Canadian Environmental Protection Act, 1999

Draft Ecological Screening Assessment Report on 4,6-dinitro-o-cresol (DNOC)

July 2006
Introduction

The *Canadian Environmental Protection Act, 1999* (CEPA 1999) requires the Minister of the Environment and the Minister of Health to conduct screening assessments of substances that meet the categorization criteria set out in the Act and Regulations to determine, in an expeditious manner, whether substances present or may present a risk to the environment or to human health. Based on the results of a screening assessment, the Ministers can propose taking no further action with respect to the substance, adding the substance to the Priority Substances List (PSL) for further assessment or recommending that the substance be added to the List of Toxic Substances in Schedule 1 and, where applicable, the implementation of virtual elimination.

A screening assessment involves an evaluation of a substance to determine whether the substance is “toxic” or capable of becoming “toxic” as defined in CEPA 1999. This ecological screening assessment examines various supporting information and develops conclusions based on a weight of evidence approach as specified under Section 76.1 of CEPA 1999. The screening assessment report does not present an exhaustive review of all available data; rather, it presents the most critical studies and lines of evidence supporting the conclusions. One line of evidence includes consideration of risk quotients to identify potential for ecological effects. However, other concerns that affect current or potential risk, such as persistence, bioaccumulation, chemical transformation and trends in ambient concentrations, are also considered.

4,6-Dinitro-\(o\)-cresol (DNOC) was included in a pilot project of 123 substances for screening assessment under CEPA 1999, on the basis of its anticipated high potential for human exposure. DNOC is also considered to be persistent and inherently toxic to aquatic organisms.

Data relevant to the ecological screening assessment of DNOC were identified in original literature, review documents and commercial and government databases. Searches were conducted of the open literature, conference proceedings and the Internet for relevant information. Data obtained as of August 2004 were considered in this document. Original studies that form the basis for determining whether the substance is “toxic” under CEPA 1999 have been critically evaluated by Environment Canada. The data from key toxicity studies were evaluated using Robust Study Summary forms similar to those recommended by the Organisation for Economic Co-operation and Development for the evaluation of studies for the Screening Information Data Sets of high production volume substances (OECD, 2003).

This ecological screening assessment report and associated unpublished supporting working documentation were written by Environment Canada evaluators under the lead of Barbara Elliott. The substance matter in this report has been subjected to a science review by
individuals having relevant technical expertise, including Pierrette Blanchard (Meteorological Service of Canada, Environment Canada), Mark McMaster (National Water Research Institute, Environment Canada) and Suzanne Lesage (National Water Research Institute, Environment Canada).

The conclusions of the ecological and human health screening assessment reports were approved by the joint Environment Canada/Health Canada CEPA Management Committee. The supporting working documentation for the ecological assessment is available upon request by e-mail from existing.substances.existantes@ec.gc.ca. The supporting working documentation for the human health assessment is available upon request by e-mail from ExSD@hc-sc.gc.ca. Information on ecological and human health screening assessments under CEPA 1999 may be linked from the CEPA Registry at http://www.ec.gc.ca/ceparegistry.

The critical information and considerations upon which the assessment is based are summarized below.

Properties of DNOC

DNOC (CAS No. 534-52-1) is a yellow crystalline solid at ambient temperature (U.S. EPA, 1986a). It is also referred to as 2,4-dinitro-6-methylphenol; phenol, 2-methyl-4,6-dinitro-; 3,5-dinitro-2-hydroxytoluene; 4,6-dinitro-2-methylphenol; 6-methyl-2,4-dinitrophenol; Antinonin; Antinonnin; Arborol; Degrassan; Dekrysil; Detal; Dillex; and Kreozan.

Physical and chemical properties of DNOC, both measured and predicted, are presented in Table 1.

Table 1: Physical and chemical properties of DNOC

<table>
<thead>
<tr>
<th>Property</th>
<th>Measured Value</th>
<th>Predicted Model</th>
</tr>
</thead>
<tbody>
<tr>
<td>Molecular formula</td>
<td>C₇H₆N₂O₅</td>
<td></td>
</tr>
<tr>
<td>Molecular mass</td>
<td>198.1348</td>
<td></td>
</tr>
<tr>
<td>Melting point (°C)</td>
<td>86.6</td>
<td></td>
</tr>
<tr>
<td></td>
<td>85.8</td>
<td></td>
</tr>
<tr>
<td>Boiling point (°C)</td>
<td>378</td>
<td></td>
</tr>
<tr>
<td></td>
<td>220</td>
<td></td>
</tr>
<tr>
<td>Density (g/cm³)</td>
<td>1.58 at 20°C</td>
<td></td>
</tr>
<tr>
<td>Vapour pressure (Pa)</td>
<td>4.31 × 10⁻² at 20°C</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.40 × 10⁻² at 25°C</td>
<td></td>
</tr>
<tr>
<td></td>
<td>4.79 × 10⁻² at 35°C</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.6 × 10⁻² at 25°C</td>
<td></td>
</tr>
<tr>
<td>Henry’s law constant (Pa·m⁻³/mol)</td>
<td>1.4 × 10⁻¹ at 25°C</td>
<td></td>
</tr>
<tr>
<td></td>
<td>3.09 × 10⁻³ at 20°C</td>
<td></td>
</tr>
<tr>
<td></td>
<td>4.94 × 10⁻⁶</td>
<td></td>
</tr>
<tr>
<td></td>
<td>6.49 × 10⁻⁵</td>
<td></td>
</tr>
<tr>
<td></td>
<td>4.3 × 10⁻² at 20°C</td>
<td></td>
</tr>
</tbody>
</table>

### Measured and Predicted Values

<table>
<thead>
<tr>
<th>Property</th>
<th>Measured (Value, References)</th>
<th>Predicted (Value, Model)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Octanol/water partition coefficient (log K_{ow})</td>
<td>2.564 (2.12, 7)</td>
<td>2.7 (EPIWIN v. 3.11)</td>
</tr>
<tr>
<td></td>
<td>1.78 at pH 4</td>
<td>1.91 (PALLAS v. 4.0)</td>
</tr>
<tr>
<td></td>
<td>0.087 at pH 7</td>
<td>1.61 (Hansch MMP v. 1.44)</td>
</tr>
<tr>
<td></td>
<td>1.32 at pH 10</td>
<td>2.1366 (Ghose MMP v. 1.44)</td>
</tr>
<tr>
<td>Water solubility (mg/L)</td>
<td>1000 at 15°C</td>
<td>691.9 (EPIWIN v. 3.11)</td>
</tr>
<tr>
<td></td>
<td>198 at 20°C</td>
<td>467.3 (EPIWIN v. 3.11)</td>
</tr>
<tr>
<td></td>
<td>21.3 at pH 4</td>
<td>48 875.4 (MMP v. 1.44)</td>
</tr>
<tr>
<td></td>
<td>694 at pH 7</td>
<td></td>
</tr>
<tr>
<td></td>
<td>3300 at pH10</td>
<td></td>
</tr>
<tr>
<td></td>
<td>100 000 (DNOC Na salt)</td>
<td></td>
</tr>
<tr>
<td>Other solubilities (g/g)</td>
<td>4.3/100 (ethanol)</td>
<td>691.9 (EPIWIN v. 3.11)</td>
</tr>
<tr>
<td></td>
<td>100/100 (acetone)</td>
<td>467.3 (EPIWIN v. 3.11)</td>
</tr>
<tr>
<td></td>
<td>37/100 (benzene)</td>
<td>48 875.4 (MMP v. 1.44)</td>
</tr>
<tr>
<td>Organic carbon/water partition coefficient (log K_{oc})</td>
<td>2.35–2.77 (2.564, 4)</td>
<td>2.779 (EPIWIN v. 3.11)</td>
</tr>
<tr>
<td></td>
<td>1.3 (DNOC Na salt)</td>
<td>2.58 (ASTER)</td>
</tr>
<tr>
<td>Solid/water distribution ratio (K_d)</td>
<td>&lt;5 (loam soil with 0.8–3%)</td>
<td>4.32 (PALLAS v. 4.0)</td>
</tr>
<tr>
<td></td>
<td>1.93</td>
<td></td>
</tr>
<tr>
<td>pKa</td>
<td>4.4</td>
<td>4.32 (PALLAS v. 4.0)</td>
</tr>
<tr>
<td></td>
<td>4.31</td>
<td></td>
</tr>
<tr>
<td>Air/water partition coefficient (K_{aw})</td>
<td>3.0 × 10^{-5} at 25°C</td>
<td></td>
</tr>
</tbody>
</table>

1. References:
   1. ChemFinder
   2. SRC PhyProp Database
   3. Verschueren, 2001
   4. HSDB
   5. ATSDR, 1995
   6. IPCS, 2000
   7. Schwarzenbach et al., 1988
   8. UNEP/FAO, 2002
   9. Vogue et al., 1994
   12 Schwarzenbach et al., 2003

### Scope of Assessment

Owing to the chemical nature of DNOC, it readily forms water-soluble sodium, potassium and ammonium salts, and virtually 100% of dissolved DNOC will be in the ionized form at environmentally relevant pHs (pH 6–8). Based on this information, Environment Canada reviewed the use of DNOC salts to determine if they should also be included in this screening.
assessment. The sodium, potassium and ammonium salts of DNOC are not on the Domestic Substances List (DSL), although DNOC sodium salt is on the Non-Domestic Substances List (NDSL). If a company were intending to manufacture or import these substances, they would be considered to be new to Canada and subject to notification under the New Substances Notification Regulations. Therefore, the ecological screening assessment focused on the uses of DNOC, but a review of the fate and effects of its salts was also carried out.

DNOC used to be registered as a pesticide under the Pest Control Products Act (PCPA); however, its registration was discontinued in December 1990. Three of the DNOC salts (sodium, ammonium and potassium) have been used as pesticides internationally but were never registered under the PCPA in Canada.

**Manufacture, Importation and Use of DNOC**

Globally, DNOC is used mainly in the plastics industry as an inhibitor of polymerization in styrene and aromatic vinyl products; it is also used as an intermediate in the synthesis of fungicides, dyes and pharmaceuticals (IPCS, 2000; UNEP/FAO, 2002).

Two Canadian companies reported manufacture or import of DNOC in 1986, with amounts in the range of 100–1000 tonnes (Environment Canada, 1990). Historically, DNOC was used in Canada as an antioxidant, corrosion inhibitor, tarnish inhibitor and antiscaling agent, for a total of 99.9% of the Canadian market (Environment Canada, 1990).

An industry survey conducted by Environment Canada resulted in use pattern information for the year 2000 (Environment Canada, 2003a). Under Section 71 of CEPA 1999, this Notice with Respect to Certain Substances on the Domestic Substances List applied to any person who, during the 2000 calendar year, manufactured or imported DNOC, whether alone or in a mixture or in a product, in a total quantity greater than 10 000 kg. The survey indicated that DNOC is not manufactured in Canada; however, between 100 and 1000 tonnes of DNOC were imported in 2000 (Environment Canada, 2003a).

**Releases of DNOC**

Only one company, located on the St. Clair River near Sarnia, Ontario, reported releases of DNOC to the National Pollutant Release Inventory (NPRI) (Environment Canada, 2003b). Although the facility reported only off-site transfers and no releases to water, air or soil, it was used to develop a conservative scenario to estimate the quantity of DNOC that could be released into the St. Clair River from such a site. It is assumed, in a conservative scenario, that one customer is receiving the total annual import quantity (100–1000 tonnes). The scenario assumes releases of 0.2% of the annual import quantity of DNOC into the St. Clair River, which, based on professional judgement, recognizes routine process losses and waste from equipment cleaning for a substance handled in bulk. This accounts for releases to solid waste and wastewater. Using this percentage results in an estimated annual release of 200–2000 kg. It is further assumed that DNOC is in use throughout the year and that there is continuous release (24 hours per day) over
the year (350 operating days). Daily releases over a period of 350 days correspond to approximately 0.57–5.7 kg/day. Sewage treatment plant (STP) removal rates were also considered. The STP model (CEMC, 2001) estimated that 27% of DNOC would be removed and that 73% would enter the environment in the form of final wastewater effluent from an STP.

Two main sources of atmospheric nitrophenols (a category that includes DNOC) have been reported in the literature. These include secondary formation by reactions in the troposphere and emissions from automobiles. Researchers have examined the atmospheric occurrence and formation of DNOC (Nojima et al., 1976; Alber et al., 1989; Richartz et al., 1990). DNOC has been shown to form as a secondary pollutant via the reaction of toluene and 2-methylphenol with nitrogen monoxide and hydroxyl radicals. It is difficult to estimate the quantity that may result from the anthropogenic release of precursor species. Direct emission of DNOC from car exhaust is likely only of minor importance. Under experimental conditions, exhaust from an automobile motor was found to contain DNOC at <0.01 ng/m$^3$ (Tremp et al., 1993).

**Disposal of DNOC**

The NPRI (Environment Canada, 2003b) reported that amounts of up to 2 tonnes of DNOC and its salts were annually “transferred for disposal” by Nova Chemicals (Table 2). For all years before 2002, the methods of treatment were biological, such as biooxidation, and incineration or thermal. For the year 2002, disposal was to a landfill.

Table 2: NPRI data for DNOC and its salts (all reporting is for the Nova Chemicals plant in Sarnia, Ontario)

<table>
<thead>
<tr>
<th>Year</th>
<th>Transfers for disposal (tonnes)</th>
<th>Biological treatment (tonnes)</th>
<th>Incineration (tonnes)</th>
<th>Landfill (tonnes)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2002</td>
<td>1.49</td>
<td></td>
<td></td>
<td>1.49</td>
</tr>
<tr>
<td>2001</td>
<td>0.01</td>
<td>0.01</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2000</td>
<td>0.25</td>
<td>0.03</td>
<td>0.22</td>
<td></td>
</tr>
<tr>
<td>1999</td>
<td>0.02</td>
<td>0.02</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1998</td>
<td>0.68</td>
<td>0.04</td>
<td>0.64</td>
<td></td>
</tr>
<tr>
<td>1997</td>
<td>0.36</td>
<td>0.06</td>
<td>0.3</td>
<td></td>
</tr>
<tr>
<td>1996</td>
<td>2.00</td>
<td>0.08</td>
<td>1.92</td>
<td></td>
</tr>
<tr>
<td>1995</td>
<td>0.03</td>
<td>0.03</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1994</td>
<td>0.04</td>
<td>0.04</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Environmental Fate and Partitioning**

Environmental fate analysis integrates information on the chemical behaviour of the substance with the properties of the receiving environment. The objective of fate analysis is to determine the multimedia distribution of the substance after its release into the environment. This includes consideration of the persistence and bioaccumulation of the substance in the environment.
Level III fugacity modelling (CEMC, 2002) predicts the following fate for DNOC (estimated for the neutral form of DNOC) released into different media:

- With releases solely to water (as would be expected from the stated use), the majority of DNOC would remain in water (99%), with a minor amount partitioning to sediments (1%). Loss of the substance from the aquatic environment at steady state will be mainly a result of transport (~55% advection) out of the “unit world” and degradation (~45%).
- With emissions solely to air, the majority of DNOC would partition to soil (91%) and water (7%).
- With releases solely to soil, the majority of DNOC would remain in soil (98%), with some transfer to water (2%).

The following information was considered in evaluating whether DNOC meets the criteria for persistence and bioaccumulation as defined under the Persistence and Bioaccumulation Regulations of CEPA 1999:

- **Persistence:**
  a) in air, estimated half-life of 8 hours to 129 days (U.S. EPA, 1986b; Atkinson, 1987; Howard et al., 1991);
  b) in surface water, half-life ranges from 3 to 5 weeks (Mill and Mabey, 1985; Kelly et al., 1994; IPCS, 2000);
  c) in sediments, half-life is estimated to be 150 days (EPI version 3.11);
  d) in soil, half-life ranges from 14 hours to less than 2 months (Callahan et al., 1979; HSDB).

- **Bioaccumulation:**
  a) bioaccumulation factor (BAF) of 25 calculated for DNOC (Gobas and Arnot, 2003).

Persistence criteria are half-lives of greater than or equal to 2, 182, 365 and 182 days for air, water, sediment and soil, respectively.

Bioaccumulation criteria are BAFs or bioconcentration factors (BCFs) of greater than or equal to 5000 or a log $K_{ow}$ of greater than or equal to 5.0.

**Based on the Persistence and Bioaccumulation Regulations, DNOC is persistent in air and not bioaccumulative.**

**Environmental Concentrations**

**Atmosphere and Precipitation**

No monitoring data for DNOC in the atmosphere or precipitation in Canada were identified. Monitoring data from other countries are summarized in Table 3.
Table 3: Concentrations of DNOC in the atmosphere and precipitation

<table>
<thead>
<tr>
<th>Location</th>
<th>Sampling period</th>
<th>No. of samples</th>
<th>Detection limit (µg/L)</th>
<th>Mean concentration (µg/L)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Denmark</td>
<td>October–November 2001</td>
<td>5</td>
<td>ns</td>
<td>[0.07–3.2 ng/m³]</td>
<td>Bossi and Andersen, 2003</td>
</tr>
<tr>
<td>The Netherlands</td>
<td>2000–2001</td>
<td>18</td>
<td>ns</td>
<td>&gt;0.1</td>
<td>Duyzer and Vonk, 2002</td>
</tr>
<tr>
<td>Italy, Milan</td>
<td>November 1998</td>
<td>12</td>
<td>ns</td>
<td>[600–7200], rainwater</td>
<td>Belloli et al., 2000</td>
</tr>
<tr>
<td>Germany, Bavaria, rainwater</td>
<td>1995–1998</td>
<td>ns</td>
<td>ns</td>
<td>[0.1–2.4] (approximated from graph)</td>
<td>Schüssler and Nitschke, 2001</td>
</tr>
<tr>
<td>Germany, Bavaria</td>
<td>July 1998 – March 1999</td>
<td>&gt;100</td>
<td>ns</td>
<td>3.4 [0.5–4.2], fogwater</td>
<td>Römpp et al., 2001</td>
</tr>
<tr>
<td>Germany, Hanover, rain and snow</td>
<td>1988</td>
<td>ns</td>
<td>0.1–1.0</td>
<td>Qualitatively identified</td>
<td>Alber et al., 1989</td>
</tr>
<tr>
<td>England, Great Dun Fell</td>
<td>April–May 1993</td>
<td>6</td>
<td>ns</td>
<td>0.7 [0.26–2.13], cloudwater</td>
<td>Lüttke and Levsen, 1997</td>
</tr>
<tr>
<td>Germany, Mount Brocken</td>
<td>June 1994</td>
<td>6</td>
<td>ns</td>
<td>4.2 [0.1–10], cloudwater</td>
<td>Lüttke et al., 1999</td>
</tr>
<tr>
<td>Switzerland, Dübendorf, rain</td>
<td>March–November 1985</td>
<td>3</td>
<td>ns</td>
<td>0.05 µg/m³, ambient air</td>
<td>Leuenberger et al., 1988</td>
</tr>
<tr>
<td>and ambient air</td>
<td></td>
<td></td>
<td></td>
<td>[0.95–1.6 µg/L], rain</td>
<td></td>
</tr>
</tbody>
</table>

1 ns = not specified.
2 Unless otherwise specified. The range of values is indicated in square brackets, if available (e.g., [minimum–maximum]).

DNOC has been detected in atmospheric air and precipitation at a number of locations in Europe, and the presence of nitrated phenols in rain is not explained solely by input from pesticide applications (Leuenberger et al., 1988). DNOC has been shown to partition favourably from the gas phase into the aqueous phase, and it is not surprising that the substance has been found in rainwater (Schwarzenbach et al., 2003). DNOC was detected in Denmark, even though the substance has not been used for the last 10 years (Danish Environmental Protection Agency, 2001). The concentrations found in rain in Denmark are of the same order of magnitude as have been detected in England, Germany and Switzerland.

As no atmospheric or precipitation monitoring data for DNOC in Canada could be located, a series of release scenarios were developed to estimate the amount of DNOC that could be released into receiving waters in Canada as a result of rainfall scavenging of DNOC in the atmosphere. The scenarios incorporated precipitation data for 12 Canadian cities, an estimate of the amount of DNOC in rainwater and a calculation of runoff from built-up and natural areas into the receiving STPs. It was assumed that the rain event that would result in DNOC being removed from the atmosphere would be a heavy rainfall and that DNOC would be washed out in the early stages of the rain event and not over the length of the rainfall. The concentration of DNOC used in the scenario is based on precipitation values from Europe that were considered realistic possible levels of DNOC in air in Canada. The mean concentration of DNOC in cloudwater from
northern Germany (4.2 µg/L) was selected. It was assumed that rainwater would be released as a point source from an STP, but that it would not undergo STP treatment, as STP removal efficiency during a storm event is likely to be poor. The highest concentrations of DNOC were estimated in receiving waters from the STPs in London, Ontario (0.0023 mg/L), Guelph, Ontario (0.0023 mg/L), and Granby, Quebec (0.0025 mg/L).

Aquatic Concentrations

No recent aquatic monitoring data for DNOC in Canada were identified. Older data on levels of DNOC in Canadian waters as well as in other countries are summarized in Table 4.

Table 4: Concentrations of DNOC in surface water

<table>
<thead>
<tr>
<th>Location</th>
<th>Sampling period</th>
<th>No. of samples</th>
<th>Detection limit (µg/L)</th>
<th>Mean concentration (µg/L)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Italy, River Po</td>
<td>January 1994 – December 1996</td>
<td>ns</td>
<td>0.1</td>
<td>nd</td>
<td>Davi and Gnudi, 1999</td>
</tr>
<tr>
<td>Germany, Elbe River</td>
<td>1994</td>
<td>ns</td>
<td>0.05</td>
<td>[ns– 0.06]</td>
<td>Pietsch et al., 1995</td>
</tr>
<tr>
<td>Denmark, Hølvads Rende area, soil water, drainage water, stream water</td>
<td>October 1989 – December 1991</td>
<td>ns</td>
<td>ns</td>
<td>0.005 (soil water) nd (drainage water) [0.02–0.16] (stream water)</td>
<td>Mogensen and Spliid, 1995</td>
</tr>
<tr>
<td>Denmark, Bolbo Bæk area, soil water, stream water</td>
<td>April 1990 – December 1991</td>
<td>ns</td>
<td>ns</td>
<td>0.005 (soil water) 0.16 (stream water)</td>
<td>Mogensen and Spliid, 1995</td>
</tr>
<tr>
<td>Denmark, four ponds</td>
<td>November 1989 – December 1990</td>
<td>ns</td>
<td>ns</td>
<td>[nd–0.64]</td>
<td>Mogensen and Spliid, 1995</td>
</tr>
<tr>
<td>The Netherlands, Meuse River and Rhine River; Slovakia, Danube River and Nitra River</td>
<td>ns</td>
<td>4</td>
<td>0.4</td>
<td>nd</td>
<td>Brouwer and Brinkman, 1994</td>
</tr>
<tr>
<td>Germany, Bavaria, Mount Ochsenkopf and University of Bayreuth campus</td>
<td>Fall 1988</td>
<td>ns</td>
<td>1.98</td>
<td>[nd–12.5]</td>
<td>Richartz et al., 1990</td>
</tr>
<tr>
<td>Point source</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ontario, St. Clair River near Sarnia (industrial area)</td>
<td>1979</td>
<td>24</td>
<td>1</td>
<td>[nd–10]</td>
<td>Munro et al., 1985</td>
</tr>
</tbody>
</table>
### Sampling Data

<table>
<thead>
<tr>
<th>Location</th>
<th>Sampling period</th>
<th>No. of samples</th>
<th>Detection limit (µg/L)</th>
<th>Mean concentration (µg/L)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ontario, St. Clair River near Sarnia (industrial area)</td>
<td>1980</td>
<td>25</td>
<td>1</td>
<td>nd</td>
<td>Munro et al., 1985</td>
</tr>
<tr>
<td>Ontario, St. Clair River near Sarnia, industrial effluent, process/sewer water, township ditch water</td>
<td>1979</td>
<td>119</td>
<td>1</td>
<td>[nd–10 000]</td>
<td>Munro et al., 1985</td>
</tr>
<tr>
<td>Ontario, St. Clair River near Sarnia, industrial effluent, process/sewer water, township ditch water</td>
<td>1980</td>
<td>61</td>
<td>1</td>
<td>nd</td>
<td>Munro et al., 1985</td>
</tr>
<tr>
<td>United States, California groundwater</td>
<td>ns</td>
<td>ns</td>
<td>ns</td>
<td>ns–35</td>
<td>Hallberg, 1989</td>
</tr>
<tr>
<td>Italy, Taranto, surface seawater contaminated by oil refinery or iron and steel factory wastes</td>
<td>ns</td>
<td>2</td>
<td>0.017</td>
<td>[0.030–0.065]</td>
<td>Cardellicchio et al., 1997</td>
</tr>
<tr>
<td>Unspecified location, oil refinery effluent, paper mill effluent</td>
<td>ns</td>
<td>ns</td>
<td>0.5</td>
<td>nd</td>
<td>Paterson et al., 1996</td>
</tr>
</tbody>
</table>

1 ns = not specified; nd = not detected.
2 The range of values is indicated in square brackets, if available (e.g., [minimum–maximum]).
3 Mean concentration in effluent is presented as an indication of resulting exposure. This value was not included in the section on releases of DNOC, as details on effluent quantities and release rate were not provided.

As no recent Canadian surface water monitoring data were identified, aquatic exposure estimates were modelled. The scenario uses the ChemSim model (Environment Canada, 2003c) to predict estimated exposure values. ChemSim model runs were done for three river flow estimates and two loading rates (calculated in the section on releases of DNOC), for a total of six model runs. As indicated in the release scenario, it is assumed that DNOC is in use throughout the year and that there is continuous release (24 hours per day) over the year (350 operating days). Two estimates of low river flow (2.5th and 10th percentiles) were selected to derive Estimated Exposure Values (EEVs) under low-flow conditions. The 50th-percentile flow value was also selected to estimate EEVs under more typical conditions. The maximum concentration of DNOC at 20 m downstream of the reporting facility with a worst-case scenario release of 5.7 kg/day and a 2.5th-percentile river flow is estimated to be less than 0.006 mg/L. If STP treatment is considered, an EEV of 0.0014 mg/L is estimated.

### Sediment, Sewage Sludge and Soil

Monitored soil, sediment and sludge concentrations of DNOC are summarized in Table 5. The high flow and velocity of the St. Clair River rapidly dilute and disperse the substance.
Therefore, loadings of DNOC do not appear likely to cause significant exposure to benthic organisms, as only a minor amount of DNOC is expected to partition to sediments (1%). Based on the results of modelling, at a release rate of 5.7 kg/day, 0.057 kg/day (or 1%) would be available to be adsorbed onto sediments.

Table 5: Concentrations of DNOC in soil, sediment and sludge

<table>
<thead>
<tr>
<th>Location</th>
<th>Sampling period</th>
<th>No. of samples</th>
<th>Detection limit(^1) (ng/g)</th>
<th>Mean concentration(^2) (ng/g)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ontario, old urban parkland soil</td>
<td>ns</td>
<td>60</td>
<td>100</td>
<td>Ontario typical range</td>
<td>OMEE, 1994</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>&lt;W(^3)</td>
<td></td>
</tr>
<tr>
<td>Ontario, rural parkland soil</td>
<td>ns</td>
<td>101</td>
<td>100</td>
<td>Ontario typical range</td>
<td>OMEE, 1994</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>&lt;W(^3)</td>
<td></td>
</tr>
<tr>
<td>Canada, agricultural soil</td>
<td>ns</td>
<td>30</td>
<td>50</td>
<td>nd</td>
<td>Webber, 1994</td>
</tr>
<tr>
<td>11 sites across Canada, sludge samples</td>
<td>September 1993–February 1994</td>
<td>12 samples/site</td>
<td>ns</td>
<td>nd</td>
<td>Webber and Nichols, 1995</td>
</tr>
<tr>
<td>Sediment, artificial islands, Beaufort Sea</td>
<td>ns</td>
<td>ns</td>
<td>ns</td>
<td>&lt;10 (dry weight)</td>
<td>Fowler and Hope, 1984</td>
</tr>
<tr>
<td>Poland, Holy Cross mountains, soil</td>
<td>July 3–6, 1996</td>
<td>8</td>
<td>1</td>
<td>nd</td>
<td>Migaszewski, 1999</td>
</tr>
<tr>
<td>Italy, Taranto, sediment contaminated by oil refinery or iron and steel factory wastes</td>
<td>ns</td>
<td>2</td>
<td>ns</td>
<td>nd</td>
<td>Cardelicchio et al., 1997</td>
</tr>
</tbody>
</table>

\(^1\) ns = not specified; nd = not detected.
\(^2\) The range of values is indicated in square brackets, if available (e.g., [minimum–maximum]).
\(^3\) <W is a qualifier, given to indicate that the sample may contain the analyte but the level would probably not exceed the laboratory method detection limit (MDL). W is approximately one-third to one-fifth of the MDL (OMEE, 1994).

DNOC was detected in 13% of Canadian municipal sludges sampled during the period 1980–1985 at concentrations ranging from 1200 to 1500 ng/g dry weight, with a median concentration of 1300 ng/g dry weight (Webber and Lesage, 1989). It was not detected (detection limit not stated) in sludge or sludge compost from various locations in Canada sampled in 1993–1994 (Webber and Nichols, 1995).

DNOC was not detected (method detection limit = 100 ng/g) in 101 samples of “rural parkland” soil or in 60 samples of “old urban parkland” soil in Ontario (OMEE, 1994). Similarly, DNOC was not detected (detection limit = 50 ng/g) in agricultural soil from various locations across Canada (Webber, 1994).

**Biota**
DNOC was not detected in fish composites (detection limit not stated) from the United States (DeVault, 1985).

As indicated in the section on environmental fate and partitioning, DNOC has a relatively low bioaccumulation potential. However, as will be seen in the section on effects characterization, results of repeated oral dose toxicity studies indicate that mammals may be fairly sensitive to DNOC. Therefore, wildlife exposure to DNOC from food and water has been estimated.

An EEV for wildlife was estimated based on a calculation of the total daily intake of the substance by mink and otter. An energetics model based on the general exposure model for wildlife from the U.S. Environmental Protection Agency’s (EPA) Exposure Factors Handbook (U.S. EPA, 1993) was used.

\[
TDI = \left( \frac{C_i \cdot P_i}{GE_i \cdot AE_i} \right) \cdot Pt
\]

where:

- **TDI** = total daily intake (mg/kg-bw per day)
- **FMR** = normalized free metabolic rate of wildlife receptor of interest (250 kcal/kg-bw per day for mink and river otter)
- **C** \(_i\) = concentration of contaminant in the ith prey species (mg/kg-bw) (see below)
- **P** \(_i\) = proportion of the ith prey species in the diet (unitless) (default = 35% for mink; 100% for otter)
- **GE** \(_i\) = gross energy of the ith prey species (default = 850 kcal/kg-bw prey)
- **AE** \(_i\) = assimilation efficiency of the ith prey species by the wildlife receptor (default = 0.91)
- **Pt** = proportion of the time the receptor spends in the contaminated area (= 9% for mink and 0.06% for otter).

The model incorporated the metabolic rate of the wildlife receptors of interest (mink and otter), the proportion of food uptake by the receptors and the amount of time the animals spend in the contaminated area (St. Clair River), which is based on the typical habitat range of the wildlife receptors.

The concentration of the substance in a fish (C\(_i\)) must be estimated based on the highest EEV\(_{\text{water}}\) and a BAF. The BAF was estimated using the Modified Gobas Model (Gobas and Arnott, 2003). The BAF represents a benthic/pelagic food chain and estimates the accumulation from all sources in a mid-trophic-level fish that would typically be eaten by a mammalian piscivore.

\[
C_i = EEV_{\text{water}} \cdot BAF
\]
where:

\[ C_i = \text{concentration in a prey fish (mg/kg-bw)} \]
\[ \text{EEV}_{\text{water}} = \text{EEV calculated for surface water (mg/L) (see section on aquatic concentrations)} \]
\[ \text{BAF} = \text{bioaccumulation factor for substance (L/kg) (see section on environmental fate and partitioning).} \]

\[ C_i = 0.0014 \cdot 25 = 0.035 \]

The model estimated EEVs of 0.0004 mg/kg-bw per day and 0.000 007 mg/kg-bw per day for mink and otter, respectively.

**Effects Characterization**

**Biotic Effects**

Key studies of the toxicity of DNOC to organisms in different environmental media are presented in Tables 6 to 9. Studies primarily on the acute toxicity of DNOC to microorganisms, aquatic invertebrates, insects, terrestrial invertebrates and vertebrates were located in the literature. No acute or chronic marine toxicity data were identified.

Table 6: Toxicity of DNOC to aquatic organisms

<table>
<thead>
<tr>
<th>Organism</th>
<th>Endpoint‡</th>
<th>Concentration (mg/L)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Microorganisms</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bacterium</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Pseudomonas putida</em></td>
<td>Toxic threshold, 16-hour EC₃ (growth)</td>
<td>16</td>
<td>Bringmann and Kühn, 1980</td>
</tr>
<tr>
<td>Cyanobacterium</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Microcystis aeruginosa</em></td>
<td>Toxic threshold, 72-hour EC₃ (growth)</td>
<td>0.15</td>
<td>Bringmann and Kühn, 1978</td>
</tr>
<tr>
<td>Green alga</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Scenedesmus quadricauda</em></td>
<td>Toxic threshold, 7-day EC₅ (growth)</td>
<td>13</td>
<td>Bringmann and Kühn, 1980</td>
</tr>
<tr>
<td>Green alga</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Scenedesmus subspicatus</em></td>
<td>96-hour EC₅ₒ (biomass)</td>
<td>6</td>
<td>Sewell <em>et al.</em>, 1995a</td>
</tr>
<tr>
<td></td>
<td>48-hour EC₅ₒ (growth rate)</td>
<td>12</td>
<td>Sewell <em>et al.</em>, 1995a</td>
</tr>
<tr>
<td>Protozoan</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Entosiphon sulcatum</em></td>
<td>Toxic threshold, 72-hour EC₃ (growth)</td>
<td>5.4</td>
<td>Bringmann and Kühn, 1980</td>
</tr>
<tr>
<td>Protozoan</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Chilomonas paramecium</em></td>
<td>Toxic threshold, 72-hour EC₃ (growth)</td>
<td>5.4</td>
<td>Bringmann and Kühn, 1981</td>
</tr>
<tr>
<td>Protozoan</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Uronaemia parduezi</em></td>
<td>Toxic threshold, 72-hour EC₅ (growth)</td>
<td>0.012</td>
<td>Bringmann and Kühn, 1981</td>
</tr>
<tr>
<td><strong>Aquatic plants</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Lemna minor</em></td>
<td>Specific growth rate, 7-day exposure</td>
<td>0.32</td>
<td>Sloof and Canton, 1983</td>
</tr>
</tbody>
</table>
### Table 7: Acute toxicity of DNOC to terrestrial plants

<table>
<thead>
<tr>
<th>Organism</th>
<th>Endpoint</th>
<th>Concentration (mg/L)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tobacco Nicotiana sylvestris</td>
<td>ED\textsubscript{50}</td>
<td>0.466</td>
<td>Strube et al., 1991</td>
</tr>
</tbody>
</table>

\textsuperscript{1} ED\textsubscript{50} = median effective dose.

### Table 8: Acute toxicity of DNOC to terrestrial invertebrates

<table>
<thead>
<tr>
<th>Organism</th>
<th>Endpoint</th>
<th>Concentration</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Earthworm Eisenia fetida</td>
<td>7-day LC\textsubscript{50}</td>
<td>17 mg DNOC/kg of soil</td>
<td>van der Hoeven, 1992</td>
</tr>
<tr>
<td></td>
<td>14-day LC\textsubscript{50}</td>
<td>15 mg DNOC/kg of soil</td>
<td>van der Hoeven, 1992</td>
</tr>
<tr>
<td></td>
<td>14-day NOEC</td>
<td>10 mg DNOC/kg of soil</td>
<td>van der Hoeven, 1992</td>
</tr>
</tbody>
</table>

\textsuperscript{1} EC = effective concentration; LC\textsubscript{50} = median lethal concentration; NOEC = No-Observed-Effect Concentration.
Table 9: Toxicity of DNOC to terrestrial vertebrates

<table>
<thead>
<tr>
<th>Organism</th>
<th>Endpoint</th>
<th>Concentration (mg/kg-bw)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Japanese quail</td>
<td>24-hour LD$_{50}$</td>
<td>14.8 (95% CI 13–17)</td>
<td>Dickhaus and Heisler, 1980</td>
</tr>
<tr>
<td>Coturnix japonica</td>
<td>8-day LC$_{50}$</td>
<td>106</td>
<td>Til and Kengen, 1980</td>
</tr>
<tr>
<td>Pheasants</td>
<td>LD$_{50}$</td>
<td>8.4</td>
<td>Janda, 1970</td>
</tr>
<tr>
<td>Partridges</td>
<td>LD$_{50}$</td>
<td>8.3</td>
<td>Janda, 1970</td>
</tr>
<tr>
<td>Rat</td>
<td>90-day LOEL</td>
<td>2.5 (per day)</td>
<td>Den Tonkelaar et al., 1983</td>
</tr>
</tbody>
</table>

$^1$ CI = confidence interval.

The most sensitive aquatic vertebrates reported in the literature are rainbow trout (Mayer and Ellersieck, 1986; Sewell et al., 1995c). The authors reported LC$_{50}$ values of 0.066 and 0.45 mg/L, respectively. The 96-hour LC$_{50}$ study reported by Sewell et al. (1995c) is an unpublished study; however, it was cited in a peer-reviewed report (IPCS, 2000). Atlantic salmon and bluegill are also sensitive, with 96-hour LC$_{50}$ values of 0.20 mg/L and 0.23 mg/L, respectively (Zitko et al., 1976; Buccafusco et al., 1981).

The effect of DNOC on terrestrial vertebrates (mink and otter) (Critical Toxicity Value [CTV] for wildlife) was calculated using the repeated mammalian (rat) oral dose toxicity data provided for the substance (2.5 mg/kg-bw per day for a 90-day rat dietary exposure study, Lowest-Observed-Effect Level [LOEL]) (Den Tonkelaar et al., 1983). The CTV$_{wildlife}$ is calculated by taking the chronic value (geometric mean of the No-Observed-Effect Level [NOEL] and LOEL) from the rat study and correcting it for body weight of a predictive sentinel species (Sample et al., 1996). In this case, the predictive sentinel species are the piscivorous mammals mink and river otter.

The CTV$_{wildlife}$ is thus calculated as:

$$CTV_{wildlife} = ChV_{ts} \cdot \left(\frac{BW_{ts}}{BW_{pss}}\right)$$

where:

- $ChV_{ts}$ = chronic value for test species (geometric mean of LOEL [2.5 mg/kg-bw per day] and NOEL [0.25 mg/kg-bw per day] = 0.8 mg/kg-bw per day)
- $BW_{ts}$ = mean body weight of test species (0.35 kg)
- $BW_{pss}$ = body weight of predictive sentinel species (0.807 kg for mink; 6.01 kg for otter) (Martin, 2004).

Therefore, $CTV_{wildlife} = 0.8 \times (0.35/0.807) = 0.35$ for mink and $0.8 \times (0.35/6.01) = 0.047$ for otter.

The ENEV$_{wildlife}$ is calculated from the CTV$_{wildlife}$ as follows:
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\[
\text{ENEV}_{\text{wildlife}} = \text{CTV}_{\text{wildlife}} / \text{AF}
\]

where:

\[
\text{ENEV}_{\text{wildlife}} = \text{wildlife Estimated No-Effects Value (mg/kg-bw per day)}
\]
\[
\text{AF} = \text{application factor (interspecies variation, laboratory to field extrapolation)}
\]

Therefore, the ENEV_{\text{mink}} is 0.035 mg/kg-bw per day, and the ENEV_{\text{otter}} is 0.0047 mg/kg-bw per day.

Abiotic Effects

Stratospheric Ozone Depletion

The ozone-depleting potential (ODP) of a substance is defined as the ratio of calculated ozone column change for each unit mass of a gas emitted into the atmosphere relative to the depletion calculated for an equal mass of reference gas chlorofluorocarbon-11 (CFC-11), an ozone-depleting substance with an ODP of 1. The formula used to calculate ODP is applicable only for substances with chlorine or bromine atoms. As DNOC does not contain any chlorine or bromine atoms, it has an ODP of zero, and it is concluded that DNOC will not contribute to ozone depletion.

Ground-Level Ozone Formation

To estimate ground-level ozone formation, a Photochemical Ozone Creation Potential (POCP) index is used (Environment Canada, 1996). The POCP is a measure of the relative effectiveness for ozone formation of a unit mass of organic substance compared with that of an equivalent mass of ethene. By definition, ethene has a POCP value of 100.

The episodic ozone formation can be estimated from a reactivity scale based on the rate constant for the hydroxyl–hydrocarbon reaction and the molecular weight of the subject substance relative to those properties of ethene:

\[
\text{POCP} = \left( \frac{k_s}{M_s} \right) \left( \frac{M_{\text{ethene}}}{k_{\text{ethene}}} \right) \times 100
\]

where:

\[
k_s = \text{reaction rate constant at 298 K for the reaction with the hydroxyl radical for the substance (3.0 \times 10^{-11} \text{ cm}^3/\text{mol per second})}
\]
\[
k_{\text{ethene}} = \text{reaction rate constant at 298 K for the reaction with the hydroxyl radical (8.5 \times 10^{-12} \text{ cm}^3/\text{mol per second})}
\]
\[
M_s = \text{molecular mass of the substance (198.13 g/mol for DNOC)}
\]
\[
M_{\text{ethene}} = \text{molecular mass of ethene (28 g/mol)}.
\]
The POCP value for DNOC is 50.

Although it is recognized that DNOC may have the potential to contribute to ground-level ozone formation, the relative contribution is dependent on the concentration as well as on the reactivity of DNOC relative to other volatile organic compounds (VOCs) in an area. Nevertheless, DNOC is a reactive VOC. VOCs “that participate in atmospheric photochemical reactions” were added as a class to Schedule 1 of CEPA 1999 (List of Toxic Substances) in July 2003 and are subject to a range of risk management activities focusing on total VOCs, rather than on individual compounds. Since VOCs that participate in atmospheric photochemical reactions already appear on Schedule 1, a given existing substance’s ability to participate in photochemical reactions is not used as a basis for concluding that it is toxic under Section 64 of CEPA 1999.

It is further worthy of note that releases of DNOC from non-pesticidal uses are believed to be largely to aquatic systems, with little partitioning to air. The extent and mechanism of possible DNOC formation in the atmosphere from precursor species are not yet well understood.

Global Warming Potential

Global Warming Potential (GWP) is defined as the ratio of calculated warming for each unit mass of a gas emitted into the atmosphere relative to the calculated warming for a reference gas, CFC-11. The GWP for DNOC was estimated to be $7.2 \times 10^{-12}$, and therefore DNOC is not expected to contribute significantly to climate change.

Risk Characterization

As part of risk characterization, one line of evidence includes consideration of risk quotients to identify potential for ecological effects. Other factors that affect current or potential risks, such as persistence, bioaccumulation and trends in ambient concentrations, are also considered.

Risk Quotient Analysis

Critical exposure and effects results and risk quotients are summarized in Table 10 and described in more detail below.

Table 10: Summary of data used in risk quotient (RQ) analysis of DNOC

<table>
<thead>
<tr>
<th>Scenario</th>
<th>EEV</th>
<th>CTV</th>
<th>AF</th>
<th>ENEV</th>
<th>RQ (EEV/ENEV)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pelagic organisms</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Industrial release; rainbow trout</td>
<td>0.0014 mg/L</td>
<td>0.26 mg/L</td>
<td>100</td>
<td>0.0026 mg/L</td>
<td>0.54</td>
</tr>
</tbody>
</table>
Scenario | EEV | CTV | AF<sup>1</sup> | ENEV | RQ (EEV/ENEV)
---|---|---|---|---|---
Rainfall; rainbow trout | 0.0025 mg/L | 0.26 mg/L | 10 | 0.026 mg/L | 0.096

Soil organisms

Earthworm | 0.1 mg/kg | 15 mg/kg dry weight | 100 | 0.15 mg/kg dry weight | 0.67

Wildlife consumers

Mink | 0.0004 mg/kg-bw per day | 0.35 | 10 | 0.035 mg/kg-bw per day | 0.011
River otter | 0.000007 mg/kg-bw per day | 0.047 | 10 | 0.0047 mg/kg-bw per day | 0.0015

<sup>1</sup> AF = application factor.

**Pelagic Organisms**

For pelagic organisms, a risk quotient was developed using the average 96-hour LC<sub>50</sub> values of rainbow trout reported by Mayer and Ellersieck (1986) (0.066 mg/L) and Sewell et al. (1995c) (0.45 mg/L). The average of the two studies, which is the CTV, is 0.26 mg/L.

For the *industrial release scenario*, if STP treatment is considered (27% removal efficiency), the EEV will be 0.0014 mg/L. Using an application factor of 100 on the CTV to account for acute to chronic extrapolation and intra- and interspecies variations, differently sensitive biological endpoints and laboratory to field extrapolations, the ENEV is calculated to be 0.0026 mg/L.

The risk quotient is therefore calculated as:

\[
\frac{\text{EEV}}{\text{ENEV}} = \frac{0.0014 \text{ mg/L}}{0.0026 \text{ mg/L}} = 0.54
\]

Even with STP removal considered, this represents a conservative scenario due largely to the very high quantity of DNOC assumed to be used by a single facility.

The maximum EEV under the defined *rainfall scenario* was determined to be 0.0025 mg/L with no STP treatment due to the assumption of a heavy rainfall. As rainfall represents an acute exposure scenario, the application factor does not need to account for acute to chronic extrapolation. Therefore, using an application factor of 10 and the same CTV of 0.26 mg/L for rainbow trout, an ENEV of 0.026 mg/L is calculated. The risk quotient is therefore:

\[
\frac{\text{EEV}}{\text{ENEV}} = \frac{0.0025 \text{ mg/L}}{0.026 \text{ mg/L}} = 0.096
\]

**Soil Organisms**
There are no quantified amounts of DNOC concentrations in Canadian soils. OMEE (1994) did not detect DNOC in 161 soil samples collected from soils in Ontario. The method detection limit of 0.1 mg/kg (100 ng/g) will be used as a surrogate for the level of DNOC in Canadian soil and is selected as the EEV.

One study was located in the literature on the effects of DNOC on terrestrial organisms. The LC$_{50}$ from a 14-day acute toxicity study on the earthworm is 15 mg/kg of soil. This value is selected as the CTV for exposures of soil organisms to DNOC. Dividing the value by a factor of 100 to account for extrapolation from laboratory to field conditions, acute to chronic ratio and interspecies and intraspecies variations in sensitivity gives an ENEV of 0.15 mg/kg.

The risk quotient for soil organisms is therefore:

\[
\frac{\text{EEV}}{\text{ENEV}} = \frac{0.1 \text{ mg/kg}}{0.15 \text{ mg/kg}} = 0.67
\]

**Aquatic Wildlife**

The EEVs for the mink and river otter were estimated to be 0.0004 mg/kg-bw per day and 0.000 007 mg/kg-bw per day, respectively. The ENEV for the mink was estimated to be 0.035 mg/kg-bw per day, and the ENEV for the river otter was calculated to be 0.0047 mg/kg-bw per day.

The risk quotients for aquatic wildlife are thus calculated to be:

\[
\frac{\text{EEV}_{\text{mink}}}{\text{ENEV}_{\text{mink}}} = \frac{0.0004 \text{ mg/kg-bw per day}}{0.035 \text{ mg/kg-bw per day}} = 0.011
\]

\[
\frac{\text{EEV}_{\text{otter}}}{\text{ENEV}_{\text{otter}}} = \frac{0.000 007 \text{ mg/kg-bw per day}}{0.0047 \text{ mg/kg-bw per day}} = 0.0015
\]

**Benthic Organisms**

No monitoring data for DNOC in sediments in Canada were identified. IPCS (2000) has stated that if released to water, DNOC is only moderately adsorbed onto aquatic sediments. Level III multimedia fate simulation estimated that only about 1% of DNOC is expected to partition to sediments. It is therefore believed that there will be minimal exposure of benthic organisms to DNOC.

*Weight of Evidence Analysis*

The risk quotient analyses for pelagic and soil organisms and wildlife have shown that it is unlikely that organisms are currently exposed to concentrations of DNOC above known effect.
thresholds. This conclusion is based on current import levels, the locations where DNOC is being used industrially and the current state of knowledge of its atmospheric chemistry.

A conservative scenario based on concentrations of DNOC in precipitation that could be expected to enter Canadian receiving water indicated that the potential for risk to aquatic organisms from this source is low.

In addition, modelling estimates of industrial releases to the St. Clair River indicate that DNOC is not likely to cause adverse effects on pelagic or benthic organisms. This is based on a conservative release scenario developed for the one company importing the substance that currently reports releases of DNOC to the NPRI, although the company has not reported releases to water.

Although sorption is low at environmentally relevant pHs, little leaching to groundwater has been found, likely due to biodegradation.

Potential sources of release of DNOC to the environment are to air and water. Based on its properties, DNOC is persistent in air but not bioaccumulative. Long-range transport modelling estimates that it will be transported over moderate distances, and a decreasing concentration with increasing latitude is expected.

Proposed Conclusions for the Environment

DNOC meets the criteria for categorization owing to its persistence in air and inherent toxicity to aquatic organisms. DNOC is not bioaccumulative.

When comparing effect thresholds with conservatively estimated exposure values for different media, resultant risk quotients are less than unity. DNOC is believed to be used by a limited range of industrial facilities in Canada.

Based on available data and weight of evidence, it is proposed that DNOC is not entering the environment in a quantity or concentration or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity or that constitute or may constitute a danger to the environment on which life depends.

Further Considerations

A release scenario using European monitoring data to determine the effects of concentrations of DNOC in precipitation released to Canadian rivers indicated that there is a low likelihood of risk. This is believed to be a conservative scenario; however, there is significant uncertainty. Studies are in progress in Europe to improve understanding of the extent and mechanisms of formation of DNOC in the atmosphere. As the science evolves or Canadian monitoring data become available, this information could be used in setting priorities on the need for further assessment.
Uncertainties

There are uncertainties associated with development of the ENEVs used in this assessment. However, a moderate number of empirical studies from different sources were identified, which increases confidence in the values. Application factors of 10–100 were used to account for information gaps relating to chronic toxicity, effects in the field and effects on potentially more sensitive species. Effects data were not identified for marine species.

Very few Canadian monitoring data are available for DNOC, and those that were identified were fairly old. To both support the limited amount of empirical data and provide greater insight into the potential range of levels of DNOC in the environment, releases were estimated and fate and exposure were modelled. Entry of DNOC to the environment from two sources was considered — industrial releases and precipitation containing DNOC scavenged from the atmosphere. To address the significant uncertainty in these estimations, conservative assumptions were used to ensure that errors would be protective of the environment.

Although there have not been reports of direct releases of DNOC to water from industrial facilities, a conservative scenario was developed to estimate possible releases from an industrial source. This conservatively assumed an upper-limit estimate of the quantity of DNOC potentially used by a single facility; a slightly conservative estimate of the fraction of substance typically released due to handling practices for a substance used in bulk; and a low-percentile estimate of river flow for the receiving water body used in the scenario. Flow characteristics of the St. Clair River were used in the exposure scenario, as it is believed that the only facility in Canada currently using substantive quantities of DNOC is located near this water body. This river is extremely fast flowing and consequently disperses effluents very rapidly. Were there to be facilities having substantive releases to smaller water bodies, then the assumptions used in this scenario may not be sufficiently protective. However, it is believed that there are currently no other large users of DNOC in Canada.

Estimation of possible exposure from atmospherically generated DNOC in precipitation conservatively assumed that the concentration in the atmosphere in Canada would be similar to that in more heavily populated regions of Europe; that the rainfall event would be particularly heavy; that a high percentage of precipitation from a census subdivision would be released to the receiving river body through a single discharge point; and that there would be no removal of DNOC by the municipal STP. In particular, the assumption that atmospheric concentrations in Canada would be the same as average to high concentrations in Germany, which is much more heavily populated and industrialized, is uncertain. While it is believed that use of monitoring data from Germany in the scenario is conservative, the origins of atmospherically generated DNOC are at present not well understood, and no Canadian atmospheric monitoring data were identified for comparison.
References


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