

# A Framework for Climate Biomonitoring with Lichens in British Columbia's Inland Temperate Rainforest

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## Abstract

Rapid climate change is predicted for British Columbia's inland temperate rainforest, a globally unique ecosystem. Lichens, which may serve as effective biomonitors of ecosystem health, have been proposed for use as climate change indicators for various ecosystems globally. This research presents a climate biomonitoring protocol that uses arboreal macrolichen communities in the inland rainforest in British Columbia. We report our initial findings of 39 lichen taxa, including several rare species and cyanolichens, which may be especially sensitive to climate. Comparisons of these data with future measurements will provide an indication of how the inland rainforest may be responding to climate change.

**KEYWORDS:** biomonitoring; lichen; inland temperate rainforest

## Introduction

Some of the world's most famous temperate rainforests occur along the west coast of North America. A lesser known, but equally unique ecosystem is the inland temperate rainforest (ITR), which is predominantly located in British Columbia between 50° and 54°N (Goward & Spribille 2005). The ITR shares many characteristics with its coastal counterparts, including large diameter western redcedar (*Thuja plicata* Donn ex D. Don), some speculated to be over 1000 years old (Radies et al. 2009), and a dense understorey of devil's club (*Oplopanax horridus* [Sm.] Miq.).

This ecosystem provides important habitat for many species, including the threatened mountain ecotype of woodland caribou (Stevenson et al. 2001), and many disjunct populations of characteristically coastal lichen species (Goward & Spribille 2005). Old-growth coniferous stands in the ITR support exceptionally rich epiphytic lichen communities, especially cyanolichens (Campbell & Fredeen 2004; Radies & Coxson 2004; Goward & Spribille 2005; Radies et al. 2009; Spribille et al. 2009), lichens that have a cyanobacterial photobiont as one of their symbiotic partners. Cyanolichens require greater hydration than chlorolichens (those which have only algal photobionts) to be photosynthetically active (Lange et al. 1986), making them especially sensitive to moisture and temperature gradients (e.g., Ellis et al. 2009). The tripartite (i.e., lichens with both a green algal and cyanobacterial photobiont) cyanolichen *Lobaria pulmonaria* (L.) Hoffm. was previously found to be an indicator of macrolichen diversity in the inland rainforest (Campbell & Fredeen 2004), including many bipartite cyanolichens, which are likely more sensitive to climate variables than chlorolichens (e.g., Ellis et al. 2009).

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Given the limited geographical extent of the ITR, it is critical that land management strategies ensure stands with high biodiversity values are represented within protected areas and that appropriate buffer zones are designated around these cores (Stevenson et al. 2011). Unfortunately, in much of the ITR, land management practices have had the opposite effect, where stands with high biodiversity values, such as old forests in wet toe-slope positions, have been disproportionately targeted for logging (Radies et al. 2009). In the Slim Very Wet Cool Variant of the Interior Cedar–Hemlock zone (ICHvk2; DeLong 2003) of the upper Fraser River watershed, for instance, old cedar stands now represent less than 7% of the landscape (Radies et al. 2009). Less than 1.5% of the ICHvk2 landscape has full protected- area status, suggesting a need for greater conservation efforts.

In addition to habitat loss and fragmentation, climate change is threatening old forest-dependent taxa in the ITR. Small, irregularly shaped old forest remnants will be particularly vulnerable to stochastic climate events, due to edge effects and changes in local microclimate (Stevenson et al. 2011). Understanding how the ITR responds to climate change is an urgent priority for managers. Climate change predictions for the ITR show expected increases in mean annual temperature of 2.8–4.0°C in the Very Wet Cool Interior Cedar–Hemlock (ICHvk) subzone (Meidinger & Pojar [editors] 1991), and 1.9–4.1°C in the Wet Cool Interior Cedar–Hemlock (ICHwk) subzone (Meidinger & Pojar [editors] 1991) by the end of the century (Stevenson et al. 2011). Climate models also predict that the increase in temperatures in the ITR will be more pronounced during the winter months, leading to less precipitation falling as snow (Stevenson et al. 2011). This may have major consequences for the ecology of ITR forest stands, as wet soils in the valley bottoms and toe-slope positions have historically been maintained by ground seepage from the annual melt of upslope winter snowpack. These wet soils have resulted in significantly long fire-return intervals in the ITR of between 800 and 1200 years (Sanborn et al. 2006). An increase in fire occurrence, as well as the increased vulnerability of drought-stressed species to pests and pathogens, has the potential to change the composition and dynamics of these forests as soils become drier.

The present study is focussed on creating a biomonitoring protocol that may be useful in the future for understanding climate impacts in the ITR. Epiphytic macrolichens inhabiting riparian mountain alder trunks have been surveyed in old-growth ITR forests. Present abundance levels of these lichens may be compared with future measurements to help understand shifting species distributions that may be attributed to climate change. Epiphytic lichen communities are likely to be disproportionately sensitive to changes in stand climate, given their dependence on atmospheric precipitation and humidity. This dependence also makes these lichens ideal early warning indicators of ecosystem change, as they may show measurable changes before more obvious landscape features such as trees. Lichens have been popular biomonitoring tools for decades, as they absorb pollutants directly from the air around them and generally show sensitivity to these compounds. Historically, lichens have proven to be effective biomonitors for atmospheric pollution, including primarily sulphur dioxide, but also nitrogen compounds, radioactive fallout, and various heavy metals (see Nimis et al. [editors] 2002 for review). More recently, the sensitivity of several lichen species to climate variables has prompted their use as indicators for climate change in many regions around the world (e.g., Insarov & Schroeter 2002; Sancho et al. 2007). For example, evidence that lichen communities are responding to climate change has been documented through long-term monitoring of lichen populations in the Netherlands, where changes in species composition are likely due to increasing temperature (van Herk et al. 2002).



Current lichen biomonitoring efforts for climate change are based on modelling inputs from large data sets in regions where the lichen flora and detailed distributions are very well known (Ellis et al. 2007; Giordani & Incerti 2007; Ellis & Coppins 2010). Distributional data on lichens is often compared with major climatic variables such as rainfall and temperature in order to group lichens by climatic requirements (Will-Wolf et al. 2002; Giordani & Incerti 2007). These guilds may then be used to monitor climatic variables in long-term monitoring efforts (Gombert et al. 2005); however, care must be taken when grouping species by “climatic requirements,” as these are often more complex than assumed (Ellis & Coppins 2010). Species with similar climatic requirements may, for example, have different responses to climate change (Ellis et al. 2009). The use of biomonitoring protocols that employ the entire lichen community, as opposed to one or a few species, may therefore provide a better picture of climate change response in a given ecosystem. The monitoring of entire lichen communities over time has been used extensively in Europe (e.g., Asta et al. 2002; van Herk et al. 2002) and elsewhere in North America (McCune 2000).

Information on the lichen flora of British Columbia and the geographical distributions of lichen species in British Columbia remains somewhat unknown, as indicated by an abundance of new species descriptions from the province in recent years (e.g., Tønnsberg & Goward 2001; Goward & McCune 2007; Spribille et al. 2009; Goward et al. 2010; Jørgensen & Tønnsberg 2010; Lumbsch et al. 2011; Goward et al. 2012), with a great many more soon to be described (T. Goward, pers. comm., May 2011). The comparison of lichen distribution maps from previous site visits with macroclimatic gradients is therefore not a feasible method for understanding the climate responses of lichens in British Columbia, as species distributions are too poorly known. In Europe, where lichenology has a much longer history, more detailed distributional data for many species is available from past field research (e.g., van Herk et al. 2002; Ellis & Coppins 2007). Considerable research has been conducted on the large-scale impacts of climate on *Lobaria pulmonaria* and other *Lobaria* species in Europe (Ellis & Coppins 2007; Ellis et al. 2009).

In the inland rainforest, cyanolichens such as *L. pulmonaria* are exceptionally abundant and diverse in areas where humidity levels are high (Goward & Arsenault 2000b; Radies et al. 2009). Cyanolichens have an important ecological role in temperate forests, as they are able to fix atmospheric nitrogen and may be the sole nitrogen input in these ecosystems (Campbell & Fredeen 2004). Riparian zones are examples of humid habitats that may serve as important lichen biodiversity hotspots and corridors in western North America (McCune et al. 2002; Peterson & McCune 2003; Doering & Coxson 2010), supporting species that are rare or absent from the surrounding upland (non-riparian) forest. The biological significance of riparian zones in maintaining landscape-level lichen biodiversity gives them special conservation importance in forest management (Peterson & McCune 2003). Riparian zones in the inland rainforest are also nutrient-receiving sites (Stevenson et al. 2011), and this may further augment the diversity of epiphytic cyanolichens (Doering & Coxson 2010), which are more abundant where soil nutrient levels are high (Goward & Arsenault 2000b).

For the present research, we have focussed on riparian zones along small first- and second-order ITR creeks with populations of *Alnus incana* ssp. *tenuifolia* (mountain alder) (Stevenson et al. 2011), a genus that supports high cyanolichen diversity (Goward & Arsenault 2000c). Doering and Coxson (2010) found riparian corridors near British Columbia's wet ITR forests to be refugia for old-growth-associated macrolichens, including many rare cyanolichens. Peterson and McCune (2003) similarly studied lichen hotspots in riparian zones in the Pacific Northwest, and found that epiphytic cyanolichens

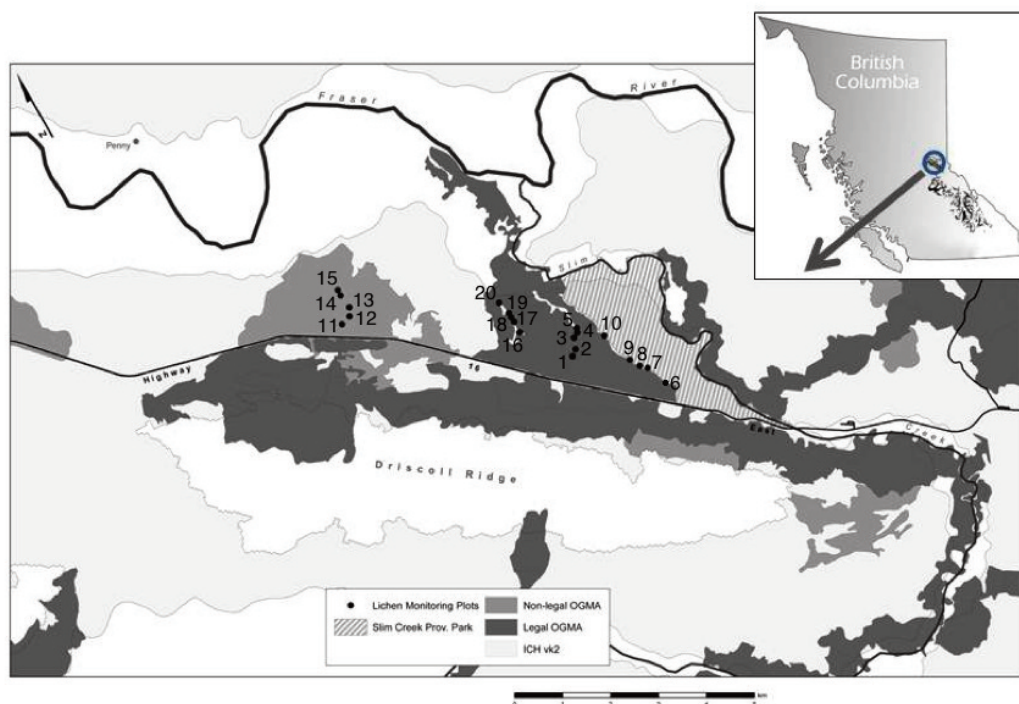


were strongly correlated with the abundance of hardwoods. Light availability, as well as a high pH (Gilbert 1986), likely allow for the diversity of epiphytic cyanolichens found on *Alnus* spp. in the inland rainforest (Goward & Arsenault 2000a). These hardwood trees are recognized as important for maintaining lichen biodiversity in temperate forests around the world (Neitlich & McCune 1997; Dettki & Esseen 1998).

The present study has been designed as a baseline for future research on the effects of climate change in the inland rainforest. The importance of deciduous trees in riparian zones has prompted us to select discrete riparian stands of mountain alder as the location for long-term climate change monitoring plots using epiphytic macrolichens. Although riparian zones will potentially be somewhat buffered from climate changes when compared to upland forests, the diversity of these lichen communities and the narrow climate envelope of the more sensitive species will likely make them effective climate monitors. It is expected that these communities will show measurable changes over time as climate shifts in the inland rainforest, giving advance warning to forest managers of how the ITR may be responding to climate change.

## Methods

Field sites were selected within protected areas on the south side of the upper Fraser River watershed (locally known as the Robson Valley), between Sugarbowl–Grizzly Den and Slim Creek provincial parks, approximately 110 km east of Prince George, B.C. (53°45'27"–53°48'15" N, 121°11'35"–121°15'52" W). Valley-bottom forests in this region fall within the Very Wet Cool Sub-Boreal Spruce (SBSvk) biogeoclimatic subzone (Meidinger and Pojar [editors] 1991), showing a transition at mid-elevation to the Very Wet Cool Interior Cedar–Hemlock (ICHvk2) biogeoclimatic subzone. Field sites were located in the ICHvk2 variant. SBSvk forests are dominated by hybrid white spruce (*Picea engelmannii* x *glauca*) and sub-



**Figure 1.** Location of climate change lichen biomonitoring stands within protected areas (circles). These plots fall within provincial parks (Slim Creek Provincial Park, cross-hatch), legally established Old Growth Management Areas (dark grey), and non-legal (guidance) Old Growth Management areas (medium grey).





alpine fir (*Abies lasiocarpa*). ICHvk2 forests are dominated by western redcedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*), with some Douglas-fir (*Pseudotsuga menziesii*), subalpine fir, and hybrid white spruce, depending on site conditions.

Four drainages were sampled, with five sites located in each drainage (Figure 1). Drainages were separated by areas of upland forest, and were all at approximately the same elevation (685–785 m), aspect (south-facing), and mesoslope position (toe-slopes). Five sites were selected from each of the four drainages, for a total of 20 field sites, and GPS co-ordinates were recorded for each site. Sites consisted of stands of *Alnus incana* ssp. *tenuifolia* (Nutt.) Breitung (mountain alder) clustered along gullies of the drainages. Within each drainage, the alder stands are distinct, separated by conifer stands. The average stand width (based on three measurements) and number of individual alders were documented for each site.

Three individual trees were selected from each field site (total = 60 trees sampled), choosing trees with a high *Lobaria pulmonaria* abundance. All sample trees were located with a GPS unit and marked with a permanent marker tag for future reference. The following variables were measured for each alder tree sampled: tree height, diameter at breast height (DBH), and distance from the creek. Lichen communities on alder stems were sampled at heights of 1.5–3 m above ground, above the average late winter snowpack. Sites were established in late winter 2011, a year of record snowfall, so we do not anticipate future snowpack burial of measured stem segments, although future extreme climate variability cannot be discounted. Only live trunks were sampled. The main trunk of each tree was sampled, as were secondary trunks with a DBH of more than 15 cm.

All macrolichen species present within the sample region on these trunks were recorded and their abundances rated based on the scale in Table 1. The abundance scale that we used is an adapted version of the Braun-Blanquet method, with an added level of definition after more than 10 thalli. Our abundance class definitions were very similar to the United States Department of Agriculture Forest Health lichen monitoring protocols (see <http://www.fia.fs.fed.us/library/field-guides-methods-proc/>), although we applied these to individual trunk segments rather than forest plots.

**Table 1: Abundance classes for macrolichens sampled on the trunks of *Alnus incana* at 1.5–3 m above ground level**

Abundance class	Definition
0	No individuals of this species present
1	1–2 individual thalli of this species present
2	3–10 individual thalli of this species present
3	> 10 thalli, but covering < 50% of trunk area sampled
4	Thalli covering > 50% of sampled trunk area

Correlations between stand width, plot elevation, and stand size (number of alders) were examined against macrolichen species richness using linear regressions. Similarly, the influence of stem distance from the nearest stream edge, diameter at breast height, and stem height were assessed against lichen species richness using linear regression. Data were transformed where necessary to meet assumptions of normality of distribution. Analyses were carried out using STATA statistical software (StataCorp 2011). Unless otherwise noted, all reported mean values are accompanied by their standard deviation.



**Table 2: Average abundance ratings for the lichen taxa observed at all 20 field sites.**  
**Species names follow Esslinger (2010). Species denoted with an asterisk (\*) are cyanolichens.**

Species	Average abundance rating for each of 20 sites																				Average across all sites
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	
<i>Alectoria sarmentosa</i> (Ach.) Ach	0.7	0.0	0.0	1.3	0.7	3.0	2.3	1.0	1.7	2.0	0.7	0.7	0.0	0.0	0.3	0.7	0.0	1.7	0.0	0.0	0.83
<i>Bryoria fuscescens</i> (Gyeln.) Brodo & D. Hawksw.	1.0	2.0	1.7	1.7	2.0	0.7	1.7	1.7	2.0	2.0	2.3	2.0	0.7	1.7	1.7	1.7	2.0	1.7	0.7	1.0	1.58
<i>Cladonia</i> spp.	0.7	1.0	0.3	0.3	1.0	0.7	1.3	0.3	1.7	0.0	0.0	0.0	0.3	0.3	0.3	0.7	0.0	0.0	0.3	0.7	0.50
<i>Dendrocoaulon</i> spp.	1.3	1.3	1.7	2.0	1.3	1.0	0.3	0.7	1.7	1.0	1.7	1.7	1.0	2.3	1.3	1.3	0.7	1.7	2.3	2.0	1.42
<i>Hypogymnia enteromorpha</i> (Ach.) Nyl.	0.7	1.0	0.3	0.3	1.0	0.3	0.0	0.3	1.0	1.3	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.3	0.38
<i>Hypogymnia physodes</i> (L.) Nyl.	0.7	0.3	1.3	1.7	1.0	1.3	0.7	0.3	2.3	1.0	0.3	1.7	1.0	1.7	2.7	0.7	1.3	1.0	0.7	0.0	1.08
<i>Hypogymnia tubulosa</i> (Schaerer) Hav.	0.7	1.7	1.0	0.3	0.7	0.0	0.7	0.7	0.3	0.3	0.0	1.7	0.0	0.7	1.3	0.7	0.3	0.3	0.7	0.7	0.63
<i>Hypogymnia vittata</i> (Ach.) Parrique	3.0	2.7	2.7	2.7	2.7	2.7	3.0	2.7	3.0	3.0	2.7	2.7	2.0	2.7	3.0	2.0	2.3	2.7	1.3	1.3	2.53
* <i>Leptogium burnetiae</i> C. W. Dodge	0.0	0.0	0.0	0.7	0.3	0.0	0.0	1.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.13
* <i>Lobaria pulmonaria</i> (L.) Hoffm.	4.0	3.0	3.7	3.0	3.0	3.7	3.7	3.0	3.3	3.0	3.3	3.7	3.7	3.0	2.7	3.3	3.7	2.0	3.0	3.0	3.23
* <i>Lobaria retigera</i> (Bory) Trevisan	1.0	0.3	0.0	0.0	0.0	0.7	0.7	0.3	0.0	0.0	0.0	0.3	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.7	0.23
* <i>Lobaria scrobiculata</i> (Scop.) DC.	0.3	1.3	1.0	0.0	1.3	0.7	0.0	1.0	1.3	0.0	1.3	1.7	0.0	1.3	1.3	1.7	1.0	0.3	0.7	0.7	0.85
<i>Melanelixia subaurifera</i> (Nyl.) O. Blanco et al.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	1.0	0.0	0.0	1.3	0.0	0.0	0.0	0.0	0.0	0.7	0.3	0.0	0.18
* <i>Nephroma bellum</i> (Sprengel) Tuck.	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	1.3	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.15
* <i>Nephroma helveticum</i> Ach. ssp. <i>sipeanum</i> (Gyelnik) Goward & Ahti	0.0	0.7	0.3	0.3	1.0	0.7	0.0	0.7	0.3	0.0	0.3	1.7	0.7	1.7	1.0	0.0	1.3	0.0	0.3	0.3	0.57
* <i>Nephroma isidiosum</i> (Nyl.) Gyelnik	1.7	1.3	1.7	1.0	1.0	1.0	1.0	1.7	1.3	0.0	1.7	1.7	1.3	2.0	2.0	0.3	1.3	0.3	0.7	0.0	1.15
* <i>Nephroma occultum</i> Wetmore	0.3	0.3	0.0	0.7	0.0	0.7	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.15
* <i>Nephroma parile</i> (Ach.) Ach.	2.7	3.0	2.7	3.0	2.0	3.0	2.7	3.0	2.7	2.3	3.0	3.0	2.7	3.0	3.0	3.0	2.3	3.0	3.0	2.7	2.78
* <i>Nephroma resupinatum</i> (L.) Ach.	0.0	0.0	0.0	0.7	0.7	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.10
<i>Parmelia hygrophila</i> Goward and Ahti	0.0	0.0	1.3	0.7	1.3	1.7	2.0	2.0	3.0	2.7	1.3	2.7	1.7	1.3	1.0	1.7	2.0	2.3	2.0	1.7	1.62
<i>Parmelia sulcata</i> Taylor	3.0	3.0	3.0	3.0	3.0	3.0	3.0	3.7	3.0	3.0	3.0	3.0	3.0	3.3	3.0	3.3	3.0	3.0	3.0	3.0	3.07
<i>Parmeliopsis ambigua</i> (Wulfen) Nyl.	0.0	0.3	0.3	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.05
<i>Parmeliopsis hyperopta</i> (Ach.) Arnold	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.03
* <i>Peltigera collina</i> (Ach.) Schrader	1.0	0.7	0.0	0.7	0.7	0.0	1.7	0.7	1.3	0.3	0.0	0.3	1.3	0.7	0.0	0.3	0.0	0.0	2.0	0.3	0.60
<i>Physcia alnophila</i> (Vainio) Loht., Moberg, Myllys & Tehler	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.02
<i>Platismatia glauca</i> (L.) W.L. Culb. & C.F. Culb.	0.3	1.0	2.0	2.7	1.7	2.0	2.7	1.0	3.0	3.0	0.7	2.0	0.3	1.0	2.3	0.0	1.3	1.7	1.0	1.3	1.55
* <i>Leptogidium dendriscum</i> Nyl.	0.0	0.0	0.3	0.0	0.0	0.7	0.0	0.0	0.3	0.0	1.0	0.7	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.19
* <i>Pseudocyphellaria anomala</i> Brodo and Ahti	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	1.0	2.3	0.0	0.3	1.7	0.7	0.7	0.0	0.3	0.7	0.42



## Results

A total of 39 different macrolichen taxa were found on the alder trunks on the sample trees (Table 2). *Alectoria sarmentosa* and *Lobaria pulmonaria* were the only species to reach an abundance class of 4. *Lobaria pulmonaria* was the most abundant species, found on sample trees at all sites, with an average abundance rating greater than 2 at each site (Table 2). The least abundant species were *Physcia alnophila* and *Ramalina pollinaria* (Table 2).

Tree heights ranged from 3.6 to 13.4 m (average:  $9.5 \pm 2.3$  m) measured in vertical distance from ground level (i.e., not considering the increased actual height of leaning trees). Diameter at breast height ranged from 7.0 to 26.1 cm (average:  $18.8 \pm 3.9$  cm). The distance of each tree from the creek ranged from 0 to 32 m (average:  $7.7 \pm 6.6$  m).

The total macrolichen species counts on the sampled alder trunks for each drainage were 34, 29, 36, and 32 species. Species diversity at each site ranged from 18 to 29 species, with an average of  $23.2 \pm 3.0$  species per site. The number of macrolichen species on individual alder trunks surveyed ranged from 9 to 24 species (average:  $16.2 \pm 3.5$ ). A total of 17 field-discernable cyanolichen taxa were observed on our sample trees, with a maximum of 13 at a single site (Table 2). The mean number of cyanolichen species at each site was  $9.25 \pm 2.17$ .

Elevation did not have an effect on site diversity ( $p = 0.41$ ), and neither did the width of the stand being surveyed ( $p = 0.38$ ). Stand size (number of trees per stand) also did not have a significant effect on site-level macrolichen diversity on the three stems measured in each stand ( $p = 0.09$ , data transformed with an inverse square root function).

At the level of individual trees, tree height had no significant effect on species richness ( $p = 0.369$ ), and neither did the distance from the sample tree to the stream ( $p = 0.399$ , data transformed using the square root function). Diameter at breast height had a significant negative effect on species richness ( $r = -0.289$ ,  $p = 0.014$ ), even after studentized residuals greater than 2 were dropped ( $r = -0.305$ ,  $p = 0.007$ ).

## Discussion

Our research has noted numerous provincially rare and uncommon species, many of which are cyanolichens, as well as high *L. pulmonaria* abundance. These results are consistent with recent lichenological research in the area that has found diverse lichen assemblages (Goward & Arsenault 2000b; Goward & Spribille 2005; Radies et al. 2009) and further validates the ITR as an especially diverse and unique ecosystem. The number of locally and globally rare species found in this study also highlights the importance of ITR riparian zones in maintaining regional biodiversity. The cryptic paw (*Nephroma occultum*), a species of special concern (Committee on the Status of Endangered Wildlife in Canada 2006), was found on 13% of the studied trees. The British Columbia Conservation Data Centre (B.C. Ministry of Environment 2012) ranks this species, as well as *Nephroma isidiosum*, *Lobaria retigera*, *Leptogidium dendriscum*, *Sticta oroborealis*, *Sticta wrightii*, and *Usnea glabrata*, as rare and of conservation priority in the province. A number of the species found along the creek drainages may also be considered old-growth indicators in the ICH, including *Nephroma helveticum*, *N. isidiosum*, and *Sticta fuliginosa* (Goward & Arsenault 2000b; Campbell & Fredeen 2004).

The observed negative influence of diameter at breast height on macrolichen species richness was, at first, puzzling. We suggest that this may occur because of competitive exclusion of lichen taxa on the larger sampled trees, especially by the dominant species *L. pulmonaria*. Based on field observations, alder stems smaller than the ones included in our study also hosted less lichen diversity. Doering and Coxson (2011) also found stems



below 10 cm DBH to have much lower lichen diversity. We therefore suggest that the smaller diameter, higher diversity trunks that were sampled represent the peak diversity lichen floras of this study, and that similar-sized trunks should be added in future monitoring efforts as this cohort of alders ages. The variables of elevation, stand width, number of trees per stand, tree height, and distance from the sample tree to the stream did not appear to influence species richness. This supports the similarity of sample units in this study, indicating that none of these variables are currently exerting a strong influence on the lichen communities within the sampled area.

Future effects of climate change on lichens such as the communities we have documented will be complex. Climate and other variables must be suitable for all symbionts before establishment can occur (Goward 2011). Although establishment may be limited by climate, established thalli may also continue to persist where climate becomes unsuitable for further establishment (Ellis & Coppins 2007). This can be misleading for climate monitoring, as persisting thalli may influence models to deduce the climatic requirements of indicator species (Ellis & Coppins 2007). We have attempted to account for this by recording the abundance class of each species. We expect that abundance levels will change more rapidly over time than will absolute presence/absence data, and therefore be a more sensitive climate biomonitoring protocol, as declining abundance will be measured even as older thalli persist.

Winter climate warming, as is predicted for the ITR (Stevenson et al. 2011), has the potential to harm lichen communities (Bjerke 2011). In mild subfreezing temperatures, which are warm enough for hydration, physiologically active lichens will respire and use up valuable carbon reserves during the low light of the winter months, when rates of photosynthesis are low (Schroeter & Scheidegger 1995; Bjerke 2011). The formation of extracellular ice crystals is also a concern as temperatures fluctuate (Schroeter & Scheidegger 1995). Climate change response in lichens depends on several variables, from the differing light requirements of photobiont partners, to ecosystem-level changes in environmental conditions. The diverse assemblage of lichen species we have documented here will likely show an array of responses to climate change in the inland rainforest. It is hoped that this diversity of response will shed light not only on how climate change is affecting the ITR, but also on how the individual species we have documented are likely to respond to climate change in general. Climate change itself, as mediated by policies around the globe, will determine the success or failure of each individual species.

Many issues should be addressed when designing a long-term biomonitoring protocol (Ferretti & Erhardt 2002; Insarov & Schroeter 2002; Will-Wolf et al. 2002). An important issue for epiphytic lichens is longevity of substrate. In the ITR, branches of *A. incana* are frequently bent or broken off in years of heavy snowpack. The present study was therefore restricted to alder trunks, which last longer than individual branches. Climate change has the potential to indirectly affect the lichens we have studied, through influences on their mountain alder phorophytes. It is likely, however, that alder communities will continue to persist along the bottom of creek drainages, where the soil moisture is too high for conifers, until drastic changes in ITR climate have occurred. Changes around individual alder stems may also have an impact on lichen communities, by changing microenvironments. Lichen microclimate availability may therefore be altered by climate change impacts.

We have tried to ensure that field sampling methods will be repeatable, although future sampling will require trained lichenologists. We have chosen a diverse lichen community for biomonitoring, providing a greater probability of differential response by various com-





munity members. Ideally, long-term biomonitoring plots should be situated in protected areas, where plots will not be disturbed by other human-caused activities (Insarov & Schroeter 2002). Our plots all fall within areas with some degree of protection, from the quite high level of protection provided by provincial parks (Slim Creek Provincial Park) to the lesser protection provided by Old Growth Management Areas (OGMAs), which prevent logging but not other resource extraction or road access development (see DataBC 2009 for a description of legal OGMAs, and Forest Practices Board (2008) for a discussion of the limitations of “guidance” or non-legal OGMAs). The information gathered from these plots will both assist management of these protected areas and provide a benchmark against which changes in surrounding forests in the timber harvesting land base can be assessed.

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# Test Your Knowledge

How well can you recall the main messages in the preceding article?  
Test your knowledge by answering the following questions.

## **A Framework for Climate Biomonitoring with Lichens in British Columbia's Inland Temperate Rainforest**

A FRAMEWORK  
FOR CLIMATE  
BIOMONITORING  
WITH LICHENS IN  
BRITISH COLUMBIA'S  
INLAND TEMPERATE  
RAINFOREST

MacDonald,  
Coxson, &  
Björk

1. Which of the following most favours cyanolichen growth?
  - a) Humid habitats
  - b) Open forests
  - c) Alpine ecosystems
  
2. Climate change impacts in the inland rainforest are expected to:
  - a) Lower winter temperatures
  - b) Increase summer humidity levels through increased snowmelt
  - c) Decrease summer humidity levels through decreased snowmelt
  
3. Lichens in the inland rainforest may be effective biomonitors for climate change because:
  - a) They are not overly abundant and are likely to disappear quickly
  - b) They are easily identified
  - c) They are highly diverse and will show differential responses

