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The attenuation of ultraviolet radiation in high dissolved organic carbon waters of wetlands and lakes on the northern Great Plains

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Abstract

We used a scanning spectroradiometer to conduct underwater optical surveys of 44 waterbodies during the ice-free seasons of three consecutive years in wetlands and lakes in central Saskatchewan, Canada. The waterbodies ranged widely in dissolved organic carbon (DOC) concentration (4.1–156.2 mg L^{-1}) and conductivity (270–74,300 μohms cm^{-1}). Although penetration of UV radiation (UV-R; 280–400 nm) in these systems was largely a function of DOC concentration, as has been reported previously, UV-R penetrated more deeply in saline waterbodies than in freshwater systems with similar DOC concentrations. Power models representing our $K_{d\text{UV-B}}$ or $K_{d\text{UV-A}}$ versus DOC relationships were described by $K_{d\text{UV-B}} = 0.604\text{DOC}^{1.287}$ ($r^2 = 0.76$, $N = 23$) and $K_{d\text{UV-A}} = 0.428\text{DOC}^{1.136}$ ($r^2 = 0.55$, $N = 24$) for freshwater systems and $K_{d\text{UV-B}} = 2.207\text{DOC}^{0.732}$ ($r^2 = 0.40$, $N = 20$) and $K_{d\text{UV-A}} = 1.436\text{DOC}^{0.600}$ ($r^2 = 0.18$, $N = 20$) for saline systems. Our data, when combined with data from other researchers, resulted in the more general freshwater models $K_{d\text{UV-B}} = 0.705\text{DOC}^{1.248}$ ($r^2 = 0.84$, $N = 43$) and $K_{d\text{UV-A}} = 0.470\text{DOC}^{1.112}$ ($r^2 = 0.70$, $N = 44$).

UV-B radiation (280–320 nm) is not expected to penetrate deeply (typically $<$50 cm) in prairie lakes and wetlands because of high intrinsic DOC concentrations. However, the central plains are characteristically windy and this, coupled with the shallowness of many of these systems, suggests that biota may still be at risk from present-day and future-enhanced levels of UV-B (which may result from ozone depletion). Moreover, this risk may be exacerbated in saline systems. This could be significant, especially because saline waterbodies are often highly productive and represent important North American staging areas for shorebirds and waterfowl.

Ultraviolet radiation (UV) has influenced the evolution of life on earth since it first appeared. Even in recent times, in some environments, such as alpine lakes, there is evidence that aquatic organisms have had to adapt to recurring high levels of UV-B (280–320 nm) due primarily to fluctuations in the concentration of dissolved organic carbon (DOC) entering these systems from surrounding catchment basins (Leavitt et al. 1997). Similarly, changes in the ratio of UV-B to UV-A (320–400 nm) and photosynthetically active radiation (PAR = 400–700 nm), as a result of clear-cutting of riparian habitat, for example, are predicted to have profound impacts on benthic diatom and invertebrate species composition in streams (Bothwell et al. 1994). The pronounced shifts in species composition accompanying these changes in optical climate are indicative of the potency of UV radiation to shape lentic and lotic community structure.

The development of Arctic (Rex et al. 1997) and Antarctic (Jones and Shanklin 1995) ozone holes has been attributed to anthropogenically induced reductions in ozone levels over the poles. This has resulted in increased concern regarding the specter of enhanced UV-B levels. Enhanced UV-B ra-

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diation as a result of the thinning ozone layer is, however, not solely restricted to the polar regions; there are strong suggestions that UV-B levels at midlatitudes are also increasing (Kerr and McElroy 1993; Björn et al. 1998). In addition, Yan et al. (1996) have speculated that anthropogenic acidification (by further reducing the diffuse attenuation coefficients of UV-B) and ozone thinning may act in concert to exacerbate the effects of UV-B in poorly buffered aquatic systems.

The effects of UV-R on aquatic life are varied and have been extensively reviewed (e.g., Häder et al. 1995; Vincent and Neale 2000). However, effective prediction of the impacts of UV-R on aquatic organisms must also take into consideration the apparent optical properties (sensu Kirk 1994) of natural waterbodies in terms of UV-R attenuation. Models that allow us to predict the attenuation of UV-R as a function of depth are useful in that they provide insights not only into how present and past levels of UV-R might affect aquatic life, but also how projected increases in UV-B might alter relative productivity or species composition in aquatic habitats. By far the most successful predictor of UV-R attenuation in aquatic systems is the DOC concentration (e.g., see models of Scully and Lean 1994; Morris et al. 1995; Granéli et al. 1996; Lean 1998). These models clearly show that low DOC waters are particularly vulnerable to enhanced UV-B radiation that might result from predicted declines in stratospheric ozone concentrations.

What is not clear, however, is how comprehensive these models are (i.e., in a geographical sense) or how far their generalizations can be extended. This is largely because the chemical nature of the DOC itself varies tremendously (Morris et al. 1995) from region to region. In addition, systems with high chlorophyll (e.g., Hodoki and Watanabe 1998), high turbidity, or pronounced ‘‘surface slicks’’ may produce further, small deviations from simple models that predict UV attenuation solely from DOC concentration. Physical factors, such as wind, and other phenomena that promote turbulent mixing (stream riffles, power-dam outfalls, etc.), can also either decrease or increase the proportion of biota exposed to harmful UV-B fluxes, depending on the circumstances. This is particularly pertinent in the millions of shallow, high DOC, waterbodies scattered across the great central plains of North America, a region where high winds are a normal climatic occurrence.

In this paper, we provide diffuse attenuation coefficients for UV-B and UV-A radiation in a series of lakes and wetlands typical of the great central plains of North America. We demonstrate that models predicting the attenuation of UV-R based on DOC concentration will likely be region specific, especially where there is an expectation that the water chemistry of the region in question will deviate substantially (e.g., salinity, chromophoric properties of DOC) from previously modeled areas. We show for the first time that in freshwater and saline systems having the same DOC concentration, the depth of UV-R penetration is very different. UV-R, in general, penetrates more deeply into the water column of saline systems. This has important implications for the prairie region where saline waterbodies are abundant.

Methods

An Optronics Laboratories Inc. scanning spectroradiometer (Model OL-754) was used to measure UV-B, UV-A, and PAR in 44 waterbodies in central to southern Saskatchewan, Canada. The waterbodies consisted of both lakes and wetlands that were typical of the range and types of aquatic systems of the Great Central Plains of North America (Table 1). The majority were eutrophic and had relatively high concentrations of DOC, typical of standing water on the prairies. Twenty of the waterbodies studied were saline and some of these, although apparently clear, had high ($>$20 mg L^{-1}) DOC concentrations (e.g., Basin Lake, Manito Lake, Redberry Lake). Waters with $>$3.0 mg L^{-1} total dissolved solids were classified as saline (Hammer 1986). All surveys were conducted between 0900 and 1500 h from May to October.

Prior to each field trip, the scanning spectroradiometer was calibrated (wavelength calibration and optical gain) using an Optronics (Model OL 752-150) dual calibration and gain check source module. An Optronics underwater sensor was then attached to the distal end of a long, light, twin-tube, aluminum swing arm based on the principle of a parallelogram (design schematic available upon request). This arm could be clamped to the transom of the boat, or, in the case of very shallow wetlands, to a sawhorse bolted to a piece of plywood that was then weighted down with sandbags. The waterproof quartz-fiber optic cable (Optronics Model OL 730-7Q-WP) connecting the underwater sensor to the optics head (Optronics Model OL 754-O-PMT) was bolted to the swing arm. In the case of the submersible sphere sensor, the total length of the parallelogram swing arm was 1.9 m. However, since the minicosine sensor is much lighter than the submersible sphere sensor it was possible to use an additional 1.4 m extension rod. The parallelogram swing arm allowed us to deploy the underwater sensors far away from the boat, thus essentially eliminating shading effects while maintaining the sensor heads parallel with the water surface.

When a plastic, opal-glass, or Teflon diffuser is immersed in water, its light transmissivity is less than it was in air. Since an instrument's irradiance responsivity is calibrated in air, a correction for this change in collector transmissivity must be applied to obtain irradiance responsivity coefficients for underwater measurements (Mueller and Austin 1995). Accordingly, we applied immersion correction factors specific to both the Teflon diffuser of the minicosine sensor and to the acrylic dome of the submersible sphere sensor to the unweighted energy measurements obtained from each scan.

We calculated vertical attenuation coefficients K_d following the method outlined in Kirk (1994). Briefly, we separately integrated the immersion-corrected irradiance measurements over each of the three wavebands (UV-B, UV-A, and PAR) and at each depth. Diffuse attenuation coefficients for downward irradiance (K_d) were then determined from the slope of the linear regression of the natural logarithm of these integrated downwelling irradiance values (E_d) for each waveband versus depth (z). We are confident that these measurements are robust for the following reasons: (1) We took care to make our measurements only on days when it was

Table 1. Survey dates, locations, dissolved organic carbon (DOC) concentrations, conductivities, and diffuse vertical attenuation coefficients (K_d) for the lakes and wetlands included in this study. Saline waterbodies are italicized. ID, insufficient data.

Site identification	Survey date	Latitude	Longitude	Conductivity (μohms cm^{-1})	DOC (mg L^{-1})	$K_{d\text{UV-B}}$	$K_{d\text{UV-A}}$	$K_{d\text{PAR}}$
Saskatchewan lakes								
Arthur Lake	31 Aug 1998	52°34′N	105°26′W	18,300	50.7	69.1	39.6	10.0
Basin Lake	11 Sep 1996	52°38′N	105°17′W	20,700	58.7	20.4	6.4	1.6
Big Quill Lake	22 Oct 1997	51°55′N	104°22′W	30,900	81.7	63.0	26.0	8.7
Burton Lake	6 Aug 1997	52°16′N	105°07′W	800	20.1	31.8	9.7	1.3
Constance Lake	16 Jul 1997	53°10′N	106°56′W	1,036	29.0	8.0	2.8	0.5
Crawford Lake	8 Jul 1998	52°04′N	106°16′W	11,940	42.3	60.4	16.9	1.7
Deadmoose Lake	6 Aug 1997	52°19′N	105°10′W	21,200	35.7	18.8	5.4	0.6
Diefenbaker Lake	4 Oct 1996	51°10′N	105°15′W	499	4.1	7.4	2.7	0.5
Emerald Lake	17 Jun 1997	53°11′N	106°56′W	1,260	20.9	7.5	2.2	0.2
Fishing Lake	22 Oct 1997	51°51′N	103°33′W	3,580	24.8	13.4	4.9	0.7
Houghten Lake	25 Aug 1998	52°26′N	105°05′W	35,600	126.9	85.5	35.6	8.4
Humboldt Lake	23 Jul 1997	52°09′N	105°06′W	2,430	29.8	20.0	7.7	1.2
Iroquois Lake	17 Jul 1997	53°10′N	107°02′W	632	23.3	13	4.3	0.6
Last Mountain Lake	12 Aug 1997	51°55′N	106°09′W	3,160	29.8	26.2	7.3	1.0
Lenore Lake	7 Aug 1997	52°30′N	104°59′W	5,333	34.1	17.1	6.1	1.1
Little Manitou Lake	4 Oct 1996	51°48′N	105°30′W	61,100	60.8	27.5	7.5	1.4
Lost Lake	29 Jul 1996	52°43′N	107°15′W	1,142	15.6	12.5	5.9	1.3
Martin's Lake	17 Jun 1997	52°59′N	107°01′W	915	22.8	26.7	6.8	1.0
Manito Lake	27 Aug 1998	52°45′N	109°42′W	38,300	114.7	11.6	2.3	0.2
Redberry Lake	29 Jul 1996	52°43′N	107°09′W	16,000	35.6	5.1	1.6	0.3
Tramping Lake	10 Sep 1997	52°08′N	108°47′W	26,000	127.7	60.0	24.1	8.1
Unnamed Lake 1	2 Sep 1998	52°29′N	106°29′W	74,300	156.2	102.4	20.9	2.2
Unnamed Lake 2	2 Sep 1998	52°29′N	106°29′W	42,800	81.6	98.6	50.3	16.3
Wakaw Lake	29 Jul 1997	52°40′N	105°35′W	3,560	22.1	9.2	3.5	0.6
Waldsea Lake	5 Aug 1997	52°17′N	105°12′W	20,100	30.6	14.3	4.0	0.6
Waskesiu Lake	9 Aug 1996	51°10′N	105°15′W	300	7.9	5.2	2.2	0.6
St. Denis National Wildlife Refuge								
Pond 1	1 Oct 1997	52°13′N	106°06′W	1,943	41.0	54.0	20.1	5.1
Pond 15	15 May 1997	52°13′N	106°06′W	260	23.5	31.0	12.8	2.4
Pond 25	7 Jul 1998	52°13′N	106°06′W	2,090	38.0	91.6	21.2	1.6
Pond 26	7 Jul 1998	52°13′N	106°07′W	2,280	40.3	93.2	33.7	3.0
Pond 65	25 Jul 1996	52°13′N	106°06′W	2,500	28.0	45.5	17.7	1.9
Pond 66	29 Aug 1996	52°13′N	106°06′W	18,550	77.9	74.9	25.9	6.2
Pond 79E	30 Jun 1998	52°13′N	106°06′W	764	80.1	165.4	54.7	6.1
Pond 80	30 Jun 1998	52°13′N	106°06′W	6,440	53.8	70.2	31.9	6.6
Pond 4857	13 Aug 1997	52°13′N	106°05′W	1,040	40.0	42.7	19.4	2.4
Pond 5338	30 Jun 1998	52°13′N	106°06′W	24,200	68.8	58.6	15.0	1.7
Other wetlands								
Gursky's Pond	26 Jul 1996	52°09′N	106°07′W	780	27.1	37.4	15.3	2.9
Leicht Cluster	31 Jul 1996	52°17′N	104°21′W	318	15.4	43.2	18.1	3.9
Loiselle Cluster #1	30 Jul 1996	52°18′N	106°05′W	966	25.1	59.6	18.2	2.3
Loiselle Cluster #2	30 Jul 1996	52°18′N	106°04′W	1,395	32.5	41.2	21.7	2.5
Naharney Cluster	29 Jul 1996	52°42′N	107°09′W	2,900	31.3	66.5	20.1	2.6
Weiland Cluster #1	1 Aug 1996	52°23′N	105°05′W	1,336	38.3	I.D.	39.0	3.7
Weiland Cluster #2	1 Aug 1996	52°16′N	105°20′W	923	35.4	76.3	44.2	4.4
Weiland Cluster #3	1 Aug 1996	52°16′N	105°20′W	884	40.6	81.5	61.4	1.9

not overly windy ($<$15 km h^{-1} and, more typically, $<$10 km h^{-1}) or in quiet, sheltered, bays. (2) We usually used a fine stainless steel rod attached to the sensor head and marked off in 1-cm graduations to measure the depth of each scan (*but see below*). (3) We only made measurements on clear–cloudless days or, when clouds were present, only during times when they were not obscuring the sun. (4) Measurements were made in the surface mixed layers where there is little or no variation in DOC concentration with depth. (5) The use of the parallelogram swing arm ensured that there were no shading effects (e.g., ship effects, shadows from the observers, etc.). The regression coefficients from the ln (UV-B or UV-A irradiance) versus depth regressions reflected the robustness of this data in that with the exception of four waterbodies with $0.90 < r^2 < 0.97$, they were always $>$0.98. In systems where the attenuation of downwelling irradiance was especially great, we placed a stack of ceramic tiles (each tile being 0.7 cm thick) on the bottom of the pond and placed the sensor on top of the tiles. The boat was then backed away from this tile platform (≥5 m) and a scan was taken. Subsequently, we removed one tile and repeated the scan. In this way, very discrete measurements of irradiance at depth were obtained.

In 1996 we used an Optronics (Model OL 86-T-WP) submersible right-angle Teflon cosine receptor (hereafter called the minicosine sensor) in our surveys, whereas in 1997–1998 an Optronics Model OL IS-470-WP submersible sphere assembly (hereafter called the submersible sphere sensor) was used. This substitution was made because new information (R. Young pers. comm.) indicated that the performance (cosine response) of the minicosine sensor was somewhat inferior to that of the submersible sphere sensor. K_d values for UV-B, UV-A, and PAR obtained using the two different sensors were compared and verified. This was done by sampling the full spectrum irradiance (280–800 nm) as a function of depth in 11 waterbodies spanning a range of DOCs (20.6–127.7 mg L^{-1}) using both sensors and comparing the resulting $K_{d\text{UV-B}}$, $K_{d\text{UV-A}}$, and $K_{d\text{PAR}}$ values. Because the slopes of these three relationships were all ≥0.97 and coefficients of determination (r^2) were all >0.98, we included the K_d values measured with the minicosine sensor in our data set. A small (<2%) correction factor was applied to the K_d values from the minicosine sensor based on the coefficients from the least squares regressions of the K_d minicosine sensor versus K_d submersible sphere sensor relationships.

On each trip, we also collected samples for chlorophyll (Chl) and DOC. Chl *a* was determined after filtering appropriate volumes of water through Whatman GF/C filters and then extracting the Chl *a* in 90% boiling ethanol. Chl *a* fluorescence was measured on a Turner Designs digital fluorometer (Model 10-AU) with no correction for phaeopigments. In 1996 and 1997, water samples (8 ml) for DOC were filtered through a 0.45-μm Whatman GF/F filter and acidified with 0.4 ml of concentrated phosphoric acid (85–87% w/w) to remove dissolved inorganic carbon. Samples were stored at 4°C in glass bottles with Teflon-lined tops. DOC concentration was measured using a Shimadzu (Model TOC-5050AXX) carbon analyzer connected to an Shimadzu autosampler (Model ASI-5000A) following combustion of the aqueous sample and detection of the CO_2 gas in a nondispersive infrared gas analyser. In 1998, water samples for DOC were filtered through 0.2-μm cellulose–acetate filters, and DOC was measured on a Dohrmann Model DC-190 carbon analyzer (Rosemount Analytical) equipped with an autosampler. Samples were acidified (20% H_3PO_4) and sparged (5 min, carrier air) by the autosampler just prior to injection to remove dissolved inorganic carbon. The catalyst bed on this instrument (Pt on alumina) was held at 900°C, and the CO_2 evolved from combustion of the DOC was determined by nondispersive infrared gas analyzer (Fuji Electric, Model 3300, Milton Roy). The instrument was calibrated daily with potassium hydrogen phthalate of known concentration. Each sample is reported as the average of four to five replicates and corrected for the system blank (<1 mg C L^{-1}), which was determined daily using laboratory purified (E-Pure) water. Precision of replicates was <3% at the 10 mg C L^{-1} level.

Results

The 44 lakes and wetlands in this survey are located within a ~200 km radius of Saskatoon, Saskatchewan. Conductivity ranged from 270 to 74,300 μohms cm^{-1}, DOC ranged from 4.1 to 156.2 mg L^{-1} (Table 1), and Chl *a* concentrations ranged from 0.5 to 729 μg L^{-1} over the sampling period.

The attenuation of UV-B and UV-A radiation in Saskatchewan lakes and wetlands was largely a function of DOC concentration (Figs. 1 and 2, respectively, and Table 2). However, as expected, UV-A (Fig. 2) radiation penetrated more deeply into the water column than did UV-B (Fig. 1). The linear relationships for the UV-B and UV-A versus DOC regressions in freshwater and saline systems were both significant; however, the slopes of the UV-B and UV-A versus DOC relationship for saline waterbodies was significantly less than that of the freshwater systems (Table 2). Thus, for the first time, our data demonstrate that UV radiation in prairie saline systems penetrates more deeply into the water column for a given DOC concentration than it does in freshwater (Figs. 1, 2, Table 2).

We compared our freshwater $K_{d\text{UV-B}}$ and $K_{d\text{UV-A}}$ versus DOC relationship to those of Scully and Lean (1994) and Granéli et al. (1996). There was no significant difference (ANCOVA, $P > 0.05$) between the slope of either of our relationships (UV-B or UV-A versus DOC) and those of Scully and Lean (1994); thus, these two relationships are part of the same overall trend (Table 3). The slopes of both the $K_{d\text{UV-B}}$ and $K_{d\text{UV-A}}$ versus DOC relationships of Granéli et al. (1996) were, however, significantly different (ANCOVA, $P < 0.05$) from the ones reported here.

We also constructed simple multiple regression models for both UV-B and UV-A versus DOC that included Chl *a* concentrations for both freshwater and saline systems. In all cases, the addition of Chl *a* concentration to the simple K_d versus DOC models made only a marginal improvement (<2%) to the regression coefficients (r^2); therefore, those results are not reported here.

The depths of 1% surface irradiance ($z_{1\%}$) for UV-B and UV-A provide a concise intuitive measure of the penetration of each waveband under real field conditions (Figs. 1 and 2, respectively). In these high-DOC prairie systems, UV radiation did not penetrate to great depths (<100 cm and <300 cm for UV-B and UV-A, respectively). Two of the saline lakes (Redberry Lake and Manito Lake; $z_{1\%\ \text{PAR}} = 14.2$ and 29.5 m, respectively) and one of the freshwater lakes (Emerald Lake; $z_{1\%\text{PAR}} = 23.0$ m) demonstrated exceptionally high penetration of PAR given their DOC concentrations (Table 1). The general trend for all three wavebands was for greater penetration in saline systems compared with freshwater systems of the same DOC concentration.

Discussion

It is imperative to further our understanding of the factors that affect how far ultraviolet radiation will penetrate into

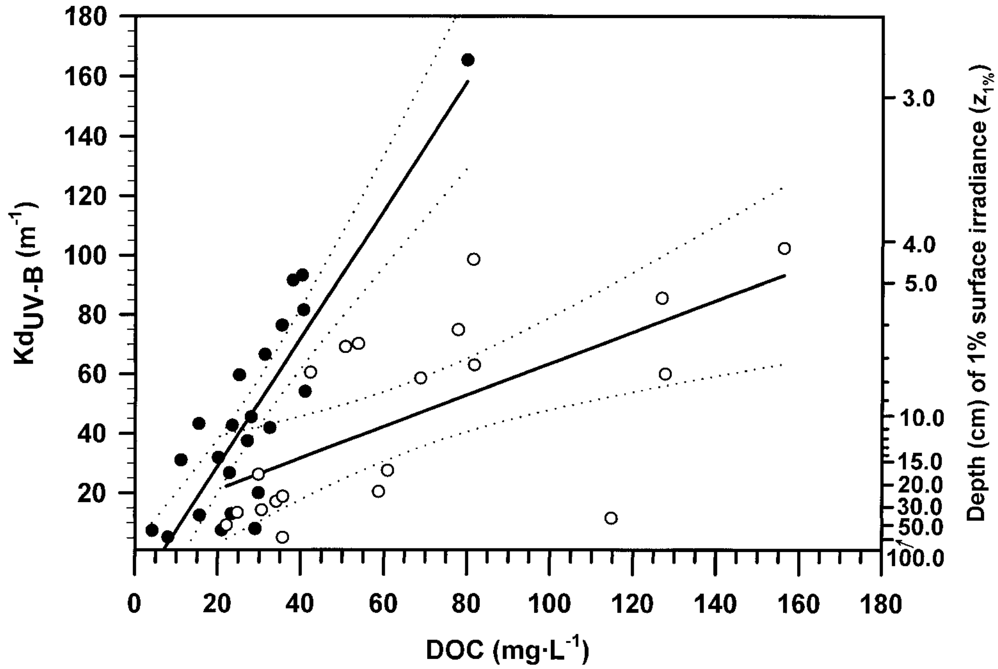

Fig. 1. Vertical attenuation coefficients (K_d, in m^{-1}) and depths ($z_{1\%}$, in cm) at which 1% of the surface irradiance remained as a function of DOC concentration for UV-B radiation (280–320 nm). Filled circles = freshwater lakes; open circles = saline lakes. Dotted lines are the 95% confidence intervals about each regression. Regression statistics are provided in Table 2.

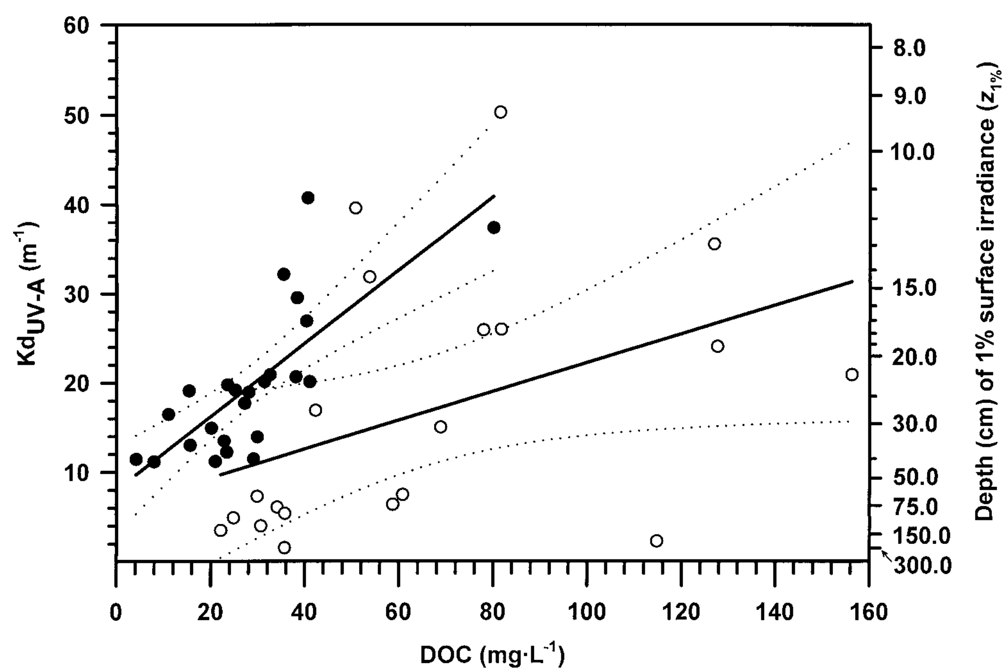

Fig. 2. Vertical attenuation coefficients (K_d, in m^{-1}) and depths ($z_{1\%}$, in cm) at which 1% of the surface irradiance remained as a function of DOC concentration for UV-A radiation (320–400 nm). Filled circles, freshwater lakes; open circles, saline lakes. Regression statistics are provided in Table 2.

Table 2. Regression statistics (ANCOVA) comparing slopes of the $K_{dUV\text{-}B}$ and $K_{dUV\text{-}A}$ versus DOC relationships between Saskatchewan fresh and saline waterbodies. *N*, number of waterbodies sampled.

	SS DOC*	SCP†	SS K_d	*N*	Slope	Intercept	r^2	Residual SS	Residual DF
UV-B									
Freshwater regression	5,146	11,064	31,280	23	2.15	−13.95	0.76	7,491	21
Saline regression	28,757	15,229	19,549	20	0.53	10.51	0.41	11,484	18
Pooled regression								18,975	39
Common regression	33,903	26,293	50,829		0.78			30,437	40
Total regression	49,160	25,945	50,837	43				37,144	41
UV-A									
Freshwater regression	5,249	4,317	6,236	24	0.82	−4.08	0.57	2,685	22
Saline regression	28,757	4,640	3,913	20	0.16	6.16	0.19	3,164	18
Pooled regression								5,850	40
Common regression	34,005	8,957	10,149		0.26			7,789	41
Total regression	49,211	7,943	10,216	44				8,934	42

* SS, sum of squares.
† SCP, sum of cross products.

the water column of lakes and wetlands. This is because UV radiation, and in particular UV-B radiation, produces a broad spectrum of genetic and cytotoxic effects in aquatic organisms (Williamson 1995; Häder 1996; Vincent and Neale 2000). UV-B radiation produces photochemical effects; for example, photoinduced toxicity of polycyclic aromatic hydrocarbons (e.g., Arfsten et al. 1996), photochemical formation of reduced oxygen species such as hydrogen peroxide (Scully et al. 1995), and photolysis and photobleaching of DOC (Miller 1998). UV radiation may also interact with other stressors such as pesticides (Kasai and Arts 1998); however, it is not known whether these sorts of interactions are in general additive or synergistic. These processes and reactions have profound effects on aquatic systems—both directly, for example, on the development and survivorship of organisms and indirectly through their influence on geochemical processes.

DOC concentration has been shown to be a key factor governing the attenuation of UV radiation in aquatic systems (Scully and Lean 1994; Morris et al. 1995; Granéli et al. 1996; Lean 1998). Scully and Lean's (1994) empirical model for freshwater temperate lakes, for example, is of the form $K_{dUV\text{-}B} = 0.415\text{DOC}^{1.86}$. Accordingly, for a freshwater wetland with a DOC concentration of, for example, 40 mg L^{-1}, the Scully and Lean exponential model predicts that $K_{dUV\text{-}B}$ should be ~400 m^{-1}. Our linear $K_{dUV\text{-}B}$ versus DOC relationship (Fig. 1; Table 2), however, predicts that $K_{dUV\text{-}B}$ should be closer to ~65 m^{-1}. Given the broader range of DOC concentrations represented by our data set compared with Scully and Lean's 1994 study and the lack of statistical difference between the slopes of the two linear relationships, we suggest that a combination of the data sets may be more representative of the general $K_{dUV\text{-}B}$ versus DOC relationship for freshwater systems.

Although linear models provide a comparable fit (r^2) and were used to justify combining our freshwater data set with

Table 3. Regression statistics (ANCOVA) comparing slopes of the $K_{dUV\text{-}B}$ and $K_{dUV\text{-}A}$ versus DOC relationships between Saskatchewan freshwater lakes and wetlands and the Scully and Lean (1994) dataset for temperate lakes. *N*, number of waterbodies sampled.

	SS DOC*	SCP†	SS K_d	*N*	Slope	Intercept	r^2	Residual SS	Residual DF
UV-B									
Freshwater regression	5,146	11,064	31,280	23	2.15	−13.93	0.76	7,491	21
Scully and Lean regression‡	89	236	714	20	2.65	−3.63	0.87	89	18
Pooled regression								7,580	39
Common regression	5,235	11,300	31,995		2.16			7,602	40
Total regression	11,449	21,518	48,794	43	1.88	−3.72	0.83	8,353	41
UV-A									
Freshwater regression	5,249	4,317	6,236	24	0.82	−4.08	0.57	2,685	22
Scully and Lean regression‡	89	86	99	20	0.97	−1.07	0.85	15	18
Pooled regression								2,701	40
Common regression	5,338	4,403	6,334		0.82			2,702	41
Total regression	11,903	8,842	9,335	44	0.74	−1.09	0.70	2,768	42

* SS, sum of squares.
† SCP, sum of cross products.
‡ Based on a simple linear regression.

Table 4. Simple power models for prairie freshwater and saline UV-B and UV-A versus DOC relationships as well as for the relationships when the prairie data for freshwater systems were pooled with those of Scully and Lean (1994). Equations are of the form: $K_d = a \times DOC^b$. *N*, number of waterbodies sampled.

Class	N	a	b	r^2
Prairie freshwater (UV-B)	23	0.60	1.29	0.76
Prairie freshwater (UV-A)	24	0.43	1.14	0.55
Pooled freshwater (UV-B)	43	0.71	1.25	0.84
Pooled freshwater (UV-A)	44	0.47	1.11	0.70
Prairie saline (UV-B)	20	2.21	0.73	0.40
Prairie saline (UV-A)	20	1.44	0.60	0.18

that of Scully and Lean (1994), the presence of significant intercepts cause these models to return negative K_d values at low DOC concentrations. For this reason, the use of power models is considered advantageous (Morris et al. 1995). Power models describing our $K_{d\text{UV-B}}$ or $K_{d\text{UV-A}}$ versus DOC relationships as well as the pooled relationships are given in Table 4. We recommend, for the reason cited above, the use of the power models (Table 4) over the linear regressions provided in Tables 2 and 3. Although the original data set of Scully and Lean (1994) was recently extended (Lean 1998) to include an additional 11 humic ponds and lakes (DOC <20 mg L^{-1}), Lean (1998) suggests that their K_d versus DOC regression may differ from those in prairie systems, particularly saline ones. Our main conclusion is that their freshwater relationship and ours are likely part of the same general extended relationship and that the K_d versus DOC relationship in saline systems is a fundamentally different one.

Our linear $K_{d\ \text{UV-B}}$ versus DOC relationship and, by extension, that of Scully and Lean (1994) differed significantly from the one presented in Granéli et al. (1996). This may be attributed to the use of different instruments to profile the underwater light climate. Granéli et al. (1996) acknowledge that the International Light Model IL 1400A radiometer that they used had ''a weak response signal for energy with 315–320 nm, which is unfortunate because most of the UV-B energy is within this range.''

Although empirical models relating $K_{d\text{UV-R}}$ to DOC concentration help us to predict what the unweighted UV-R dose will be at depth, it is by no means clear how widely applicable these models are. The exact chemical composition of the DOC, and in particular the chromophoric portion of the DOC, will largely determine the final K_d values (Miller 1998). Morris et al. (1995) predicted that $K_{d\text{UV-B}}$ versus DOC models would be region specific because of this tremendous variability in DOC composition. Our results provide a striking example in support of this hypothesis. It may also be that other factors (e.g., chlorophyll, surface slicks, turbidity, etc.) contribute, albeit in a limited way, to the attenuation of UV-R.

The coefficients of determination (r^2) of the K_d versus DOC relationships for both UV-B and UV-A in our saline systems were much lower than those of the freshwater systems. It is important to realize that this is not due to the contribution of dissolved inorganic salts, whose absorptive properties in the UV range are considered to be negligible, especially in the UV-B region (Kirk 1994). In addition, we suggest that the larger deeper saline lakes will tend to have lower K_d values for a given DOC concentration than the shallower saline wetlands. The most likely explanation for this is the inherent chemical variation in the composition of DOC. We suspect that the DOC in the deeper saline systems is older (longer residence times), more degraded, and photobleached. Four such lakes, which fall outside the 95% confidence limits for the saline linear regressions (Figs. 1, 2), include Redberry Lake, Basin Lake, and lakes Little Manitou and Manito (in increasing order of DOC concentration). Manito Lake is particularly remarkable because it has a DOC concentration of 114.7 mg L^{-1}, $K_{d\text{UV-B}}$ and $K_{d\text{UV-A}}$ of only 11.6 and 2.3 m^{-1}, respectively and a $z_{1\%\text{PAR}}$ of 29.5 m. The coefficient of determination (r^2) of the $K_{d\text{UV-B}}$ versus DOC relationship for saline systems increases from 0.41 to 0.63 when these four lakes are omitted from the regression model. Note, however, that despite the inclusion of these four lakes, the slopes of the freshwater and saline regressions are still significantly different (ANCOVA, $P < 0.005$).

Clarification of these issues with regard to inland saline systems is important because the claim has been made that ''the total volume of the world's inland saline lakes and wetlands is only slightly less (~83%) than that of freshwater systems'' (Hammer 1986). In addition, saline lakes and wetlands are very productive systems and are important staging and feeding areas for waterfowl (Batt et al. 1992) and shorebirds (Skagen and Knopf 1994). Our data show that both UV-B and UV-A radiation attenuate less at depth in saline compared with freshwater systems at a given DOC concentration (Figs. 1 and 2; Table 2). Thus, the biological and photochemical effects caused by UV-R will, on average, persist deeper in the water column in saline compared with freshwater systems. This, in conjunction with the known effects of UV-R on biota and biogeochemistry, makes it especially crucial to predict the attenuation of UV radiation in saline systems.

Many of the saline waterbodies in this study are relatively noncolored despite having high DOC concentrations (e.g., Basin Lake, Redberry Lake, and especially Manito Lake). Saline systems are often ''end systems'' located at the lowest hydrological levels possible for surface waters, and saline lakes are most common in areas with endorheic drainage basins (where precipitation is exceeded by evaporation). By virtue of long water residence times, the DOC in these saline systems is very old (Curtis and Adams 1995), relatively degraded, photobleached, and generally less available to bacteria (Waiser and Robarts 1997). Furthermore, this DOC has been characterized (by nuclear magnetic resonance spectroscopy) as having a reduced concentration of aromatic compounds compared with the inflowing source DOC (Waiser and Robarts unpubl. data from Redberry Lake). This may account in large part for increases in the $z_{1\%}$ depths for UV-R that we observed in saline systems compared with freshwater systems of comparable DOC concentration. Although we do not know how widespread these low-color, high DOC, inland saline systems are, they are known to exist in Alberta (Curtis and Adams 1995) and British Columbia (Overmann et al. 1996).

Despite the fact that UV-B penetrates only a short distance (Fig. 1) in these high DOC systems, the potential for strong ecological effects still exists. This is because prairie wetlands are typically shallow (average z_{max} ~1 m in depth) so that the upper 30-cm strata represents a significant proportion of their total volume. Furthermore, midcontinental climates are characteristically windy. For example, wind speed data collected from a meteorological station moored in the center of Redberry Lake during the ice-free seasons of 1996–1997 (Arts unpubl. data) provided average maximum and minimum daily wind speeds of 28.9 and 6.7 km h^{-1}, respectively. Such recurrent high winds, typical of the area, help ensure that these waters, and their associated plankton, are well mixed.

Although studies on the effects of UV-B on saline systems are comparatively rare, the few that have been done have demonstrated significant negative effects of the UV-R component of the solar spectra on natural phytoplankton (Kasai et al. in press) and bacteria (Ferreyra et al. 1997). This, coupled with the relatively high transparency of these systems and their exposure to high winds, signifies a greater potential for UV damage than was previously recognized. Given the volume of water represented by saline systems and their importance to wildlife, the implications for reduced productivity, especially as a consequence of enhanced UV-B levels (as a result of ozone thinning), should not be underestimated.

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