	2.5–3.2	< 0.1–3.8	5.10–5.6 (controls 5.85–6.33)	Diluent water was field collected from Upper Three Runs Creek. For fish selection, followed recommendations of ASTM (1980). QA analysis conducted based on American Public Heath Association guidelines (1985). Water chemistry measured at beginning and end of test. Reference toxicant tests performed and compared with US EPA (1980) standards. Dissolved U measured, which was slightly less (~ 11%) than total U. LC_{50} dissolved = 1460 Steep dose-response curve: 0% mortality @ 1250 µg/L, 40% @ 1880, 100% @ 2500	Secondary Only 2 reps	(Trapp 1986)
Black-banded rainbowfish	*Melanotaenia nigrans*	LC_{50} LC_1	1700 370	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	96 h SR 2 reps	7 days	Yes; No control mortalities	27	-- Continuous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported	Unacceptable Non-resident	(Bywater et al. 1991)
Black-banded rainbowfish	*Melanotaenia nigrans*	LC_{50} LC_1	1900 920	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	96 h SR 2 reps	90 days	Yes; No control mortalities	27	-- Continuous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported	Unacceptable Non-resident	(Bywater et al. 1991)
Chequered rainbowfish	*Melanotaenia splendida inornata*	LC_{50} LC_1	2660 880	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	96 h SR 2 reps	7 days	Yes; No control mortalities	27	-- Continuous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported	Unacceptable Non-resident	(Bywater et al. 1991)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^{o}C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Chequered rainbowfish	*Melan-otaenia splendida inornata*	LC_{50} LC_1	3460 260	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	96 h SR 2 reps	90 days	Yes; No control mortalities	27	-- Contin-uous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported	Unacceptable Non-resident	(Bywater et al. 1991)
Chequered rainbowfish	*Melano-taenia splendida inornata*	LC_{50} LC_1	LC_{50}: 1390 LC_1:320	"Uranium sulphate" $U(SO_4)_2.4H_2)$	Me	--	--	96 h F 2 reps	14 days	Yes; 30% control mortality	30	-- Contin-uous aeration	3.97	3.2	6.56	Locally (Australia) collected water used as a control and diluent. Several other water chemical parameters were measured. DOC was also measured: 5.8 mg/L	Unacceptable Non-resident Poor control survival	(Holdway 1992)
Northern purple-spotted gudgeon	*Mogurnda mogurnda*	LC_{50} LC_1	1110 158	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	96 h SR 2 reps	7 days	Yes; No control mortalities	27	-- Contin-uous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported	Unacceptable Non-resident	(Bywater et al. 1991)
Northern purple-spotted gudgeon	*Mogurnda mogurnda*	LC_{50} LC_1	1460 230	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	96 h SR 2 reps	90 days	Yes; No control mortalities	27	-- Contin-uous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported	Unacceptable Non-resident	(Bywater et al. 1991)
Purple-spotted gudgeon fish	*Mogurnda mogurnda*	LC_{50} LC_1 (see note for LC_1)	For 6-day-old, LC_{50}: 1570 LC_1: 700 For 40-day-old, LC_{50}: 3290 For 70-day-old, LC_{50}: 3290	"Uranium sulphate" $U(SO_4)_2.4H_2)$	Me	--	--	96 h F 2 reps	6, 40 and 70 days old	Yes; 15% in 6-day-old, 0% in 40- to 70-day -old.	30	-- Contin-uous aeration	3.97	3.2	6.56	Locally (Australia) collected water used as a control and diluent. Several other water chemical parameters were measured. DOC was also measured: 5.8 mg/L High control mortality in 6-day-old. For studies with acceptable control mortality, LC_1 could not be calculated because the binomial method was used.	Unacceptable Non-resident	(Holdway 1992)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^{o}C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Gudgeon fish	*Mogurnda mogurnda*	BEC_{10} MDEC LC_{50}	BEC_{10}: 1270 MDEC: 1298 LC_{50}: 1570 or 1360?	--	--	--	--	96 h -- --	--		--	--	4	"Low buffering"	6.0		Unacceptable Non-resident	(Markich and Camilleri 1997) and as cited in (Charles et al. 2002; Franklin et al. 2000)
Rainbow trout	*Oncor-hynchus mykiss*	LC_{50}	**6200**	Not stated	No, Me?	--	-- --	96 h F 2 reps	100–140 mm in length average: 130 mm and 27.8 g weight	Yes; Implied that there was 0% mortality	14.2 (ave) range: 14.0–14.5	7.6 (ave) range: 6.6–8.0	30.8 (average) range: 30–32	26.0 (average) range: 26–26	Ave not repor-ted Range : 6.8–7.0	Authors name rainbow trout *Salmo gairdneri*. Authors state that "uranium analyses" was conducted, but don't state measured concentration of U. Unclear whether toxicity values are reported based on nominal or measured concentration. Nominal concentration: 0, 620, 1250, 2500, 5000, 10 000 µg/L No partial mortalities. 0% @ 5000 µg/L, 100% @ 10 000 µg/L.	Secondary Unclear whether toxicity results are based on measured or nominal concentration. Fry are large compared with EC guidelines (recommend ww 0.3–5 g)	(Davies 1980)
*Rainbow trout	*Oncor-hynchus mykiss*	NOEC, LOEC, LC_{50}	**LC_{50} 4200 3900 4000 3800**	$UO_2(NO_3)_2$.$6H_2O$	Me At beginning and end of test from ~ half test conc	--	ICP-MS --	96 h S 1 rep	Fry, mean weight of 0.58 g	Yes; Control mortality 0%	14.9–15.9	8.8–9.7 aerated	20 68 126 243	11–12	6.2–7.0	No partial mortalities (all-or-none response); all survived @ U=2700 (nominal), all died @ U=6700 (nominal). Performed according to Environment Canada biological test methods	Primary Although no reps.	(Vizon SciTec Inc. 2004)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
*Fathead minnow	*Pimephales promelas*	LC_{50}	2800 @ low hardness, 135 000 @ high hardness	$UO_2SO_4.3H_20$	No	--	--	96 h -- --	Not reported	Not reported	--	--	20 (units?) 400 (units?)	18 (units?) 360 (units?)	7.4 8.2	Fathead minnow were also exposed to uranyl nitrate ($UO_2(NO_3)_2.6H_2O$) and uranyl acetate ($UO_2(C_2H_2O_2)_2.2H_2O$) in soft water. These other formulations yielded similar LC_{50} values. $UO_2(NO_3)_2.6H_2O$ LC_{50}: 3100 µg/L $UO_2(C_2H_2O_2)_2.2H_2O$ LC_{50}: 3700 µg/L Authors actually use TL_m, (median tolerance limit) not LC_{50}, but operationally seem identical No units given for hardness and alkalinity.	Unacceptable Poor water chemistry characterization, no reports of control mortality, no characterization of life stage	(Tarzwell and Henderson 1960)
*Fathead minnow	*Pimephales promelas*	LC_{50}, LC_{25} LOEC and NOEC	**LC_{50}** **2000** **2000** **2100** **1800**	$UO_2(NO_3)_2.6H_2O$	Me Before and after renewal for ~ half test conc.	--	ICP-MS --	96 h SR 4 reps	Embryo < 24 hours old	Yes; Control mortality 0–15% after 7 days	24.0–25.8	6.8–8.6 no aeration	23 72 131 244	10–14	6.3–7.0	Performed according to Environment Canada biological test methods Note there is no apparent effect of hardness on toxicity	Primary	(Vizon SciTec Inc. 2004)
Delicate blue-eyes	*Pseudo-mugil tenellus*	LC_{50} LC_1	730 71	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	96 h SR 2 reps	Juvenile	Yes; No control mortal-ities	27	-- Contin-uous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported	Unacceptable Non-resident	(Bywater et al. 1991)
Colorado squawfish	*Ptycho-cheilus lucius*	LC_{50} for swimup fry, small juvenile, large juvenile	**46 000** for all life stages	Uranyl nitrate $UO_2(NO_3)_2$	No	--	--	96 h S --	Swim up fry, small juvenile, large juvenile	Yes; No control mortality	25	40%	196	107	7.8	No partial mortalities (i.e, test concentrations produced 0% or 100% mortality). Test concentrations were 13 000, 21 600, 36 000, 60 000, 100 000, 170 000, 280 000 and 470 000 µg/L	Secondary Acceptable non-resident	(Hamilton 1995)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Brook trout	*Salvelinus fontinalis*	LC_{50}	**8000**	Not specified	No, Me?	--	-- --	96 h F 2 reps	60–100 mm in length; average: 80.7 mm and 7.8 g weight	Yes; Implied that there was 0% mortality	14.2 (ave) range: 14.0–14.5	7.6 (ave) range: 6.6–8.0	30.8 (average) range: 30–32	26.0 (average) range: 26–26	Ave not reported Range 6.8-7.0	Authors state that "uranium analyses" was conducted, but don't state measured concentrations of U. Unclear whether toxicity values are reported based on nominal or measured concentrations. Fish stock collected from the field. LC_{50} @ 120 h = 7200 µg/L Nominal conc: 0, 620, 1250, 2500, 5000, 10 000 µg/L 0% mortality @ 5000 µg/L, 87% mortality @ 10 000 µg/L for 120 h	Secondary Unclear whether toxicity results are based on measured or nominal concentration. Cf EC guidelines for rainbow trout, the fry are slightly larger than test protocol recommends.	(Davies 1980)
Brook trout	*Salvelinus fontinalis*	LC_{50}	59 000	$UO_2SO_4.3H_2O$	Me Through-out test	--	Fluorometry DL --	48 h F 4 reps	Juvenile	Yes; 0% mortality inferred from graph	16	7.4	184	146	7.4	Measured U did not differ "substantially" from nominal U	Secondary 48-h endpoint is non-traditional	(Parkhurst et al. 1984)
*Brook trout	*Salvelinus fontinalis*	LC_{50}	**LC_{50}: 5500** in soft water and 23 000 in hard water	$UO_2SO_4.3H_2O$	Me Through-out test	--	Fluorometry DL --	96h SR 2 reps	Juvenile	Yes; No control mortality	13 and 14	7 and 7	32 and 210	12 and 54	6.7 and 7.5	Measured U did not differ "substantially" from nominal U. LC_{50} calculated by graphical interpolation. No partial mortalities	Secondary Graphical interpolation method is non-traditional	(Parkhurst et al. 1984)
Razorback sucker	*Xyrauchen texanus*	LC_{50} for swim*up fry, small juvenile, large juvenile	**LC_{50}: 46 000** for all life stages	Uranyl nitrate $UO_2(NO_3)_2$	No	--	--	96 h S --	Swim up fry, small juvenile, large juvenile	Yes; No control mortality	25	40%	196	107	7.8	No partial mortalities (i.e., test concentrations produced 0% or 100% mortality). Test concentrations were 13 000, 21 600, 36 000, 60 000, 100 000, 170 000, 280 000 and 470 000 µg/L	Secondary Acceptable non-resident	(Hamilton 1995)
Fish – Long-term studies all ranks (primary, secondary, unacceptable)																		

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
White sucker	*Catostomus commersoni*	NOEC and LOEC for growth (length and dry weight)	NOEC: 7330 LOEC: 27 860 **MATC[4]: 14 300**	$UO_2(NO_3)_2 \cdot 6H_2O$	Me		ICP-MS, DL not specified	30 days SR 4	Fry	Yes; 0% mortality	14	8.9 mg/L	72	68	7.9	Were able to obtain NOEC and LOEC values, but mortality even at 27 860 µg/L was not significantly different from controls.	Primary	(Liber et al. 2004b)
Northern pike	*Esox lucius*	NOEC, LOEC, MATC	NOEC: 1510 LOEC: 4320 **MATC[4]: 2550**	$UO_2(NO_3)_2 \cdot 6H_2O$	Me	--	ICP-MS	65 days SR 4 reps	Embryo	Yes, 12–33% control mortality	8.1	10.8	63	60	7.9		Primary	Liber et al. 2005
Chequered rainbowfish	*Melano-taenia splendida inornata*	LC_{50} LC_1	LC_{50}: 1570 LC_1: 420	"Uranium sulphate" $U(SO_4)_2.4H_2)$	Me	--	--	7 days F 2 reps	31 days	Yes; 0% control mortality	30.0	-- Contin-uous aeration	4.07	1.8	6.3	Measured concentrations were not significantly different between replicates. Locally (Australia) collected water used as a control and diluent. Several other water chemical parameters were measured. DOC was also measured: 1.5 mg/L.	Unacceptable Non-resident	(Holdway 1992)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^{o}C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Purple-spotted gudgeon fish	*Mogurnda mogurnda*	LC_{50} LC_1 NOEC LOEC (see note)	LC_{50} > 1790 LC_1: 750 NOEC: 880 LOEC: 1790 MATC[4]: 1255	Uranium sulphate $U(SO_4)_2.4H_2)$	Me	--	--	14 days F 2 days	1 day old	Yes; 3% mortality	27.1	-- Contin-uously aerated	3.12	2.99	6.43	Measured concentrations were not significantly different between replicates. Locally (Australia) collected water used as a control and diluent. Several other water chemical parameters were measured. DOC = 5.07 mg/L. Larvae also placed under a 15-day post-exposure treatment to determine delayed mortality. NOECs and LOECs were based on weight and mortality; both endpoints yielded the same NOEC/LOEC value.	Unacceptable non-resident	(Holdway 1992)
Purple-spotted gudgeon fish	*Mogurnda mogurnda*	LC_{50} LC_1	LC_{50}: 1590 LC_1: 1270	Uranium sulphate $U(SO_4)_2.4H_2)$	Me	--	--	7 days F 2 reps	1 day old	Yes; 10% control mortality	30.0	-- Contin-uously aerated	4.07	1.8	6.3	Measured concentrations were not significantly different between replicates. Locally (Australia) collected water used as a control and diluent. Several other water chemical parameters were measured. DOC = 1.5 mg/L. Larvae also placed under a 7-day post-exposure treatment to determine delayed mortality. Very steep dose-response.	Unacceptable non-resident	(Holdway 1992)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^{o}C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Purple-spotted gudgeon fish	*Mogurnda mogurnda*	LC_{50} LC_1	For 40-day-old, LC_{50}: 2690 LC_1: 910 For 70-day-old, LC_{50}: 3290 LC_1 (see note)	Uranium sulphate $U(SO_4)_2.4H_2)$	Me	--	--	7 days F 2 reps	40 and 70 days old	Yes; 0% control mortality for both 40- and 70-day-old fish	30	-- Contin-uous aeration	3.97	3.2	6.56	This experiment was run under the same conditions as a reported 96-h short-term study. Measured concentrations were not significantly different between replicates. Locally (Australia) collected water used as a control and diluent. Several other water chemical parameters were measured. DOC was also measured: 5.8 mg/L. Fish also placed under a 7-day post-exposure treatment to determine delayed mortality. LC_1 could not be calculated because the binomial method was used.	Unacceptable Non-resident	(Holdway 1992)
*Rainbow trout	*Oncor-hynchus mykiss*	Percent non-viable embryos NOEC, LOEC, EC_{50}, EC_{25}	LOEC: 280 at hardness of 6 610 at hardness of 61 **EC_{10}[4]:** **260** **480**	$UO_2(NO_3)_2$.$6H_2O$	Me Before and after renewal for ~ half test conc.	--	ICP-MS --	30 or 31 days (embryo-alevin test) SR 4 reps	Embryo (30 minutes of fertil-ization)	Yes; Control non-viable 9% and 10%	13.3–15.2	9.5–10.4	6 61	6–7	6.3–7.2	Performed according to Environment Canada biological test methods. Control non-viability with recommendations, as EC method states test is non-valid if > 35% of controls non-viable at end of embryo-alevin test. Reference toxicant used. Some evidence of ameliorating effect of hardness.	Primary	(Vizon SciTec Inc. 2004)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
*Fathead minnow	*Pimephales promelas*	NOEC, LOEC, LC_{50}, LC_{25}, IC_{25} for growth	LOEC 1300–2000 at diff. hardness for survival MATC[4]: 990–1500 **LC_{10}[4]: 1200 1300 760 980**	$UO_2(NO_3)_2 \cdot 6H_2O$	Me At beginning and end of test from ~ half test conc.	--	ICP-MS --	7 days SR 4 reps	Embryo < 24 hours old	Yes; Control mortality 0–15% after 7 days	24.0–25.8	6.8–8.6 no aeration	23 72 131 244 (meas)	10–14	6.3–7.0	Same test as 96-h short-term study listed in short-term section. Performed according to Environment Canada biological test methods. Note there is no apparent effect of hardness on toxicity	Primary	(Vizon SciTec Inc. 2004)
Lake trout	*Salvelinus namaycush*	NOEC and LOEC for a variety of endpoints	NOEC: 6050 LOEC: 29 780 for survival **MATC[4]: 13 400**	$UO_2(NO_3)_2 .6H_20$	Me and Mo	Dominant species were $UO_2(CO_3)_2^{2-}$ and $UO_2(CO_3)_3^{4-}$ MINTEQA2	ICP MS --	141 days SR 4 reps for embryo-alevin 3 reps for fry	Embryo-alevin-fry	Yes; Control mortality 20.5% after 30-day fry test (not sure for 141 days)	7.6–8.6	10.1–11.0	74–80	69–77	7.9–8.1 with one 7.3	Following EC method, with some exceptions. Rather wide jumps between concentration steps; no partial mortalities. Test endpoints included survival, hatching and swim-up success, alevin and fry growth, general and feeding behaviour, some biochemical endpoints. Endpoints sensitive to U include hatching success, mean time to hatch and survival of alevin/fry. No sublethal effects occurred.	Secondary	(Liber et al. 2004a)
Brook trout	*Salvelinus fontinalis*	NOEC, LOEC for hatch, survival and growth	NOEC: > 9080	$UO_2SO_4.3 H_2O$	Me		Fluorometry (DL not stated)	60 days F	Embryo larval	Yes; inferred from graph 100% hatch, ~50% survival	13.5	7.8	201	189	8.0	No significant adverse effects noted at the highest concentration; no LOEC or NOEC values calculated. After test, 160 nominal was measured as 230 and 10 000 nominal was measured as 9080	Unacceptable Control mortality too high	(Parkhurst et al. 1984)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^{o}C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Invertebrates – Short-term studies all ranks (primary, secondary, unacceptable)																		
Water flea	*Cerio-daphnia dubia*	LC_{50}	10 500	Depleted uranium desorbed from soil (see comments)	Me	--	ICP-MS (DL not given)	96 h SR 10 reps (one animal per rep)	--	Yes; Control mortality 0% on average	25	6.49	176	126	8.36	Contaminated soil (22 500 µg/g U) from a gov't firing range was washed with local well-water. Soil wash water also contained Ag, Be, Cd, Cu, Cr, Hg, Ni, Pb and Zn in the µg/L range. Followed US EPA methods from 1993 and 1994 Control mortality read from graph; 0% with +- 20% SE.	Unacceptable Depleted uranium tested	(Kuhne et al. 2002)
Water flea	*Ceriodaphnia dubia*	LC_{50}	**60 and 89** for $UO_2(NO_3)_2$	$UO_2(NO_3)_2$, HUO_2PO_4 and UO_2	Me	--	ICP-AES Instrument detection limit: 30.6 < µg/L	48 h SR 2 reps	Neonates (2–24 h) or < 24 h	Yes; Control mortality 0% or 5%	25.8–26.0	7.7–8.0 (not aerated, but meas. before and after renewal)	6.1 (only measured in controls)	1.1 (only measured in controls)	6.87–7.76	Toxicity results for HUO_2PO_4 and UO_2 also available. Dilution water was field collected. Water chemistry may differ slightly across 3 experiments. Endpoints are the result of 3 separate tests that yielded reasonably consistent results. More data available based on nominal, and QA/QC shows good agreement between nominal and measured. Reference EPA method 1985 (US EPA 1985).	Primary Not all water chemistry was reported, but endpoints are the result of 3 separate tests that yielded reasonably consistent results. Nominal Concentra-tions used as secondary	(Pickett et al. 1993)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^{o}C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Bivalve clam	*Corbicula fluminea*	Effect was time to first valve closure EC_{50} and various other EC_{xx} values (based on nominal concentrations)	12 @ pH 5.5; 31 @ pH 6.5	$UO_2(NO_3)_2 \cdot 6H_2O$	No, Me and Mo	J-Chess; UO_2OH^+ and UO_2^{2+} @ pH 5.5 $(UO_2)_2CO_3(OH)_3^-$ @ pH 6.5 See graph for more details	ICP-OES (inductively coupled plasma with optical emission spectroscopy); DL = 10 nmol/L	5h S 3 true reps with 5 subreps (15 individuals)	Based on size Mean antero-posterior shell length of 27.50 mm	Yes; Control response based on valve movement was studied for 24 h prior to exposure	20	Not stated; contin-uously aerated	203 (calculated from mmol conc reported)	$[HCO_3^-]$ modelled as 1 x 10^{-3} mmol or 18 x 10^{-3} mmol	5.5 or 6.5	Ecological relevance for valve movement? Duration of closure not reported. Authors here state that the endpoint can be used as an "early warning system of U presence". Markich et al. (2000) using same endpoint state that valve movement is "an integrated measure that can be used to indicate physiological rate functions (e.g., feeding rate)." Difference between measured values at beginning of exposure and nominal was less than 10%. Effects reported based on nominal concentrations.	Unacceptable Endpoint not ecologically relevant, and no character-ization of length of valve closure.	(Fournier et al. 2004)
Bivalve clam	*Corbicula fluminea*	LC_{50} (valves closure)	1 872 080	Uranyl acetate $UO_2(OCOCH_3)_2.2H_2O$	No	--	--	96 h S 1 rep	2–2.5 cm in length	Yes; but control mortality not reported.	Meas-ured, but not stated	63% (not oxygen-ated)	"178 mg/mL"	--	7.86	Also noted bioaccumulation and depuration of U. BCFs of U = 55.67 x 10^{-3}. Depuration after 28 days exposure and 32 days depuration: about 56% of levels of U at day 28. Endpoint: LC_{50} where lethality was referred as: "did not close their valves when the mantle margin was mechanically stimulated."	Unacceptable Non-resident is a suitable surrogate, but unacceptable because control mortality not stated	(Labrot et al. 1999)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Water flea	*Dadaya macrops*	LC_{50} LC_1	1100 140	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	24 h S 2 reps	< 6 hours	Yes; No control mortality	27	-- Contin-uous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported	Unacceptable non-resident	(Bywater et al. 1991)
*Water flea	*Daphnia magna*	LC_{50}	5176 and 8254 @ 90.7 (geomean = **6530)**; 15 250 and 2240 @ 179 (geomean = 18 500)	$UO_2SO_4 \cdot 3H_2O$	Me (before and after exposure)	HARPHRQ Includes $Ca_2(UO_2)(CO_3)_3$	ICP-MS	48 h S (presumed) 3 reps	Neonates	Yes; Control mortality not reported, but less than 10%.	20	App-roached near-satura-tion	90.7 179	62.1 and 126	7.73 and 8.07	Toxicity values reported as UO_2, so for this table, the endpoints have been converted to U. Specific control mortality not reported, but the authors state that they did not report results where there was excessive (> 10%) control mortality LC_{50} at 24 h, 72 h and 96 h also reported; 96 h results reported under "long-term" in this table. Tox test ref method OECD 1981. Data shown are the range for clones; includes data for four clones of *D. magna.* Followed OECD methods (1981).	Primary Specific control mortality not reported, but authors state they used QA/QC criteria.	(Barata et al. 1998)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
*Water flea	*Daphnia magna*	LC_{50}	geomeans = **6320,** 36 830 46 870 depending on hardness	$UO_2(NO_3)_2 \cdot 6H_2O$	Me	--	Colorimetric (no DL stated)	48 h S 4 reps	First instar	Yes; No control mortality-.	20	--	66 130 200	Varied from 54.2 to 133	7.9-8.0	Followed US EPA (1978) *Methods for measuring short-term toxicity of effluents to aquatic organisms*. Results for 2–3 reps of short-term studies reported. The geomeans calculated from 2 or 3 different identical tests at each hardness: 5340, 6190, 7620 (geomean = 6320) @ 66; 44 570, 30 440 (geomean = 36 830) @ 130; 74 340, 2956 (geomean=46 870 @ 200). For SSD, raw values used.	Primary	(Poston et al. 1984)
Water flea	*Daphnia pulex*	LC_{50}	**220**	$UO_2(NO_3)_2 \cdot 6H_2O$	Me	--	Fluorometric DL --	48 h S 4 reps	Neonates (= 24 h)	Yes; No control mortality.	20–21	8.3–9.7	2.3–3.3	< 0.1–0.6 (one control sample was 4.0)	5.10–5.64	95% confidence intervals on 22 (170–360). Diluent water was field collected from Upper Three Runs Creek. This water met ASTM criteria for survival rates. QA analysis conducted based on American Public Heath Association guidelines (APHA et al. 1985). Water chemistry measured at beginning and end of test. Reference toxicant tests performed and compared with US EPA (1980) standards. Dissolved U measured, which was slightly less (~ 13%) than total U.	Primary	(Trapp 1986)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^{o}C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Water flea	*Diaphan-osoma excisum*	LC_{50} LC_1	1000 900	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	24 h S 2 reps	< 6 hours	Yes; No control mortality	27	-- Contin-uous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported	Unacceptable non-resident	(Bywater et al. 1991)
Green hydra	*Hydra viridissima*	NOEC for growth	NOEC > 3900, but when environ-mental sample was diluted, strong growth inhibition was noted.	U in environ-mental sample, 3900 ppb	Me	--	ICP-MS and Scintrex Time Delay Fluorimetry	4 days SR 3 reps	Mature and asexually reprodu-cing	Yes; Popula-tion growth given	30	--	--	--	8.6 or 8.0	Toxicity of environmental sample increased when diluted; authors suggest this is due to pH change associated with dilution. At higher pH, authors hypothesize increase in the carbonate complex $UO_2(CO_3)_3^{4-}$. pH adjusted with sodium carbonate or sodium bicarbonate. Data available for days 1–4. Reference to U inhibiting ATPase, and detoxification via formation of uranium phosphate microgranule. Dilution water was Magela Creek, and dilution water also run as control. Conductivity and pH measured, but not reported.	Unacceptable Classic dose-response not observed. Poor water chemical characteri-zation	(Hyne et al. 1992)
Green hydra	*Hydra viridissima*	Effect is growth inhibition BEC_{10} MDEC EC_{50}	BEC_{10}: 56 MDEC: 61 as UO_2	--	--	--	--	96 h -- --	--	--	--	--	4	"Low buffering"	6.0	Hydra in *Cnidaria* taxa	Unacceptable, very similar study also evaluated; expect data were recycled.	(Markich and Camilleri 1997) and as cited in (Charles et al. 2002; Franklin et al. 2000)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^{o}C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
*Green hydra	*Hydra viridissima*	MDEC (LOEC equivalent using regression) EC_{50} for population growth (reproduc-tion)	MDEC: 32–90, difficult to relate to hardness and alkalinity EC_{50}: 114–219 with increasing hardness	Not stated	Me and Mo	HARPHRQ	ICP MS	96 h SR 3 reps	Actively budding (asexual-ly reprodu-cing)	Back-ground U levels Control; population growth not reported	27	Petri dish - no aeration	6.6 165 330	4.0 102	6.0	Controls look like background concentrations of U (0.1 µg/L). pH, conductivity and DO all measured before and after each renewal, and there were no significant differences between days or between reps. No effect of alkalinity on toxicity. Hydra in *Cnidaria* taxa	Unacceptable non-resident	(Riethmull er et al. 2001)
Water flea	*Latonopsis fasciculate*	LC_{50} LC_1	410 170	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	24 h S 2 reps	< 6 hours	Yes; No control mortality	27	-- Contin-uous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported.	Unacceptable non-resident	(Bywater et al. 1991)
Water flea	*Moino-daphnia macleayi*	LC_{50} LC_1	1290 490	UO_2SO_4	Me, a bit of Mo	Mo (MINTEQ) Dominant species is $UO_2(CO_3)_2(H_2O)_2^{2-}$	Laser-induced uranium fluorescence	24 h S 2 reps	< 6 hours	Yes; No control mortality	27	-- Contin-uous aeration	4.56	3.26	6.57	Measured concentrations were about 70% of nominal concentrations. LC_{50} at 24 h, 48 h and 72 h also reported.	Unacceptable non-resident	(Bywater et al. 1991)
Water flea	*Moino-daphnia macleayi*	EC_{50} for death/ immobil-ization NOEC LOEC	EC_{50}: 160–390 NOEC: 100–270 LOEC: 180–370	UO_2SO_4	Me	--	ICP-MS DL --	48 h S 2 reps (whole experiment repeated)	< 6 hours	Yes; Mortality < 20%	27	98–109%	--	--	6.63–6.92	Authors compare sensitivity of 3 populations of cladoceran (lab culture, wild from pristine, wild from contaminated site from U mining). There were no significant differences in sensitivity between the three populations of cladoceran.	Unacceptable non-resident	(Semaan et al. 2001)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^{o}C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Freshwater oligochaete	*Tubifex tubifex*	LC_{50}	2050	$UO_2(CH_3COO)_2.2H_2O$	No	--	--	96 h SR 3 reps	--	Not reported	Mean: 30 Range: 29.5–31	Mean: 5.8 Range: 5.2–6.0	Mean: 245 Range: 230-250	Mean: 400 Range: 390-140	Mean: 7.6 Range : 7.5–7.7	24-h and 48-h toxicity endpoints also measured. 95% CI for 2050 mg/L (1720–2260) Followed APHA methods (APHA et al. 1981). Water only exposure (although *T. tubifex* lives in sediment: additional stress).	Unacceptable Control mortality not reported. Nominal concentration. Temperature high for Canadian environment.	(Khangarot 1991)
Bivalve	*Velesunio angasi*	Effect was valve closure BEC_{10} MDEC EC_{50}	BEC_{10}: 81–805 MDEC: 84–845 EC_{50}: 103–1080	$UO_2SO_4 \cdot 3H_2O$ (?)	Me and Mo	A mess of different UO_2 species (HARPHRQ)	ICP-MS (no DL stated)	48 h F	0.1–30 years (based on size)	Yes; Exposure phase followed control phase for each treatment	28	88–95%	3.71	--	5.0, 5.3, 5.5, 5.8, 6.0	With and without fulvic acid, 3150 and 7910 µg/L in some pH treatment groups. Background concentration of U: 0.127 µg/L. Toxicity decreased with fulvic acid and increases in pH (most toxic at pH 5.0). Authors performed stepwise linear regression to show that UO_2^{2+} and UO_2OH^+ explain 97.5% of toxic response.	Unacceptable non-resident	(Markich et al. 2000)
Invertebrates – Long-term studies all ranks (primary, secondary, unacceptable)																		
Water flea	*Cerio-daphnia dubia*	NOEC LOEC based on neonate production	NOEC: 1970 LOEC: 3910	Depleted uranium desorbed from soil (see comments)	Me	--	ICP-MS (DL not given)	7 days SR 20 reps (one animal per rep)	--	Yes; High control reproduc-tion rate	25	6.86	190	148	8.49	Contaminated soil (22 500 µg/g U) from a government firing range was washed with local well water. Soil wash water also contained Ag, Be, Cd, Cu, Cr, Hg, Ni, Pb and Zn in the µg/L range. Decreased growth with increasing DU concentrations was also observed. Followed US EPA methods from 1993 and 1994	Unacceptable Depleted uranium used	(Kuhne et al. 2002)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Water flea	*Cerio-daphnia dubia*	NOEC, LOEC, IC_{50}, IC_{25}, IC_{10}	NOEC: 1540 LOEC: 6400 EC_{50}: 3970 EC_{25}: 2700 **EC_{10} (reprod.): 1900**	$UO_2(NO_3)_2$.$6H_2O$	Me	--	ICP-MS	7 days SR 10 reps	Neonates	Yes; 0%	23.9	7.7	76	74	8.2–8.4	An EC_{10} endpoint was calculated with the slope obtained from the relation between EC_{25} and EC_{50}.	Primary	(Liber et al. 2007)
Water flea	*Cerio-daphnia dubia*	NOEC, LOEC for reproduc-tion	MATC[4]: 2	$UO_2(NO_3)_2$, HUO_2PO_4 and UO_2	Me	--	ICP-AES Instrument detection limit: 30.6 µg/L	7 days SR 1 rep	Neonates (20–24 h) or < 24 h	Yes; Control mortality 0% or 5%	24.5–26	7.0–8.0 (one reading of 6.8) (not aerated)	6.1 (but only measured in controls)	1.1 (but only measured in controls)	6.7–7.5	Toxicity results for HUO_2PO_4 and UO_2 also available. Dilution water was field collected. Water chemistry may differ slightly between experiments. Reference EPA method 1985 (US EPA 1985).	Primary Not all water chemistry was reported, but endpoints are the result of 3 separate tests that yielded reasonably consistent results.	(Pickett et al. 1993)
Water flea	*Cerio-daphnia dubia*	NOEC, LOEC, LC_{50}, LC_{25}, IC_{50}, and IC_{25}, for survival and reproduc-tion	MATC[4] (reprod): 37–100 at diff hardness MATC[4] (survival): 96–270 **IC_{10}[4] (reprod): 33 59 22 25** LC_{10}[4]: 28–140	$UO_2(NO_3)_2$.$6H_2O$	Me Before and after renewal for ~ half test conc.	--	ICP-MS --	3 brood (7 ± 1 day) SR 10 reps (1 animal per rep)	< 24 hours old	Yes; Control mortality not stated	21.4–26.2	7.1–8.4 no aeration	5 17 124 252 (meas)	5–8	6.5–7.3	Followed Environment Canada methodology.	Primary	(Vizon SciTec Inc. 2004)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Midge	*Chironomus tentans*	LOEC, NOEC, LC_{50}, IC_{50} for growth	MATC[4]: 800 IC_{50}: 10 200	$UO_2(NO_3)_2 \cdot 6H_2O$	Me	--	Not stated, but DL of 0.01 µg/L	10 days SR 4 reps	Larvae	Yes, Control mortality was always less than 20%, with average morts of 90.0%, 90.0%, 97.5%, and 92.5% for F0, F1, F2 and F3, respect-ively	23.1	7.18	125	84	7.18	Missing information for determination of data quality was provided by the author in personal communications. Study was presented as a poster presentation. Detection method not stated, but used standard method.	Primary Poster presentation; details have been provided to allow for primary classification.	(Burnett and Liber 2006)
Midge	*Chironomus tentans*	LC_{50} NOEC, LOEC, EC_{50}, EC_{25}, EC_{10}	LC_{50}: 5010 NOEC: 2240 LOEC: 9560 EC_{50}: 4320 EC_{25}: 1930 **EC_{10} (growth): 930**	$UO_2(NO_3)_2 .6H_2O$	Me	--	ICP-MS	28 days SR 10 reps	Larvae	Yes; < 10%	23.1	7.2	80	76	8.0	An EC_{10} endpoint was calculated with the slope obtained from the relation between EC_{25} and EC_{50}.	Primary	Liber et al. 2007
Midge	*Chironomus tentans*	NOEC, LOEC, MATC[4]	NOEC: 39 LOEC: 157 MATC: 78 Growth	$UO_2(NO_3)_2 .6H_2O$	Me	--	ICP-MS	10 days SR 10 reps	Larvae	Yes	23	7.2	134	66	7.8		Primary	(Muscatello et al. 2009)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Water flea	*Daphnia magna*	LC_{50} NOEC, LOEC, EC_{50}, EC_{25}, EC_{10}	LC_{50}:850 NOEC: 450 LOEC: 1810 EC_{50}: 1250 EC_{25}: 830 **EC_{10} (reprod.): 570**	$UO_2(NO_3)_2$.$6H_2O$	Me	--	ICP-MS	21 days SR 10 reps	Neonates	Yes; 10% mortality	22	8.3	75	73	8.0–8.4	An EC_{10} endpoint was calculated with the slope obtained from the relation between EC_{25} and EC_{50}.	Primary	(Liber et al. 2007)
Water flea	*Daphnia magna*	LOEC based on reproduc-tion	LOEC (reprod): 520–2250 MATC[4] (reprod): 1700 LC_{10}[4]: 319–683 **EC_{10}[4] (reprod): 123 373 1160 1360**	$UO_2(NO_3)_2$ ·$6H_2O$	Me	--	Colormetric (no DL stated)	21 days SR 1 rep	First instar	Yes; Control mortality was 0%, and high reproduc-tion rate	20	--	Only measured in dilutant water	Only measured in dilutant water	Only mea-sured in dilu-tant water	Followed ASTM (1981). Proposed standard practice for conducting *Daphnia magna* renewal long-term toxicity test. Second replicate showed unusual results (partial mortality only, apparent stimulation of reproduction above control levels), and hence are not used here. Authors hypothesize that large difference between test 1 and test 2 was due to differences in health of stocks.	Primary	(Poston et al. 1984)
Amphipod	*Hyalella azteca*	LC_{25}, LC_{50}, LC_{10}	LC_{25}: 2100 LC_{50}: 4000 LC_{10}: 1200	$UO_2(NO_3)_2$ ·$6H_2O$	Me	--	ICP MS	7 days SR 1 rep	Adult	Not reported	25	7.1–9.4	120	76	6.9–7.2	Control mortality not reported. Standard toxicity testing methods not reported.	Secondary	(Alves et al. 2009)
Amphipod	*Hyalella azteca*	LC_{25}, LC_{50}, LC_{10}	LC_{25}: 540 LC_{50}: 1100 LC_{10}:300	$UO_2(NO_3)_2$ ·$6H_2O$	Me	--	ICP MS	7 days SR 1 rep	Juvenile	Not reported	25	7.1–9.4	120	76	6.9–7.2	Control mortality not reported. Standard toxicity testing methods not reported.	Secondary	(Alves et al. 2009)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Amphipod	*Hyalella azteca*	LC_{50} in tap water and 10% tap water with DI	Only measured in soft water LC_{50}:21 In tap water, nominal LC_{50}:1651	AA standards, preserved in HNO_3	Me and No	--	ICP MS DL --	7 days S 1 rep	1–11 days	Yes; Only data with ≤20% control mortality were used	24–25	7–10 Aerated before test, but not during	18 or 124	14 or 84	7.37–8.27 Or 8.21–8.46 at the end of the test	Two water types were used: (i) dechlorinized tap water (Lake Ontario) and (ii) 10% tap and 90% de-ionized. More water chemistry: (i) contained Ca @ 35 mg/L, Mg @ 8.7 mg/L, DOC @ 1.1 mg/L (ii) contained Ca @ 5.6 mg/L, Mg @ 0.90 mg/L, DOC @ 0.28 mg/L. LC_{50} was not measured in the hard water, and there was a significant difference between measured and nominal endpoints for soft water. In soft water, nominal LC_{50} = 54 µg/L cf Measured LC_{50} = 21 µg/L	Secondary Statistics used pooled data from different concentrations in different experiments and treated them as a single experiment.	(Borgman et al. 2005)
Amphipod	*Hyalella azteca*	LC_{50} LOEC (see note)	LC_{50}: 1520 LOEC[4] ~ 1000	Depleted uranium desorbed from soil (see comments)	Me	--	ICP MS (DL not given)	14 days SR 5 reps	Known age, but not reported	Yes; Control mortality ~16% (read from graph)	23	5.13	157	137	7.91	LOEC approximated from graph, but not reported in study; no apparent NOEC. Author was contacted for an exact LOEC value. Contaminated soil (22 500 µg/g U) from a government firing range was washed with local well water. Soil wash water also contained Ag, Be, Cd, Cu, Cr, Hg, Ni, Pb and Zn in the µg/L range. No relationship between growth and increasing DU concentrations. Followed US EPA methods from 1993 and 1994.	Unacceptable Depleted uranium used	(Kuhne et al. 2002)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^{o}C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
*Amphipod	*Hyalella azteca*	LC_{50} NOEC, LOEC, EC_{50}, EC_{25}, EC_{10}	LC_{50}: 30 NOEC: 57 LOEC: 156 EC_{50}: 67 EC_{25}: 27 **EC_{10} (growth): 12**	$UO_2(NO_3)_2$.$6H_2O$	Me	--	ICP-MS	28 days SR 10 reps	2–9 days old	Yes	23.0	7.5	73	80	8.2	An EC_{10} endpoint was calculated with the slope obtained from the relation between EC_{25} and EC_{50}.	Primary	(Liber et al. 2007)
*Amphipod	*Hyalella azteca*	NOEC, LOEC, LC_{50}, and LC_{25} for survival and growth	MATC[4] for survival: 90–130 MATC[4] for growth at hardness of 238: 66 LC_{10}[4]: 55–88	$UO_2(NO_3)_2$.$6H_2O$	Me Before and after renewal for ~ half test conc.	--	ICP-MS --	14 days SR 6 reps	8-9 days old	Yes; Control mortality 28% in low hardness, < 10% at other hardness	21.4-23.2	8.3-8.8 Aeration	17 61 123 238 (meas)	8–10	6.4–7.1	Followed Environment Canada methodology. Growth generally not sensitive endpoint. Control survival at hardness of 17 was poor, so that point not used for guideline derivation. Reference toxicants also run.	Primary Dismiss lowest hardness point because of inconsistent results.	(Vizon SciTec Inc. 2004)
Green hydra	*Hydra viridissima* or *Hydra vulgaris*	LOEC for growth	*H. viridissima*: 150 or 200 *H. vulgaris:* 0.400 or 0.550 ppb See Comments	UO_2SO_4.3 H_2O	No and Me	--	ICP-MS and Scintrex Time Delay Fluorometry	5 days SR 3 reps	Mature and asexually reproducing	Yes; Population growth given	30	--	--	--	6.3	Data available for days 1–6. Toxicity increased with pH increases from 7.5 to 9.0. Two different LOECs correspond to two different dilution waters used. Slightly lower LOECs (150 and 400) correspond to water samples collected during the dry season. The other LOECs (200 and 550) correspond to water samples collected during the wet season.	Unacceptable non-resident.	(Hyne et al. 1992)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^{o}C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
Water flea	*Moin-odaphnia macleayi*	NOEC, LOEC Survival, reprod	Survival: NOEC:4–46 LOEC:7–49	UO_2SO_4	Me	--	ICP-MS DL --	5–6 days SR 2–3 reps (whole experiment)	< 6 h	Yes; Control mortality < 20%, high reprod rate	27	97–112	--	--	6.85–7.14	Authors compare sensitivity of 3 populations of cladoceran (lab culture, wild from pristine, wild from contaminated site from U mining). There were no significant differences in sensitivity between the three populations of cladoceran.	Unacceptable Non-resident	(Semaan et al. 2001)
Water flea	*Simoceph-alus serrulatus*	LC_{50} NOEC, LOEC, EC_{50}, EC_{25}, EC_{10}	LC_{50}: 3860 NOEC: 460 LOEC: 1820 EC_{50}: 1900 EC_{25}: 920 **EC_{10} (reprod.): 480**	$UO_2(NO_3)_2$.$6H_2O$	Me	--	ICP-MS	21 days SR 10 reps	Neonates	Yes; 0%	17.2	8.5	78	70	8.0–8.4	An EC_{10} endpoint was calculated with the slope obtained from the relation between EC_{25} and EC_{50}.	Primary	(Liber et al. 2007)
Algae and Plants – Long-term studies all ranks (primary, secondary, unacceptable)																		
	Chlamy-domonas reinhardtii	IC_{50}	68.3 (287 nM) @ pH 5 4000 (17 µM) @ pH 7 (but interpret with caution @ pH 7)	Not stated	No or Me? Mo	Used CHESS 3.04	-- --	48 h	Expo-nential growth rate	--	--	--	--	--	5 and 7	Not many experimental details are available, since results are from a poster presentation. By the author's own admission, the toxicity results from pH 7 should be interpreted with caution, as some solubilities were exceeded. Hence, they have not been included as an endpoint here (only pH 5 results are reported).	Unacceptable	(Fortin et al. 2004; Gilbin et al. 2003)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (^{o}C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
*Freshwater algae	*Chlorella* sp.	Effect is growth inhibition BEC_{10} MDEC EC_{50}	MDEC: 1.6–12	$UO_2SO_4 \cdot 3H_2O$	Me total; Mo species	HARPHRQ Dominant species of U are: $(UO_2)_2(OH)_3CO_3^-$, $UO_2(CO_3)_2^{2-}$ and UO_2CO_3 and relative abundance changes dramatically with total U	ICP-MS (DL not stated)	72 h S 3 reps	4–5 days old	Control growth measured; used cells that were 4–5 days old and have a good growth rate	27	--	8, 40, 100, 400	8 mg	7.0	Population growth similar to Environment Canada standards for *P. subcapitata.* Also measured surface-bound U and intracellular U. Toxicity always decreased with increasing hardness; 5- to 7.5-fold decrease in toxicity with 50-fold increase in hardness. Measured U concentrations were within 20% (typically within 10%) of nominal.	Unacceptable *Chlorella* could be acceptable, but tests were run at temperatures too high to be representative of Canadian waters.	(Charles et al. 2002)
Freshwater algae	*Chlorella* sp.	Effect is growth inhibition BEC_{10} MDEC EC_{50}	MDEC: 34 @ pH 5.7 13 @ pH 6.5	$UO_2SO_4 \cdot 3H_2O$	Me total; Mo species	U species Mo (HARPHRQ)	ICP-AES (DL not stated)	72 h S 3 reps	4–5 days old	Control growth measured; used cells that were 4–5 days old and have a good growth rate	27	--	3.91	--	5.7 and 6.5	Population growth similar to Environment Canada standards for *P. subcapitata.* Up to 40% of U absorbed to the walls of the test flasks throughout the tests. Intra- and extra-cellular U measured in algae cells. Authors postulate competition with H^+ (not speciation) is overriding factor governing toxicity. Dominant species of U include: $(UO_2)_2(OH)_3CO_3^-$, $(UO_2)_3(OH)_7^-$ and $(UO_2)_2(OH)_3CO_3^-$, which increase as U increases and as pH increased from 5.7 to 6.5.	Unacceptable *Chlorella* could be acceptable, but tests were run at temperatures too high to be representative of Canadian waters.	(Franklin et al. 2000)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
	Chlorella sp.	Growth inhibition IC_{50}, NOEC, LOEC	Synthetic water: IC_{50}: 74 NOEC: 38 LOEC: 70 Natural water: IC_{50}: 137 NOEC: 72 LOEC: 120	Not stated (see notes)	Me total; Mo species	HARPHRQ	ICP-MS, ICP-ES (DL not specified)	72 h S 3 reps		Yes; control mortality not stated, but had growth and reproduc-ibility observed in controls	27		Ca^{2+}: 0.52 mg/L Mg^{2+}: 0.64 mg/L	4.0	6.2	U added as per Reithmuller et al. (2003); Ecotoxicological testing protocols for Australian tropical freshwater organisms. Presumed standard. Study examined effects of DOM on U toxicity; indicated that increasing DOM decreases toxicity of U.	Unacceptable *Chlorella* could be acceptable, but tests were run at temperatures too high to be representativ e of Canadian waters.	(Hogan et al. 2005)
Cryptophyte	*Cryptomo-nas erosa*	NOEC, LOEC, IC_{50}, IC_{25}, IC_{10}	NOEC: 1310 LOEC: 1970 IC_{50}: 1260 IC_{25}: 440 **IC_{10} (growth): 172**	$UO_2(NO_3)_2$.$6H_2O$	Me	--	ICP-MS	6 days SR 5 reps	Not stated	Yes	20.8	7.7	101	52	7.1–9.1	An IC_{10} endpoint was calculated with the slope obtained from the relation between IC_{25} and IC_{50}.	Primary	Liber et al. 2007
*Macrophyte	*Lemna minor*	IC_{50} and IC_{25} based on frond number and dry weight	At hardness of 35: IC_{50} (frond no): 7400, and IC_{50} (dry wgt): 13 100 IC_{10}[4] (frond no): 3400, and **IC_{10}[4] (dry wt): 3100**	$UO_2(NO_3)_2$.$6H_2O$	Me Before and after renewal for ~ half test conc.	--	ICP-MS --	7d S 4 reps	Age not stated	Yes; Control response inferred from graph; slight hormetic effect	Not stated	Not stated Not aerated	35 137 (meas)	7–9	5.8–7.4	Followed Environment Canada's methodology. Alkalinity increased by the end of the test. In hardness of 35, pH decreased by the end of the test. Values reported at hardness of 137 not used for guideline derivation, as precipitation was noted, indicating inconsistent concentrations.	Primary	(Vizon SciTec Inc. 2004)

Common name	Scientific name	Endpoint	Effective U conc (µg U/L)[2]	U added as	Analytical chemistry			Duration, test type[1], replicates	Life stage	Controls	Test conditions					Comments	Data quality category	Reference
					Mo, No, Me?[2]	Model name; major species[3]	Detection method and DL				Temp. (°C)	DO (mg/L or %)	Hardness (mg $CaCO_3$/L)	Alkalinity (mg $CaCO_3$ /L)	pH			
*	*Pseudo-kirchneriella subcapitata*	NOEC, LOEC, IC_{50}, IC_{25}, IC_{10}	NOEC: 570 LOEC: 1110 IC_{50}: 730 IC_{25}: 190 **IC_{10} (growth): 57**	$UO_2(NO_3)_2$.$6H_2O$	Me	--	ICP-MS	72 h SR 5 reps	Not stated	Yes	22	6.3–8.4	70	64	7.8–9.7	An IC_{10} endpoint was calculated with the slope obtained from the relation between IC_{25} and IC_{50}.	Primary	Liber et al. 2007
*	*Pseudo-kirchneriella subcapitata* (see note)	NOEC, LOEC, IC_{50}, and IC_{25} based on growth	MATC[4]: 20–310 depending on hardness **IC_{10}[4] (growth): 5.4 55 54 120**	$UO_2(NO_3)_2$.$6H_2O$	Me Before and after renewal for ~ half test conc.	--	ICP-MS --	72 h S 4 reps 8 reps of control	Not stated	Yes; Control perfor-mance not stated	24.3–25.8	8.0–8.6	5 15 64 122 228 (meas)	7–8	Initial pH of 6.8-8.2	U of Sask reports state that this species is properly referred to as *Pseudokirchneriella subcapitata*. Followed Environment Canada's methodology.	Primary	(Vizon SciTec Inc. 2004)

[1] Indicates: static (S), static renewal (SR), flow-through (F), reps (replicates). Endpoint abbreviations: effect concentration (EC), lethal concentration (LC), inhibitory concentration (IC), 10% bound effect concentration (BEC_{10}), minimum-detectable-effect concentration (MDEC), no-observed-effect concentration (NOEC), lowest-observed-effect concentration (LOEC), maximum acceptable toxic concentration (MATC).

[2] Modelled (Mo), nominal (No) or measured (Me) concentrations. Filtered or unfiltered samples in cases where uranium has been measured has not been recorded. Differences between filtered and unfiltered samples do not apply to cases where only nominal concentrations were used. Unless otherwise stated, whether uranium was measured before or after experiments or renewals has not been noted.

[3] Model name: unless otherwise stated, all uranium species are modelled (not measured).

[4] These endpoints were not calculated in the original study. E.g., MATC = geometric mean of the NOEC and LOEC values.

*These studies were used to evaluate the effect of water chemistry parameters on toxicity.

Notes

- Data were categorized according to the “Protocol for the Derivation of Water Quality Guidelines for the Protection of Aquatic Life” (CCME 2007).
- Data points **in bold font and shaded** were used in the short-term or long-term SSD (see Tables 14 and 16, respectively).
- Studies that were deemed “unacceptable” (either as a result of poor data quality or non-resident biota) appear in grey font, and are not considered in guideline derivation.
- If uranium was measured in test solutions (as opposed to nominal), the time point of the measurement was not recorded (i.e., before renewal/start of test or after renewal/completion of test). This detail was not recorded because uranium concentrations are expected to be relatively stable over short time periods (e.g., does not volatilize, not known to adhere or precipitate rapidly). Two studies that report “before and after” measurements verify that uranium concentrations remain relatively constant over the exposure period. During a static renewal study on uranium toxicity to early life stage lake trout (*Salvelinus namaycush*) with renewal every 3 or 7 days, the average difference between the “old” and “new” water was 2%; in addition, 95% confidence intervals for the “before” and “after” measurements overlapped, showing no significant difference (Liber et al. 2004a). In separate tests on a daphnid (*Daphnia pulex*) and bluegill sunfish (*Lepomis macrochirus*) run under static conditions for 48 h and 96 h, uranium concentrations at the end of the test were on average 7% lower than initial concentrations (confidence intervals were not calculated).

Table 12. Summary of existing uranium water quality guidelines in different jurisdictions

Jurisdiction	Uranium guideline value[1]	Comments	Reference
Environmental water quality values (for the protection of aquatic life)			
Canadian Guidelines for Surface Water Quality	300 µg/L	The guideline set for aquatic life and wildlife was determined using an application f actor due to lack of sublethal data. The factor of 0.05 was used, as uranium does not biomagnify.	(Environment Canada 1983)
Quebec Regional Water Quality Objective	14 µg/L in water with hardness of 20–100 mg/L $CaCO_3$ 100 µg/L in water with hardness of 100-210 mg/L $CaCO_3$	These values are CVAC[2], aquatic life chronic value (provisional). CVAA[2], aquatic life short-term values (provisional) are 0.32 mg/L U for hardness of 20-100 mg/L $CaCO_3$ and 2.3 mg/L U for hardness of 100–210 $CaCO_3$.	(Boudreau and Guay 2002)
Saskatchewan Surface Water Quality Objectives for the Protection of Aquatic Life	15 µg/L	This guideline was developed by the Industrial, Uranium and Hardrock Mining Unit of Saskatchewan Environment.	(Saskatchewan Environment 2006)
Ontario Interim Provincial Water Quality Objective (PWQO)	5 µg/L	Report states, “this interim PWQO was set for emergency purposes based on the best information readily available. Employ due caution when applying value.”	(MOEE 1994)
Australia and New Zealand	0.5 µg/L	A freshwater low reliability trigger value of 0.5 µg/L was calculated for uranium using an application (safety) factor of 20 on limited long-term data.	(ANZECC and ARMCANZ 2000)

Jurisdiction	Uranium guideline value[1]	Comments	Reference
United States Environmental Protection Agency – National Recommended Water Quality Criteria 2002	--	Not currently listed	(US EPA 2002)
Drinking water guideline values			
Canadian Drinking Water Quality Guideline	20 µg/L	This guideline is listed as an "interim maximum acceptable concentration." Derived for the protection of human health. May include consideration of radioactive hazard.	(Federal-Provincial-Territorial Committee on Drinking Water 2003)
United States Environmental Protection Agency – National Primary Drinking Water Regulations	30 µg/L	This number is the "maximum contaminant level." Announced in the *US Federal Registrar*, December 7, 2000. Not clear if the regulation is based solely on chemical toxicity of uranium, or on combined radioactive and chemical hazard.	(US EPA 2000)
World Health Organization – Guidelines for Drinking Water Quality (First Addendum to 3rd edition, 2006)	15 µg/L	Provisional guideline value, as there is evidence of a hazard, but the available information on health effects is limited; and because calculated guideline value is below the level that can be achieved through practical treatment methods, source protection, etc. Note that this changed from the 2nd edition (1998), which listed the uranium tolerable daily intake value as 0.002 mg/L.	(WHO 1998, 2006)

[1]Unless otherwise stated, the reported jurisdictions based their guideline values only on the chemical toxicity of uranium (i.e., exclude radioactive hazard.

[2] CVAC: Critère de vie aquatique chronique (chronic aquatic life toxicity criterion). CVAA: Critère de vie aquatique aigu (acute aquatic life toxicity criterion).

Table 13. Minimum data set requirements for the derivation of a short-term exposure guideline for freshwater environments

Group	Guideline		
	Type A	**Type B1**	**Type B2**
Fish	Three species, including at least one salmonid and one non-salmonid.		Two species, including at least one salmonid and one non-salmonid.
Aquatic invertebrates	Three aquatic or semi-aquatic invertebrates, at least one of which must be a planktonic crustacean. For semi-aquatic invertebrates, the life stages tested must be aquatic. It is desirable, but not necessary, that one of the aquatic invertebrate species be a mayfly, caddisfly, or stonefly.		Two aquatic or semi-aquatic invertebrates, at least one of which must be a planktonic crustacean. For semi-aquatic invertebrates, the life stages tested must be aquatic. It is desirable, but not necessary, that one of the aquatic invertebrate species be a mayfly, caddisfly, or stonefly.
Plants	Toxicity data for aquatic plants or algae are highly desirable, but not necessary. However, if a toxicity study indicates that a plant or algal species is among the most sensitive species in the data set, then this substance is considered to be phyto-toxic, and two studies on non-target freshwater plant or algal species are required.		
Amphibians	Toxicity data for amphibians are highly desirable, but not necessary. Data must represent fully aquatic stages.		
Preferred endpoints	Acceptable LC_{50} or equivalent (e.g., EC_{50} for immobility in small invertebrates).		
Data quality requirement	Primary and secondary LC_{50} (or equivalents) data are acceptable to meet the minimum data set requirement. Both primary and secondary data will be plotted. A chosen model should sufficiently and adequately describe data and pass the appropriate goodness-of-fit test.	The minimum data requirement must be met with primary LC_{50} (or equivalents) data. The value used to set the guideline must be primary.	The minimum data requirement must be met with primary LC_{50} (or equivalents) data. Secondary data are acceptable. The value used to set the guideline may be secondary.

Table 14. Minimum data set requirements for the derivation of a long-term exposure guideline for freshwater environments

Group	Guideline		
	Type A	**Type B1**	**Type B2**
Fish	Three species, including at least one salmonid and one non-salmonid.		Two species, including at least one salmonid and one non-salmonid.
Aquatic invertebrates	Three aquatic or semi-aquatic invertebrates, at least one of which must be a planktonic crustacean. For semi-aquatic invertebrates, the life stages tested must be aquatic. It is desirable, but not necessary, that one of the aquatic invertebrate species be a mayfly, caddisfly, or stonefly.		Two aquatic or semi-aquatic invertebrates, at least one of which must be a planktonic crustacean. For semi-aquatic invertebrates, the life stages tested must be aquatic. It is desirable, but not necessary, that one of the aquatic invertebrate species be a mayfly, caddisfly, or stonefly.
Aquatic plants	At least one study on a freshwater vascular plant or freshwater algal species. If a toxicity study indicates that a plant or algal species is among the most sensitive species in the data set, then this substance is considered to be phyto-toxic, and three studies on non-target freshwater plant or algal species are required.		Toxicity data for plants are highly desirable, but not necessary. If a toxicity study indicates that a plant or algal species is among the most sensitive species in the data set, then this substance is considered to be phyto-toxic, and two studies on non-target freshwater plant or algal species are required.
Amphibians	Toxicity data for amphibians are highly desirable, but not necessary. Data must represent fully aquatic stages.		Toxicity data for amphibians are highly desirable, but not necessary. Data must represent fully aquatic stages.
Preferred endpoints	The acceptable endpoints representing the no-effects threshold and EC_{10}/IC_{10} for a species are plotted. The other, less preferred, endpoints may be added sequentially to the data set to fulfill the minimum data requirement condition and improve the result of the	The most preferred acceptable endpoint representing a low-effects threshold for a species is used as the critical study; the next less preferred endpoint will be used sequentially only if the more preferred endpoint for a given species is not available.	

Group	Guideline		
	Type A	**Type B1**	**Type B2**
	modelling for the guideline derivation if the more preferred endpoint for a given species is not available. The preference ranking is done in the following order: Most appropriate EC_x/IC_x representing a no-effects threshold > EC_{10}/IC_{10} > EC_{11-25}/IC_{11-25} > MATC > NOEC > LOEC > EC_{26-49}/IC_{26-49} > nonlethal EC_{50}/IC_{50}. Multiple comparable records for the same endpoint are to be combined by the geometric mean of these records to represent the averaged species effects endpoint.	The preference ranking is done in the following order: Most appropriate EC_x/IC_x representing a low-effects threshold > EC_{15-25}/IC_{15-25} > LOEC > MATC > EC_{26-49}/IC_{26-49} > nonlethal EC_{50}/IC_{50} > LC_{50}.	
Data quality requirement	Primary and secondary no-effects and low-effects level data are acceptable to meet the minimum data set requirement. Both primary and secondary data will be plotted. A chosen model should sufficiently and adequately describe data and pass the appropriate goodness-of-fit test.	The minimum data requirement must be met with primary data. The value used to set the guideline must be primary. Only low-effect data can be used to fulfill the minimum data requirement.	Secondary data are acceptable. The value used to set the guideline may be secondary. Only low-effect data can be used to fulfill the minimum data requirement.

Table 15. Endpoints used in the SSD to determine the short-term CWQG for uranium

Species	Endpoint	Concentration (µg U/L)	Reference
Fish			
Bluegill *Lepomis macrochirus*	96-h LC_{50}	1670	Trapp (1986)
Fathead minnow *Pimephales promelas*	96-h LC_{50}	2000	Vizon Scitech Inc (2004)
		2000	
		2100	
		1800	
		2000*	
Rainbow trout *Oncorhynchus mykiss*	96-h LC_{50}	6200	Davies (1980)
		4200	Vizon Scitech Inc (2004)
		3900	Vizon Scitech Inc (2004)
		4000	Vizon Scitech Inc (2004)
		3800	Vizon Scitech Inc (2004)
		4000*	
Brook trout *Salvelinus fontinalis*	96-h LC_{50}	8000	Davies (1980)
		5500	Parkhurst et al. (1984)
		6600*	
Flannelmouth sucker *Catostomus latipinnis*	24-h LC_{50}	43 500	Hamilton and Buhl (1997)
Bonytail *Gila elegans*	96-h LC_{50}	46 000	Hamilton (1995)
Colorado squaw *Ptychocheilus lucius*	96-h LC_{50}	46 000	Hamilton (1995)
Razorback sucker *Xyrauchen texanus*	96-h LC_{50}	46 000	Hamilton (1995)
Invertebrates			
Water flea *Ceriodaphnia dubia*	48-h LC_{50}	60	Pickett et al. (1993)
		89	Pickett et al. (1993)
		72*	
Water flea *Daphnia pulex*	48-h LC_{50}	220	Trapp (1986)
Water flea *Daphnia magna*	48-h LC_{50}	6530	Barata et al. (1998)
		6320	Poston et al. (1984)
		6400*	

*Value shown is the geomean of comparable endpoints; see Table 11 for details.

Table 16. Short-term CWQG for uranium resulting from the SSD method

	Concentration (µg U/L)
SSD 5th percentile	33
SSD 5th percentile, 90% LFL (5%)	9
SSD 5th percentile, 90% UFL (95%)	130

Note: LFL = lower fiducial limit, UFL = upper fiducial limit.

Table 17. Endpoints used in the SSD to determine the long-term CWQG for uranium

Species	Endpoint	Concentration (µg U/L)	Reference
Fish			
Rainbow trout *Oncorhynchus mykiss*	30-day EC_{10}‡ (non-viable embryos)	260	Vizon Scitech Inc (2004)
		480	Vizon Scitech Inc (2004)
		350*	
Fathead minnow *Pimephales promelas*	7-day LC_{10}‡ (survival)	1200	Vizon Scitech Inc (2004)
		1300	Vizon Scitech Inc (2004)
		760	Vizon Scitech Inc (2004)
		980	Vizon Scitech Inc (2004)
		1040*	
Northern pike *Esox lucius*	65-day MATC§ (growth)	2550	Liber et al. (2005)
Lake trout *Salvelinus namaycush*	141-day MATC§ (survival)	13 400	Liber et al. (2004a)
White sucker *Catostomus commersoni*	30-day MATC§ (growth)	14 300	Liber et al. (2004b)
Invertebrates			
Amphipod *Hyalella azteca*	28-day EC_{10}‡ (growth)	12	Liber et al. (2007)
Water flea *Ceriodaphnia dubia*	7-day EC_{10}‡ (reproduction)	1900	Liber et al. (2007)
		33	Vizon Scitech Inc (2004)
		59	Vizon Scitech Inc (2004)
		22	Vizon Scitech Inc (2004)
		25	Vizon Scitech Inc (2004)
		73*	
Water flea *Simocephalus serrulatus*	21-day EC_{10}‡ (reproduction)	480	Liber et al. (2007)
Water flea *Daphnia magna*	21-day EC_{10}‡ (reproduction)	570	Liber et al. (2007)
		123	Poston et al. (1984)
		373	Poston et al. (1984)
		1160	Poston et al. (1984)
		1360	Poston et al. (1984)
		530*	
Midge *Chironomus tentans*	28-day EC_{10}‡ (growth)	930	Liber et al. (2007)
Aquatic plants and algae			
Pseudokirchneriella subcapitata	72-h IC_{10}‡ (growth)	5.4	Vizon Scitech Inc (2004)
		55	Vizon Scitech Inc (2004)
		38	Vizon Scitech Inc (2004)
		54	Vizon Scitech Inc (2004)
		120	Vizon Scitech Inc (2004)
		57	Liber et al. (2007)
		40*	
Green algae *Cryptomonas erosa*	6-day IC_{10}‡ (growth)	172	Liber et al. (2007)
Macrophyte *Lemna minor*	7-day IC_{10}‡ (dry weight)	3100	Vizon Scitech Inc (2004)

*Value shown is the geomean of comparable endpoints; see Table 11 for details.
§MATC values calculated as the geomean of the reported NOEC/L and LOEC/L
‡Endpoint calculated from reported raw data in the original study

Table 18. Long-term CWQG for uranium resulting from the SSD method

	Concentration (µg U/L)
SSD 5th percentile	15
SSD 5th percentile, 90% LFL (5%)	8.5
SSD 5th percentile, 90% UFL (95%)	25

Note: LFL = lower fiducial limit, UFL = upper fiducial limit.

FIGURES

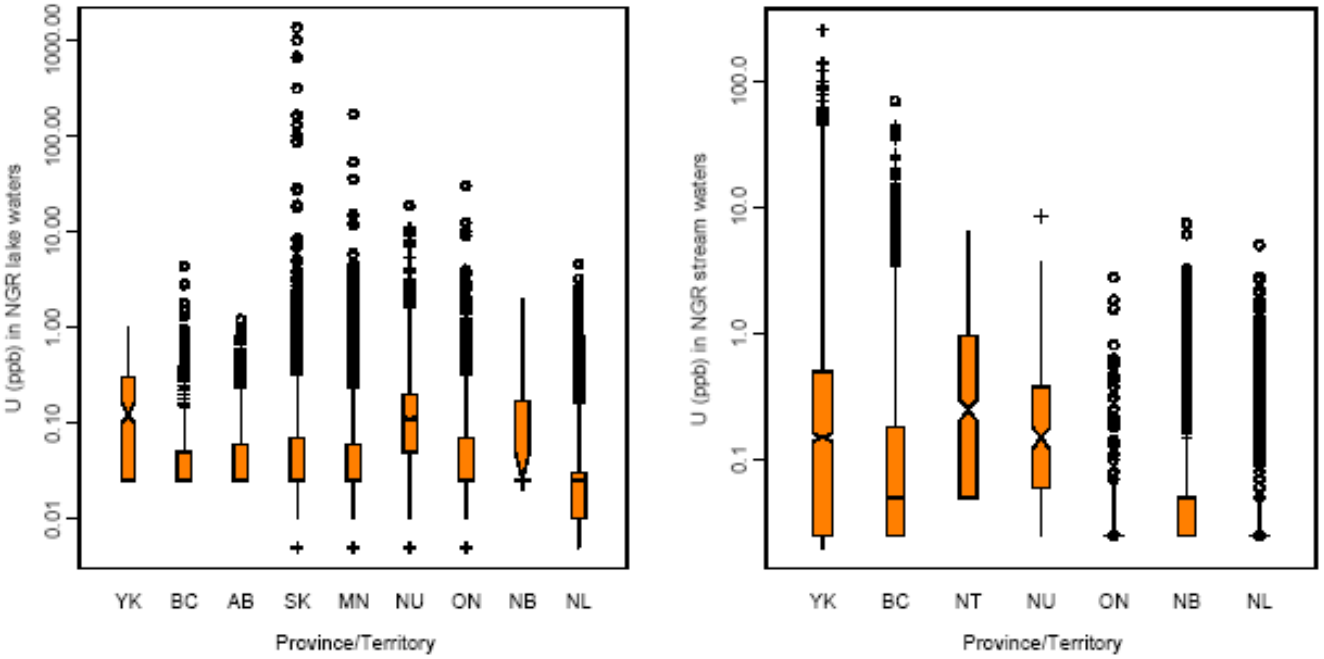

Figure 1. Uranium in National Geochemical Reconnaissance (NGR) lake and stream waters subdivided by province/territory and displayed as Tukey boxplots (RG Garrett, pers. comm. 2007).

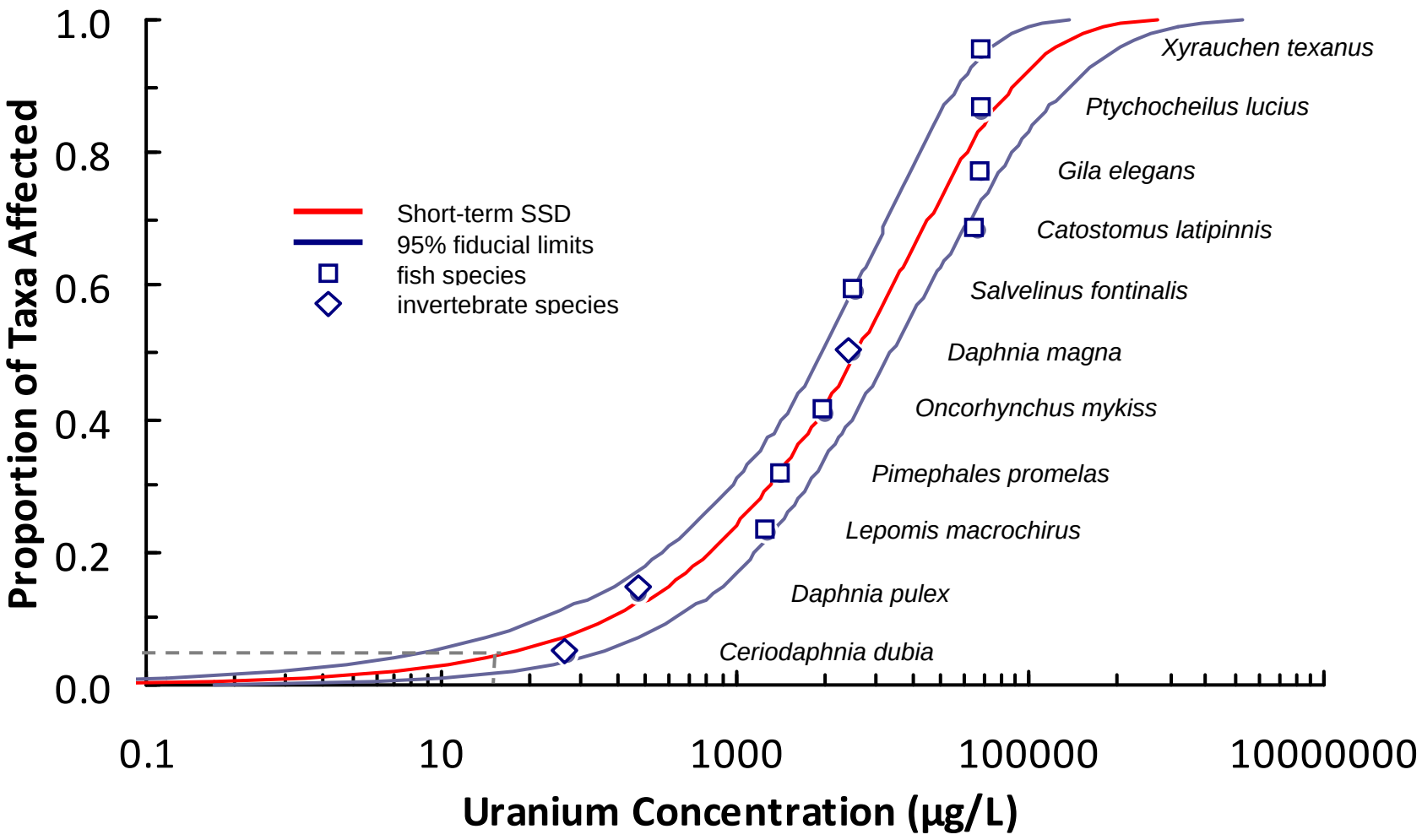

Figure 2. SSD for uranium in freshwater derived by fitting the log-Gompertz model to the short-term LC_{50}s of eleven (11) aquatic species versus Hazen plotting position. The intercept of the 5th percentile of the fitted curve (guideline value) was determined to be 33 µg U/L with 95% confidence interval of 9 and 130 µg U/L

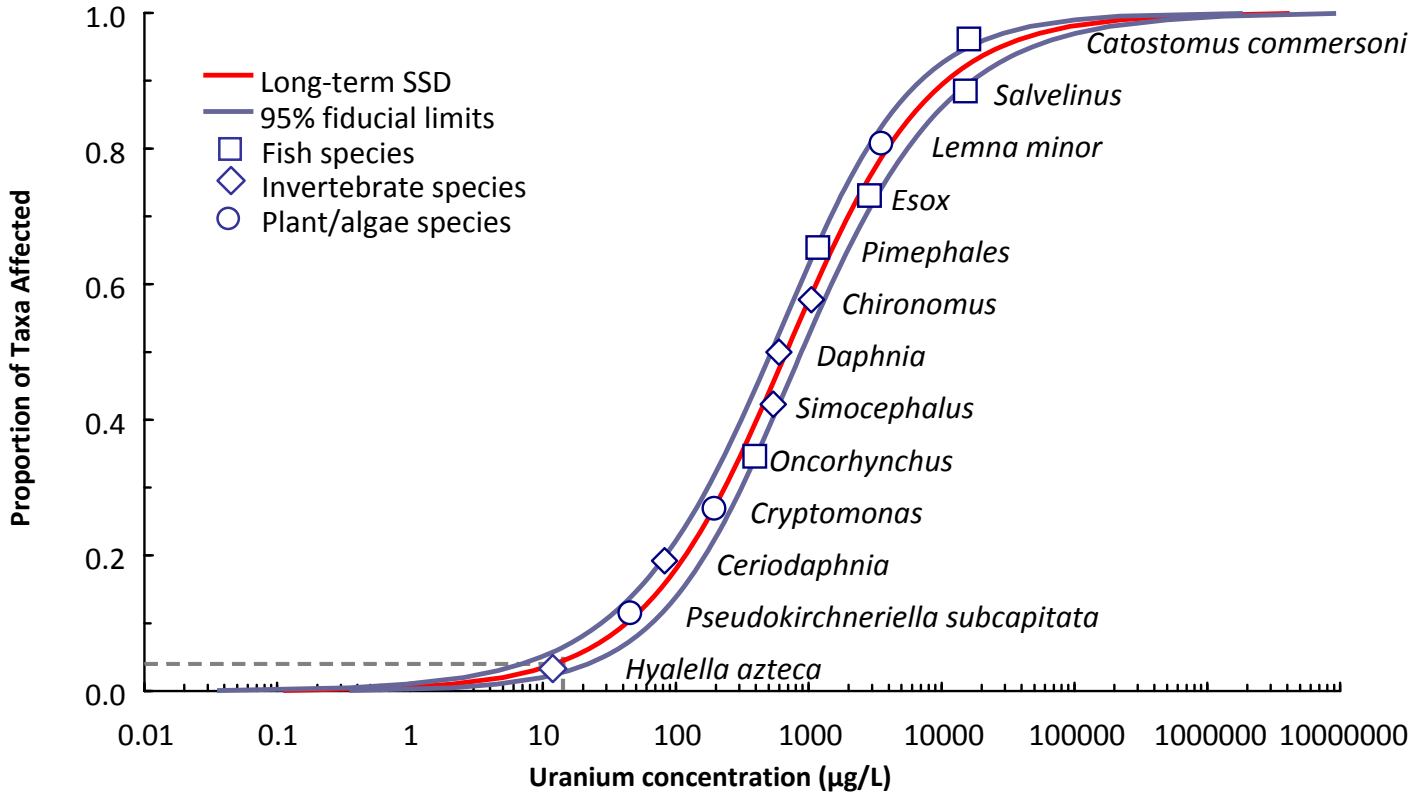

Figure 3. SSD for uranium in freshwater derived by fitting the log-logistic model to the long-term endpoints of thirteen (13) aquatic species versus Hazen plotting position. The intercept of the 5th percentile of the fitted curve (guideline value) was determined to be 15 µg U/L with 95% confidence interval of 8.5 and 25 µg U/L

APPENDICES

Appendix I : Resident Species Check

Table A 1-1. Resident species check for several suspected non-native species. The final decision reflects recent changes in the inclusion of non-resident species in guideline derivation

Common name	Species name	Reference	In NGSO species inventory table?	In textbooks or other references[1]?	Supplementary information from original study, references and the Internet	Decision[2]: (i) resident or suitable surrogate? (ii) suitable test conditions?
Chequered rainbowfish	*Melanotaenia splendida inornata*	(Bywater et al. 1991; Holdway 1992)	No	No	Specifically stated as "tropical" in document. Test temperatures were 27°C or 30°C	Not acceptable. (i) no (ii) no
Black-banded rainbowfish	*Melanotaenia nigrans*	(Bywater et al. 1991)	No	No	Specifically stated as "tropical" in document. Test temperature was 27°C.	Not acceptable. (i) no (ii) no
Northern purple-spotted gudgeon	*Mogurnda mogurnda*	(Bywater et al. 1991; Charles et al. 2002; Holdway 1992)	No	No	Specifically stated as "tropical" in document. Test temperatures were 27°C or 30°C.	Not acceptable. (i) no (ii) no
Reticulated perchlet	*Ambassis macleayi*	(Bywater et al. 1991)	No	No	Specifically stated as "tropical" in document. Test was run at 27°C.	Not acceptable. (i) no (ii) no
Delicate blue-eyes	*Pseudomugil tenellus*	(Bywater et al. 1991)	No	No	Specifically stated as "tropical" in document. Test temperature was 27°C.	Not acceptable. (i) no (ii) no
Mariana's hardyhead	*Craterocephalus marianae*	(Bywater et al. 1991)	No	No	Specifically stated as "tropical" in document. Test temperature was 27°C.	Not acceptable. (i) no (ii) no
Flannelmouth sucker	*Catostomus latipinnis*	(Hamilton and Buhl 1997)	Found "white sucker" *Catostomus commersoni*	Several *Catostomus* species found in Canada, but not this species.	This species was collected from New Mexico. Test temperature was 25°C.	Acceptable. (i) yes (ii) yes

Common name	Species name	Reference	In NGSO species inventory table?	In textbooks or other references[1]?	Supplementary information from original study, references and the Internet	Decision[2]: (i) resident or suitable surrogate? (ii) suitable test conditions?
Colorado squawfish or Colorado pikeminnow	*Ptychocheilus*[3] *lucius*	(Hamilton 1995)	No	*Ptychocheilus oregonensis* (northern squawfish) is found in BC and Alberta, but not Colorado squawfish. Scott and Crossman (1973) state that *P. lucius* occurs in "other west coast watersheds," but not in Canada.	This species is located in Utah. Comparative studies with surrogate species fathead minnow (*Pimephales promelas*) showed that the Colorado pikeminnow is similarly sensitive or slightly more sensitive than the fathead minnow to a variety of contaminants at 22°C (Sappington et al. 2001). Test temperature was 25°C.	Acceptable. (i) yes (ii) yes
Razorback sucker	*Xyrauchen texanus*	(Hamilton 1995)	No	No	This species is located in Utah. Comparative studies with surrogate species fathead minnow (*Pimephales promelas*) showed that the razorback is similarly sensitive or slightly more sensitive than the fathead minnow to a variety of contaminants at 22°C (Sappington et al. 2001). Test temperature was 25°C.	Acceptable. (i) yes (ii) yes
Bonytail or bonytail chub	*Gila elegans*	(Hamilton 1995)	No	No	This species is located in Utah. Comparative studies with surrogate species fathead minnow (*Pimephales promelas*) showed that the bonytail is similarly sensitive or slightly more sensitive than the fathead minnow to a variety of contaminants at 22°C (Sappington et al. 2001). Test temperature was 25°C.	Acceptable. (i) yes (ii) yes
Zebra fish	*Brachydanio rerio*	(Labrot et al. 1999)	Yes, but listed as a tropical species used for mosquito control	No	A few QC problems with this study. Species commonly used for genetics studies. Test temperature was not stated.	Not acceptable. (i) no (ii) n/a

Common name	Species name	Reference	In NGSO species inventory table?	In textbooks or other references[1]?	Supplementary information from original study, references and the Internet	Decision[2]: (i) resident or suitable surrogate? (ii) suitable test conditions?
Eastern mosquitofish	*Gambusia holbrooki* Girard 1859	(Keklak et al. 1994)	No	*Gambusia affinis* (mosquitofish) found in Alberta. Taxonomical differences between *G. holbrooki* and *G. affinis* recently changed. *G. holbrooki* may be present as an introduced species at Banff hotsprings.	Questionable data quality and non-standard endpoint.	Acceptable. (i) yes (ii) yes
(Invertebrate)	*Diaphanosoma excisum*	(Bywater et al. 1991)	No	No	When searched on Google, found in association with South Africa and tropical zoology reports. Collected in Australia. Test temperature was 27°C.	Not acceptable. (i) uncertain (ii) no
(Invertebrate)	*Latonopsis fasciculate*	(Bywater et al. 1991)	No	Distribution in Texas and Louisiana	Collected in Australia. Test temperature was 27°C.	Not acceptable. (i) uncertain (ii) no
(Invertebrate)	*Dadaya macrops*	(Bywater et al. 1991)	No	Distribution in Southern U.S.	Collected in Australia. Test temperature was 27°C.	Not acceptable. (i) uncertain (ii) no
(Invertebrate)	*Moinodaphnia macleayi*	(Bywater et al. 1991; Semaan et al. 2001)	No	Distribution in Louisiana and southward	Collected in Australia. Test temperature was 27°C.	Not acceptable. (i) uncertain (ii) no.
Green hydra	*Hydra viridissima* In *Cnidaria* taxa	(Hyne et al. 1992; Riethmuller et al. 2001)	*Hydra* sp. found with widespread distribution in North and South America including all of Canada.	Found under an old name (*Chlorohydra viridissima*) and common in U.S.	Note that *H. viridissima* is unique among *Hydra* sp. in that it lives symbiotically with algae (hence the green colour). When searched on Google, species found in South Dakota (which is at a similar latitude to Southern Ontario). Collected in Australia. Test temperatures were 27°C or 30°C.	Not acceptable. (i) yes (ii) no

Common name	Species name	Reference	In NGSO species inventory table?	In textbooks or other references[1]?	Supplementary information from original study, references and the Internet	Decision[2]: (i) resident or suitable surrogate? (ii) suitable test conditions?
Hydra	*Hydra vulgaris*	(Hyne et al. 1992)	*Hydra* sp. found.	Sources disagree on whether *Hydra vulgaris* is North American (see Thorpe and Covich 1991) or if *Hydra americana* has been misidentified as *H. vulgaris* in North America (see Edmondson 1959).	When searched on Google, the United States was listed as the origin for several strains of *H. vulgaris*. Test temperature was 30°C.	Not acceptable. (i) yes (ii) no
Bivalve	*Velesunio angasi*	(Markich et al. 2000)	No	No	When searched on Google, only comes up in tropical contexts. Collected in Australia. Test temperature was 28°C.	Not acceptable. (i) uncertain (ii) no
Bivalve	*Corbicula fluminea*	(Fournier et al. 2004; Labrot et al. 1999)	Yes Widespread distribution in North America, into drainages of west coast and southern tier of the States, but none in northern states.	Found; distribution in N. California and Oregon, introduced from China. Distribution coast to coast in States, but not in northern states. Distribution of *Corbicula* sp. considered widespread in North America. Clarke (1973) states that members of the same superfamily (Sphaeriacea) are found in the Canadian interior basin.	A few QC problems with Labrot et al. (1999)	Acceptable. (i) yes (ii) yes
Algae	*Chlorella* sp.	(Charles et al. 2002; Franklin et al. 2000; Hogan et al. 2005)	Yes	Found, but no distribution info given.	Test temperature was 27°C ± 1°C, indicating temperatures too high to be indicative of Canadian waters.	Not acceptable. (i) yes (ii) no

[1]Textbooks consulted include any and all of the following: Clarke (1973); Coad (1995); Edmondson (1959); Scott and Crossman (1973); Thorp and Covich (1991).

[2]Decision for non-resident species based on whether: (i) whether species was a suitable surrogate for temperate species in Canada; and (ii) test conditions are a suitable match for Canadian conditions. Based on informal surveys of opinion among National Guidelines and Standards Office (NGSO) scientists, it was decided that tests run at temperatures higher than 27°C would not be acceptable, but that tests conducted at 25°C would be acceptable. Acceptance of a study is also based on data quality, as previously outlined and scored in Table 11.
[3]Hamilton (1995) spells this species *Ptychocheilus lucius*, as do Scott and Crossman (1973). Sappington et al. (2001) have apparently misspelled it as *Ptychochelius lucius*.

Appendix II : Comparing Changes in Uranium Speciation with Toxicity Endpoints

Background: Because of the chemistry of uranium, free ion (UO_2^{2+}) may not be constant with total uranium in the range in which toxicity occurs. This lack of linear increase in free ion with total (linear increases in) uranium may add complexity to the interpretation of toxicity tests.

Objective: To compare the relative changes in percent free uranyl with uranium toxicity endpoints, both with respect to total uranium expressed in step intervals.

Methods: Three papers can be used to meet the objectives of the speciation analysis: Markich et al. (2000) and, to a lesser extent, Barata et al. (1998), Franklin et al. (2000) and Charles et al. (2002). The paper by Markich et al. (2000) has the most appropriate array of speciation graphs, so it was chosen as the main speciation study. For simplicity, the uranyl ion (UO_2^{2+}) is taken as the only toxic species, although one study suggests that UO_2OH^+ also contributes to toxicity (Markich et al. 2000). Toxicity results from these three studies, as well as from other toxicity studies, are then compared with changes in uranyl ion concentration with total uranium.

To simplify the modelling results, which are normally presented on a continuous scale, speciation data were grouped into somewhat arbitrary step intervals (based on main speciation publication, and known toxicity endpoint values). One thousand µg/L "steps" in total uranium concentration were taken between 1000 µg/L and 4000 µg/L, with higher resolution (100 and 500 µg/L interval) between 0 µg/L and 1000 µg/L where sensitive species may be affected. Also, Markich et al. (2000) include modelling results with fulvic acid, but since fulvic acid (or other natural organic matter) is measured in only two toxicity tests, those modelling results are excluded for clarity.

Results: When comparing relative change in uranyl free ion with respect to total uranium in the step intervals chosen, there are no substantial changes in free uranyl ion ($\leq$ 3%) (Markich et al. (2000) study in Table A 2-1). Further, the changes in relative percent of free uranyl ion are not predictive of the number of toxicity endpoints that fall in the respective total uranium interval. There is not enough information in the three other studies to perform a similar comparison.

Table A 2-1. Changes in percent of free uranyl ion as a function of total uranium (in step intervals), and comparison to observed uranium effects endpoints. Effects endpoints include primary and secondary data, grouped by short-term and long-term exposures. Studies referenced for speciation data include short-term bivalve *Velesunio angasi*, long-term algae (with 72-h exposure), and short-term *Daphnia magna* tests.

Total uranium (µg/L)	Change[1] in UO_2^{2+} (%)	Water Chemistry			Effect endpoint in this study (bivalve)[2]	Reference[3]
		pH	Hardness ($CaCO_3$)	Alkalinity ($CaCO_3$)		
0–100	< 1	5	3.71	--	MDEC = 84 µg/L EC_{50} = 103 µg/L	(Markich et al. 2000)
100–500	2					
500–1000	2					
1000–2000	3					
2000–3000	3					
3000–4000	3					
0–100	3	6	3.71	--	MDEC = 388 µg/L EC_{50} = 559 µg/L	
100–500	2					
500–1000	1					
1000–2000	< 1					
2000–3000	< 1					
3000–4000	< 1					
Supplementary information from Franklin et al. (2000): no speciation graphs available, MDEC values also calculated						
0.1–1000	11	5.7	3.91	--	EC_{50} = 78 µg/L (algae)	(Franklin et al. 2000)
0.1–1000 (assumed)	< 1	6.5	3.91	--	EC_{50} = 44 µg/L (algae)	
Supplementary information from Charles et al. (2000): no speciation graphs available, MDEC values also calculated						
0–0.5	< 1	7.0	8, 40, 100, 400	--	EC_{50} = 56 000–270 000 µg/L depending on hardness (algae)	(Charles et al. 2002)
Supplementary information from Barata et al. (1998): no speciation graphs available						
Not reported	< 1	7.73	90.7	62.1	LC_{50} was between 5180 and 8250 µg/L depending on clone (*Daphnia magna*)	(Barata et al. 1998)
Not reported	< 1	8.07	179	126	LC_{50} was between 15 300 and 22 400 µg/L depending on clone (*Daphnia magna*)	

[1] Unless otherwise noted, from Markich et al. (2000), estimated from graphs to the nearest one percent. Results shown are the difference between the lower bracket and the upper bracket of the interval.
[2] Only organisms with an effect concentration in the corresponding "Total uranium" range (column one) were recorded. Water quality parameters (such as pH, hardness and alkalinity) in these other studies were not taken into consideration.
[3] Speciation values were calculated by the National Guidelines and Standards Office based on values provided by the original studies.

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