Risk analysis of dissolved organic matter-mediated ultraviolet B exposure in Canadian inland waters


Abstract: With depleted ozone levels and the possibility that climate change might lower dissolved organic matter (DOM) concentrations, the risk of exposure of fresh waters in Canada to ultraviolet B (UV-B) was evaluated. First, the distribution of a UV-B-sensitive zooplankton genus, *Daphnia*, was examined as a function of DOM concentration and maximum depth (*Z*<sub>max</sub>) in 258 systems. Distribution was not restricted by UV-B, although very clear, shallow systems are underrepresented. Secondly, the depth at which 1% of surface radiance at 320 nm occurs (*Z*<sub>320,1%</sub>) was compared with *Z*<sub>max</sub> in over 1000 aquatic systems in 15 ecozones to determine the proportions of optically clear systems (*Z*<sub>max</sub> ≤ *Z*<sub>320,1%</sub>) and systems that may become clear (i.e., are “at risk”) should DOM decrease by 50%. South of the treeline, <6% of systems were clear, with the exception of two ecozones with 10%–20%. The proportion of systems at risk was 0% in most regions, with 5%–9% in four regions. DOM levels appear adequate to prevent extirpation of sensitive taxa like *Daphnia* through direct exposure to UV-B in most regions south of the treeline. However, optically clear and at-risk ponds were much more common in the three Arctic ecozones, especially the Northern Arctic and Arctic Cordillera.

Résumé : Nous avons évalué le risque de l’exposition des milieux d’eau douce canadiens aux ultraviolets (UV-B), compte tenu des réductions de la couche d’ozone et de la possibilité que les changements climatiques réduisent les concentrations de matière organique dissoute (DOM). Nous avons d’abord examiné la répartition de *Daphnia*, un genre d’animal zooplanctonique sensible aux UV-B, en fonction de DOM et de la profondeur maximale (*Z*<sub>max</sub>) dans 258 systèmes. La répartition n’est pas restreinte par l’UV-B, bien que les milieux peu profonds à eau très claire soient sous-


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Introduction

Many recent studies have investigated the effects of ultraviolet radiation (UVR; 290–400 nm) on aquatic systems. These have been stimulated in part by increases in summer solar radiation in the UV-B region (290–320 nm) associated with depletion of stratospheric ozone (e.g., see reviews in Hader 1997; de Mora et al. 2000; Helbling and Zagarase 2003). However, estimated increases in erythemal irradiance (a weighted UV-B function) since the early 1980s have only been about 6%–14% in Canada, depending on the region (Kerr et al. 2002). This suggests that ozone depletion is unlikely by itself to result in widespread large-scale extirpation in aquatic systems unless key aquatic organisms are extremely sensitive to UV-B. A complicating factor, however, is that decreases in chromophoric dissolved organic matter (DOM), which strongly absorbs radiation in the UVR region (Scully and Lean 1994), could lead to significant increases in UV-B exposure. Two regional assessments of the risk posed by UV-B to amphibians in ponds in Minnesota and the US Northwest estimated that about 85% of ponds had adequate protection (Diamond et al. 2002; Palen et al. 2002; Peterson et al. 2002). There is, however, no clear picture of the extent, if any, to which the millions of lakes and ponds in Canada are at risk from UV-B exposure, a term defined operationally below.

Exposure to UV-B varies widely among aquatic systems, mainly because of variation in levels of DOM and its optical properties. It can range from less than 1 mg C·L–1 in exceptionally clear waters to more than 300 mg C·L–1 in some saline lakes in the Prairies (P.J. Curtis, unpublished data) (analytical methods for total organic carbon, i.e., high-temperature combustion and UV/persulphate methods, do not discriminate between chromophoric and non-chromophoric DOM). There is some concern that very large decreases in chromophoric DOM might occur as a result of reduced run-off associated with climate change (Leavitt et al. 1997; Schindler et al. 1997; Pienitz and Vincent 2000), producing large increases in underwater irradiance. In fact, increased UV-B exposure from reductions in chromophoric DOM might pose more of a risk to aquatic communities than ozone depletion (Williamson et al. 1996; Pienitz and Vincent 2000; Leavitt et al. 2003). On the other hand, counteractive processes associated with climate warming may affect northern lakes and raise their DOM concentrations (Clair et al. 1999). For example, with climate warming and an expected increase in vegetation at northern latitudes, e.g., the northern shift of the treeline, increased loads of DOM to lakes and ponds in these areas could occur (Pienitz and Vincent 2000). Attenuation of UVR by chromophoric DOM is discussed in more detail below.

Evaluating the extent to which aquatic environments might be “at risk” from UV-B exposure is warranted with the detection of harmful effects of UV-B on some taxa, the restoration of pre-industrial ozone levels still some years away, and the possibility that in chromophoric DOM concentrations might decrease significantly. We explore this topic for Canadian freshwater systems by first examining the eastern North American distribution of a crustacean zooplankton genus, Daphnia, which experimental evidence indicates may be a sensitive indicator of UV-B exposure. The first objective is to determine whether UV-B exposure restricts its distribution. Second, the maximum depth of UV-B transmission (defined here as the depth at which 1% of surface irradiance at 320 nm occurs (Z<sub>320,1%</sub>) is compared with maximum depths (Z<sub>max</sub>) for aquatic systems across Canada to determine the extent to which systems are currently optically clear or may become so (at risk) should they lose 50% of their DOM. Discussions of recent variation in incident levels of UV-B and the levels predicted over the short term (years) and long term (decades) across Canada are beyond the scope of this paper.

Methods

UV-B-sensitive indicator organisms

Rather than use the more subtle ecological changes that might occur, such as changes in life-history patterns or energy allocation to UV-protection mechanisms at the expense of growth, we chose instead to analyze the presence/absence of sensitive organisms. Research has shown that some stressors, such as acid deposition and Great Lakes contamination, have led to local extirpation in advanced stages of the problem, thus presence/absence data can be a practical tool for identifying severe damage. Moreover, because presence/absence data are more widely available than other ecological measurements, a larger number of systems can be assessed.

First, the literature was reviewed to identify sensitive indicator organisms. We then analyzed the presence/absence of the indicator taxa with respect to two relevant chemical and physical variables that affect the size of UV-B-free refuges, DOM concentration and Z<sub>max</sub>, using several detailed synoptic surveys. The distribution of populations was compared with Z<sub>320,1%</sub>.
Several criteria were applied to the screening process. (i) Indicator organisms must be widely distributed among aquatic systems. (ii) Survival of indicator organisms must be impaired by UV-B and they must spend at least part of their life cycle in epilimnetic waters during summer daylight hours. For organisms to be useful as a biological indicator in this study, experimental evidence of sensitivity to UV-B must be coupled with evidence of changes in abundance. This is a practical consideration because only abundance data are likely to be widely available. (iii) Responses of indicator populations to UV-B change must be relatively rapid (time scale of weeks or months). (iv) Ideally, indicator organisms should play an important ecological role. (v) Chemical and physical indicators of UV-B exposure must be widely reported to ensure that an adequate number of lakes and ponds in various regions can be assessed.

Collectively, these criteria exclude process measurements such as primary-production and DOM-degradation rates, planktonic taxa at the species level (because of wide geographic variation and influence of non-UV-B-related factors), and long-lived taxa even where widely distributed (e.g., fish). Variation in nutrient loading with DOM loading (Dillon and Molot 1997) also precludes using a standing-crop indicator such as chlorophyll a. Certain experimental designs also preclude the use of their results in the search for indicators. For example, Bothwell et al. (1993) pointed out that “conclusions about the long-term ecological impacts of changes in UVB cannot be accurately forecast from short-term incubation studies”. Furthermore, results of studies that remove UV-B through selective use of filters do not necessarily imply that a relationship between a response variable and UV-B intensity can be extrapolated to an increase in UV-B. Organisms must be exposed to UV-B levels above their “ambient” levels, perhaps by incubation in shallower waters or in waters with lower DOM concentration or by supplementation with an artificial source.

Distribution of systems at risk in Canada

DOM and $Z_{\text{max}}$ data for aquatic systems across Canada were gathered from a variety of published and unpublished sources (Table 1). The surveys include mostly discrete (one time) measurements, although some means are included (e.g., the Nova Scotia and Newfoundland surveys provided long-term means), as long-term data are usually collected only from a small number of research sites. $Z_{\text{max}}$ was not available from the Ontario Ministry of the Environment (OMOE) and Canadian Wildlife Service (CWS) Ontario surveys and was available for only some of the systems in the Arctic surveys. DOM was measured with either UV/persulfate digestion or high-temperature combustion. $Z_{320,1\%}$ was compared with $Z_{\text{max}}$ to determine the extent to which systems are currently optically clear or may become so (at risk) should they lose 50% of their DOM. $Z_{120,1\%}$ was chosen because deeper waters are virtually UV-B-free (see Attenuation of UVR penetration by chromophoric DOM below). Relative proportions of optically clear systems and

<table>
<thead>
<tr>
<th>Ecozone</th>
<th>Province</th>
<th>Source*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Atlantic Maritime</td>
<td>Nova Scotia</td>
<td>Clair et al. 2001</td>
</tr>
<tr>
<td></td>
<td>New Brunswick</td>
<td>T.A. Clair</td>
</tr>
<tr>
<td>Mixedwood Plains</td>
<td>Quebec (Townships)</td>
<td>Y.T. Prairie</td>
</tr>
<tr>
<td>Boreal Shield</td>
<td>Ontario</td>
<td>P.J. Dillon</td>
</tr>
<tr>
<td></td>
<td>Ontario</td>
<td>D.K. McNicol</td>
</tr>
<tr>
<td></td>
<td>Quebec (Abitibi)</td>
<td>Y.T. Prairie</td>
</tr>
<tr>
<td></td>
<td>Newfoundland</td>
<td>Clair et al. 2001</td>
</tr>
<tr>
<td>Prairies</td>
<td>Saskatchewan</td>
<td>Arts et al. 2000</td>
</tr>
<tr>
<td>Saline</td>
<td>Saskatchewan</td>
<td>Arts et al. 2000</td>
</tr>
<tr>
<td>Non-saline</td>
<td>Alberta</td>
<td>P.J. Curtis</td>
</tr>
<tr>
<td>Taiga Plains</td>
<td>Northwest Territories</td>
<td>Pienitz et al. 1997a</td>
</tr>
<tr>
<td>Hudson Plains</td>
<td>Manitoba</td>
<td>Macrae 1998</td>
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<td>Pienitz et al. 1997b</td>
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<td></td>
<td>Northwest Territories</td>
<td>Rühland et al. 2003</td>
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<tr>
<td>Montane Cordillera</td>
<td>British Columbia</td>
<td>P.R. Leavitt</td>
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<td></td>
<td>British Columbia</td>
<td>R.N. Nordin</td>
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<tr>
<td></td>
<td>British Columbia</td>
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<td>Boreal Cordillera</td>
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<tr>
<td>Southern Arctic</td>
<td>Yukon, Northwest Territories</td>
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</tr>
<tr>
<td>Northern Arctic</td>
<td>Nunavut, Northwest Territories</td>
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</tr>
<tr>
<td></td>
<td>Nunavut</td>
<td>J.P. Smol and M.S.V. Douglas</td>
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*Names of authors of this study are provided if the reference is not cited.
The vast majority of the sampled sites (over 7200) were from two surveys of the Boreal Shield ecozone in Ontario (some sites were sampled in both surveys), with 791 sites sampled in 14 other ecozones across Canada. There are insufficient sampled sites from the Taiga Plains and Taiga Cordillera, with five and six sites, respectively. Hence, results for these ecozones should be treated with caution until additional data are available. Finally, some of the limnological surveys south of the tree line may have ignored shallow, small ponds (<1 m maximum depth), which could bias the results for regions where they are numerous. The exceptions to this are the Prairies survey of lakes and ponds in Saskatchewan, which includes 18 ponds (Arts et al. 2000), the CWS survey of lakes and ponds in the Boreal Shield in Ontario, which includes an unknown number of ponds (D. McNicol, unpublished data), and the Atlantic Maritime survey in Nova Scotia (six ponds) (Clair et al. 2001). By definition, a greater proportion of ponds than lakes should be at risk because of the shallow nature of the former.

Results

Attenuation of UVR Penetration by chromophoric DOM

The attenuation of incident irradiance of a specific wavelength is described by exponential decay with depth:

\[ \frac{dE_{\lambda,Z}}{dZ} = -K_{\lambda} \]

\[ E_{\lambda,Z} = E_{\lambda,0} \exp(-K_{\lambda}Z) \]

where \( E_{\lambda,Z} \) is the solar-radiation intensity at wavelength \( \lambda \) at depth \( Z \) below the surface, and \( K_{\lambda} \) is the attenuation coefficient for wavelength \( \lambda \). The depth at which 1% of surface irradiance for a given wavelength occurs (e.g., 99% attenuation) is given by

\[ Z_{1%} = -\frac{\ln(0.01)}{K_{\lambda}} = 4.61/K_{\lambda} \]

\( K_{\lambda} \) is a function of photon absorption by water and its constituents and photon scattering by particles (Vincent et al. 2001). UV-B is strongly absorbed by chromophoric DOM relative to light in the visible region (400–700 nm) (Crump et al. 1999; Markager and Vincent 2000).

The effect of chromophoric DOM and particulate matter on attenuation of solar radiation is illustrated with stepwise regressions developed using \( K_{\lambda} \) and DOM and particulate organic matter (POM) data for 14 lakes along a latitudinal gradient between 41 and 51°N (from Scully and Lean 1994). Attenuation coefficients for six wavelengths are reported between 300 and 400 nm. The attenuation coefficient at 550 nm was estimated by extrapolating the linear relationship between \( \lambda \) and \( \ln K_{\lambda} \) for each lake. Regressions for attenuation coefficients at 320, 340, 360, 380, and 550 nm as functions of DOM and POM concentrations are shown in Table 2.

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<thead>
<tr>
<th>Eq, ( \lambda ) (nm)</th>
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<td>4 550</td>
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Table 2. Regression equations for \( \ln(K_{\lambda}) \) as functions of \( \ln(\text{DOM}) \) and \( \ln(\text{POM}) \) between 320 and 550 nm.

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The crustacean genus *Daphnia* appear sensitive to UV-B (Zagarese et al. 1994; Zellmer 1998; Grad et al. 2001) and UV-A (320–400 nm) (Rautio and Korhola 2002a), showing decreased survival when exposed. *Daphnia* was the most UV-B-sensitive of several zooplankton taxa tested (Cywinska et al. 2000). Significant in-situ mortality was observed after 2 days of exposure at and above 73% of surface radiation at 320 nm but not at 63% (Zagarese et al. 1994). Survival of *Daphnia magna* was impaired after 4 and 8 h of exposure to UVR at DOM concentrations ≤1 mg C·L⁻¹, while 5 mg C·L⁻¹ provided complete protection from 4 but not 8 h of exposure (Hessen and Alstad Rukke 2000).

There are several mitigating factors that may affect tolerance to UV-B, such as calcium level, pigment production, temperature, and behaviour. Hessen and Alstad Rukke (2000) found that low calcium concentrations can increase *Daphnia* sensitivity to UV-B. Declining calcium levels in some areas of eastern North America related to atmospheric deposition (Keller et al. 2001) may, therefore, complicate risk assessment. Some *Daphnia* produce melanin, a photoprotective pigment that absorbs UV-B (Hessen et al. 2002). Melanin concentration in subarctic lakes was a function of DOM concentration, increasing sharply below 3 mg C·L⁻¹, while 5 mg C·L⁻¹ provided complete protection from 4 but not 8 h of exposure (Hessen and Alstad Rukke 2001).

Table 3. Summary of lakes and ponds from four surveys with maximum depth < 17 m and DOM < 5 mg C·L⁻¹ used to analyze *Daphnia* distributions.

<table>
<thead>
<tr>
<th>Source</th>
<th>No. of lakes and ponds</th>
<th>Region</th>
</tr>
</thead>
<tbody>
<tr>
<td>Norman Yan, Ontario Ministry of the Environment</td>
<td>18</td>
<td>Muskoka–Haliburton, Ontario</td>
</tr>
<tr>
<td>Bill Keller, Laurentian University</td>
<td>151</td>
<td>Algoma and Sudbury, Ontario</td>
</tr>
<tr>
<td>Don McNicol, Canadian Wildlife Service</td>
<td>8</td>
<td>Sudbury, Ontario</td>
</tr>
<tr>
<td>US Environmental Protection Agency Environmental Monitoring and Assessment Program, Eastern Lake Survey</td>
<td>81</td>
<td>Northeastern USA</td>
</tr>
</tbody>
</table>

*DOM measured by acidifying samples, flushing with N₂ to remove inorganic C, then oxidizing with a UV digester in acid persulfate media. CO₂ is colorimetrically detected. Analytical method for DOM described in Hillman et al. (1986). Data are available on-line at www.epa.gov/docs/emap/html/dataI/surfwatr/data/napap/els.html.

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greater than 1% of surface irradiance at 320 nm. Because shorter wavelengths have higher extinction coefficients, relative irradiance levels for wavelengths < 320 nm are less than 1% at \( Z_{320,1\%} \). Hence, waters deeper than \( Z_{320,1\%} \) are essentially UV-B-free. Also, \( Z_{320,1\%} \) is very similar to \( Z_{360,10\%} \), which means that waters above \( Z_{320,1\%} \) will receive significant levels of UV-A > 360 nm. Note that atmospherically acidified lakes may still be clearer than their historical values in spite of recovery (Gunn et al. 2001).

Next, we define systems at risk as those that will become optically clear should they lose 50% of their DOM. However, DOM levels vary with time, and thus there is the danger that organisms in systems with low DOM concentrations may be temporarily exposed to UV-B. For example, the chromophoric portion of DOM can vary diurnally (Gibson et al. 2001) and seasonally (Clair et al. 1996; Morris and Hargreaves 1997; Laurion et al. 2000). The most plausible explanation is that as DOM ages it becomes more photo-bleached. This is consistent with the decline in colour–DOM ratios observed between headwater streams and lake outflows (Molot and Dillon 1997).

Variation in allochthonous loading will affect DOM concentrations. Mean annual DOM varied <1 mg C·L⁻¹ in three central Ontario lakes (regional runoff 0.4–0.6 m·year⁻¹) over a 12-year period from 1980 to 1992: 1.7–1.9 mg C·L⁻¹ in Blue Chalk Lake, 3.9–4.6 mg C·L⁻¹ in Crosson Lake, and 4.7–5.6 mg C·L⁻¹ in Dickie Lake (P.J. Dillon, unpublished data). In any given year the maximum deviation from the long-term mean was between 7% and 11%. Thus, in a lake with a long-term mean DOM of 3.7 mg C·L⁻¹, the concentration may decline to 3.3 mg C·L⁻¹ in some years, and in a lake with a long-term mean DOM of 1.7 mg C·L⁻¹, it might decline to 1.6 mg C·L⁻¹ in some years. The DOM concentration varied more in other regions: in Lake 239 in northwestern Ontario (regional runoff 0.2 m·year⁻¹) (Schindler et al. 1997), DOM ranged from a high of about 8.6 mg C·L⁻¹ in 1982 to a low of about 5.2 mg C·L⁻¹ in 1984, a decline of 3.4 mg C·L⁻¹ in 2 years. In Kejimkujik Lake in southwestern Nova Scotia (regional runoff 1 m·year⁻¹), DOM ranged from 3.0–13.0 mg C·L⁻¹ with a mean of 6.9 mg C·L⁻¹ between 1983 and 1999, although 91% of the samples were between 3 and 8.3 mg C·L⁻¹ (Clair et al. 2001). Shallow systems can also exhibit large seasonal variation: in 11 of 13 wetland sites studied at St. Denis, Saskatchewan, DOM concentrations increased from spring to fall by an average of 64% (Waiser and Robarts 2004). Specific absorptivity in Prairie ponds decreases during the summer, even as DOM increases (Waiser and Robarts 2004).

In this study, a simplifying assumption is that temporal variation in chromophoric DOM concentration is relatively small in low-DOM systems (<5 mg C·L⁻¹), which are the primary systems of interest, while it is acknowledged that the risk analysis below might have to be revisited if this assumption proves to be dramatically violated. We also assume that regional variation in the chromophoric content of DOM is not significant in low-DOM systems. Data from Scully and Lean (1994) suggest that regional variation in chromophoric content is not large. However, other studies have shown that attenuation of UVR can vary regionally, in part because vegetation affects the chromophoric content of DOM. In the Tyrolian Alps in Europe, specific absorptivity at 320 nm was lowest among lakes with poor soil cover and highest among lakes with forest cover (Laurion et al. 2000). Regional differences in the chromophoric content of DOM are suggested by differences in specific UV-B attenuation (broadband UV-B attenuation coefficient \( K_d/\text{DOM} \)) between Prairie and non-Prairie freshwater lakes (Scully and Lean 1994; Arts et al. 2000). However, specific UV-B attenuation in non-Prairie lakes sampled by Scully and Lean (1994) is similar to that in Prairie ponds (Arts et al. 2000).
Saline systems are quite different from freshwater systems (Fig. 2) (Arts et al. 2000; Waiser and Robarts 2000). Most prairie saline systems have much lower specific attenuation than freshwater systems and high DOM concentrations (>20 mg C·L⁻¹) (Fig. 2). None of the 3 saline ponds and only 2 of the 17 saline lakes surveyed by Arts et al. (2000) in Saskatchewan had a $K_d$ value < 10 m⁻¹, which is the maximum value in the freshwater, low-DOM lakes (<5 mg C·L⁻¹) measured by Scully and Lean (1994). Although saline ponds have lower specific absorbivity and thus a higher proportion of non-chromophoric DOM (Arts et al. 2000; Waiser and Robarts 2000), their much higher DOM levels result in significant UV-B attenuation.

We operationally define systems at risk here as systems that are not now optically clear but in which loss of 50% of DOM would result in $Z_{320,10%} \geq Z_{\text{max}}$. We are now in a position to estimate the number of systems in ecozones across Canada that are naturally optically clear or at risk of becoming optically clear should DOM decrease.

Data were collected from all 15 ecozones across Canada (Table 4). The number of sites with complete data varied dramatically from 5 in the Taiga Plains ecozone to over 500 in the Boreal Shield ecozone. All ecozones are represented by at least 24 sites except the Taiga Cordillera (5) and Taiga Plains (6). Results from the latter two ecozones should therefore be treated with caution.

Median DOM concentrations varied dramatically among ecozones, ranging from 1.1 mg C·L⁻¹ on Victoria Island in the Northern Arctic to 14.7 mg C·L⁻¹ in the Boreal Cordillera, with the exception of the Prairies (Table 4). Median Prairie non-saline and saline concentrations were 39 and 59.8 mg C·L⁻¹ in Saskatchewan and 46 and 120 mg C·L⁻¹ in Alberta, respectively.

The proportion of systems currently deemed optically clear is quite low (<6%) in most regions south of the treeline, with 10%–20% in the Montane Cordillera (Kamloops, Prince George, and Continental Divide regions) and Pacific Maritime ecozones (Table 4). Optically clear ponds are more common in the Hudson Plains and Southern Arctic ecozones (0%–13%) and are the norm in the Northern Arctic and Northern Cordillera ecozones (31%–100%). Community structure in these clear systems may, perhaps, reflect the effects of UV-B exposure. If, however, a sensitive aquatic biota was able to tolerate, say, no more than 10% of surface radiation at 320 nm rather than 1%, then the proportion of optically clear systems would be smaller.

The proportion of systems deemed to be at risk was 0% in most regions south of the treeline and between 5% and 9% in the Boreal Shield (Ontario and Newfoundland), Atlantic Maritime (Nova Scotia), and Montane Cordillera ecozones along the Continental Divide (Table 4). The proportion of ponds at risk ranged from 0% to 28% in the Southern Arctic and from 11% to 62% in the Northern Arctic. No systems at risk occurred at Cape Herschel on Ellesmere Island in the Arctic Cordillera because all ponds were optically clear. These estimates are based on a worst case scenario in which it is assumed that there exist very sensitive organisms incapable of tolerating more than 1% of surface irradiance at 320 nm. If these sensitive organisms are able to tolerate up to, say, 10% of surface irradiance at 320 nm, then the proportion of systems at risk (i.e., systems without refuge below $Z_{320,10%}$) would be smaller.

There is a note of caution regarding wetlands: many of these shallow systems can lose considerable water volume and therefore depth as a result of strong evaporation. Consequently, deeper water refuges from harmful UV-B may disappear with relatively low DOM concentration, especially in very dry years.

DOM < 5 mg C·L⁻¹ in a large fraction of systems on the Boreal Shield in Ontario (53% and 39% in the OMOE and CWS surveys, respectively). Because depth data are not available, we take these estimates to represent the upper bounds for the proportion of systems at risk. The actual proportion is likely much less, as indicated by the small proportions of Boreal Shield systems at risk in the Algoma, Sudbury, and Muskoka–Haliburton regions of Ontario, the Abitibi region of Quebec, and Newfoundland (5%, 0%, and 6%, respectively).

Discussion and conclusions

This study suggests that DOM levels are adequate to prevent large-scale loss of *Daphnia* and perhaps other similarly sensitive taxa from direct exposure to UV-B at current incident levels in most regions below the treeline in Canada. In contrast, a large proportion of ponds in the Canadian Arctic are either optically clear or at risk. These conclusions are based on two approaches: (1) a comparison of $Z_{\text{max}}$ with $Z_{320,10%}$ in conjunction with the presence/absence of *Daphnia*, and (2) biogeochemical estimates of the numbers of “optically clear” and “at-risk” systems across Canada. Although we tentatively conclude that UV-B-induced extirpation of *Daphnia* and similarly sensitive taxa is probably not a major concern south of the treeline, even if a 50% loss of DOM should occur, generalizing this to conclude that loss of DOM will not affect species richness requires that (i) additional chemical and morphometric data be collected, especially from shallow ponds south of the treeline, and (ii) additional sensitive taxa be identified to which a
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cols can sometimes generate unrealistic conditions. Therefore, the percentage of wetlands in the US Pacific Northwest and northern Minnesota that provide adequate protection to developing amphibians is probably underestimated.

This study did not assess the extent to which DOM may decrease. Significant regional variation in response to climate change is predicted by the Canadian General Circulation Model, especially north of the treeline and on the Prairies (Canadian Centre for Climate Modelling and Analysis 2003). Hence, the probability of experiencing decreasing DOM and increasing exposure is not constant among ecozones. In general, DOM export from catchments to surface waters will decrease if there is a reduction in peatland area (Dillon and Molot 1997). This, in turn, may occur if there is a significant long-term reduction in runoff.

We express a word of caution at a time when the search for sentinel indicators of ecosystem health is in full swing: our study suggests that obtaining experimental evidence of sensitivity of organisms to a stressor should not lead us to automatically conclude that these organisms are effective in our study suggests that obtaining experimental evidence of a significant long-term reduction in runoff. (Dillon and Molot 1997). This, in turn, may occur if there is a significant long-term reduction in runoff.

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There are two issues worth mentioning. (1) This study defined risk incurred through increased UV-B exposure only in terms of loss of DOM. If risk is to be more fully defined by loss of DOM, then a suite of indicators should be expanded to include other impacts associated with loss of DOM, such as changes in heating and visible-light penetration to name just two. Loss of DOM would lead to a larger, clearer epilimnion and smaller hypolimnion in lakes deep enough to stratify (Gunn et al. 2001). An empirical formula recently developed to predict optimal lake trout (Salvelinus namaycush) habitat in Ontario (Dillon et al. 2003) predicts that a decline in DOM from 8 to 7 mg C·L−1 would result in the depth at which 10 °C occurs at the end of the summer (Z10 °C) becoming 0.2 m deeper but a decline from 3 to 2 mg C·L−1 would result in Z10 °C becoming 1.9 m deeper. As is the case with UV-B, systems with relatively low DOM are more at risk of heating from loss of DOM (Gunn et al. 2001), and as we have shown here, most ecozones outside the Prairies have large numbers of low-DOM (<3 mg C·L−1) systems. (2) Owing to variation in the proportions of chromophoric and non-chromophoric DOM, we recommend that future surveys also measure underwater radiation transmission in addition to DOM. Because direct measurement of in-situ irradiance requires expensive equipment not available in many laboratories, we recommend instead measuring absorbance of unfiltered surface samples throughout the UVR and visible regions (a path length > 1 cm will be needed for longer wavelengths and clear waters).

Although there are no guidelines for judging what percent loss is acceptable, current critical load targets for acid deposition in North America provide a model. Critical load targets are based on the desirability of maintaining 95% of lakes in each deposition region above the critical pH thresh-

old of 6; that is, 5% of lakes will not be protected (Jeffries 1997). Using this criterion, DOM levels in most regions south of the treeline appear to be adequate to protect aquatic systems from significant species losses, with the possible exception of 5%–9% of systems in Nova Scotia and Newfoundland and along the Continental Divide. However, DOM levels are probably not adequate to protect 95% of systems north of the treeline.

This should not be construed to suggest that UV research is no longer necessary. In fact, there is a consensus that less dramatic UVR-mediated impacts are quite important. For example, UV-A induces photorepair (Quesada et al. 1995) and may inhibit photosynthesis (Callieri et al. 2001). UV-A is also responsible for the majority of photooxidative loss of DOM (Molot and Dillon 1997b). Our understanding of these roles is only emerging, therefore process-based ecological research on both UV-B and UV-A should continue. Future research may shape our understanding of how aquatic organisms have evolved to cope with UVR and the extent to which subtler impacts occur.

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**References**


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