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Defining protected area boundaries based on vascular-plant species richness using hydrological information derived from archived satellite imagery

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ABSTRACT

Surface and near-surface hydrological conditions are not generally considered in the selection of protected-area boundaries; however, these hydrological conditions may be critical in the maintenance of hydrologically driven biodiversity areas and should factor heavily in systematic conservation planning and impact-mitigation efforts. Here we demonstrate the use of Synthetic Aperture Radar (SAR) satellite imagery to model surface and near-surface hydrological dynamics and related vascular-plant species richness surrounding Sundance Provincial Park, Alberta, Canada. Based on a relationship established between field-sampled soil moisture and remote-sensed SAR backscatter coefficient, we generated landscape-scale maps of surface hydrological condition and related vascular-plant species richness surrounding the protected area. After processing, SAR data were classified into three hydrological classes: unsaturated, saturated and inundated. By assembling a multi-year series of SAR images acquired across the natural range of hydrological conditions, the probability of surface saturation and inundation was developed. Using the resulting probability map to select sites for biodiversity assessment, we found relatively strong relationships between total vascular plants (species richness, especially ferns and herbs), and the probability of wet area (both saturated and inundated areas) occurrence. Using this relationship, we mapped vascular-plant species richness across the study region revealing opportunities to mitigate impacts from adjacent developments and to expand the park to increase its ecological value. This study presents an important approach for assessing biodiversity of existing parks and generating valuable input layers that help to define ecologically relevant protected-area boundaries.

1. Introduction

Systematic conservation planning requires relevant information with which to set landscape targets for species, vegetation types or other features (Margules and Pressey, 2000; Pierce et al., 2005; Rondinini et al., 2006; Knight and Cowling, 2007; Linke et al., 2011). Protected areas are often the legacy of such exercises, so ideally targets used in conservation planning include those relevant to ecological processes (e.g., migration) and patterns (e.g., biodiversity) mapped at either local or landscape scales. Creating such maps particularly at landscape scales is a challenge, especially within limited conservation budgets. Consequently, researchers have explored the use of remote sensing to generate cost-effective predictive maps of biodiversity (Nagendra, 2001; Gillispie et al., 2008; Wang et al., 2010), which subsequently may form the basis of decision-making regarding conservation within the area of interest.

Most remote sensing studies focusing on terrestrial-based conservation planning typically use traditional land-based metrics such as fragmentation, connectivity, or industrial footprint derived from image classification based land-use/land cover maps (e.g., Eyre et al., 2003; Seto et al., 2004; Rouget et al., 2006). These studies are based on the assumption that habitat size and quality are reasonable proxies of biodiversity, most often regarding species richness. More recently, image spectral characteristics of the surface (vegetation or soils) (e.g., Rocchini et al., 2010) and vertical structural information from laser altimetry measurements (e.g., Lucas et al., 2010) were used directly to predict metrics of biodiversity such as species richness. Other advanced techniques use hyperspectral imagery to detect upper-canopy pigments, water and nitrogen content that have been found to explain patterns in species richness in Hawaiian rainforests (Carlson et al., 2007). Fewer studies have explored other fundamental environmental determinants of biodiversity such as moisture regimes (but see Parks and Mulligan, 2010).

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Water is one of the basic environmental demands of plants along with other abiotic factors such as temperature, light, oxygen and nutrients (Witte et al., 2004). Water is used directly by plants for growth and maintenance, but it is also essential in driving nutrient and energy cycling through ecosystems. Each plant has different environmental requirements encapsulated as part of their environmental site conditions. Knowledge of the environmental site conditions of a plant provides the ability to both use the presence of a certain plant an indicative of environmental conditions and vice versa, environmental conditions can suggest the type of plants to be expected in a given area. However, determining environmental site conditions of plants is time consuming, and thus more general methods are needed that can be used to establish links between environmental conditions and biodiversity.

Therefore, understanding the distribution and dynamics of water across landscapes is one of the key steps in understanding the distribution of plants. Hydrological systems are temporally dynamic and span spatially complex (longitudinal, lateral, and vertical) dimensions (Ward, 1989; Thorp et al., 2006), making them arguably more complex than other environmental processes such as energy cycling. The multi-dimensionality of hydrological systems makes them more easily and directly affected by anthropogenic influences. These influences can readily travel across boundaries of conservation areas and negatively impact the ecology of these sensitive areas. For example, an intact hydrological system is crucial to the long-term function of protected areas defined by or dominated by streams, lakes and/or wetlands. Unfortunately, protection of eco-hydrological systems is generally limited to fixed-width riparian buffers around larger streams and lakes (Buttle, 2002). Smaller and more ephemeral hydrological features such as wetlands and ephemeral draws that feed into larger water-bodies and thus serve as critical source areas are usually not protected (Devito et al., 2000; Creed et al., 2008b).

While the link between water availability and biodiversity has been known for a long time, relatively little work has been done exploring the relevance of hydrological mapping in conservation planning, specifically linking surface hydrological conditions of terrestrial ecosystems to species richness (but see Ranganathan et al., 2007). In a conservation planning context, hydrological mapping is typically limited to delineating hydrological features such as lakes, rivers and larger permanent wetlands via photo-interpretation of aerial photographs at coarse spatial scales (1:65,000–1:85,000) (e.g., Canada's National HydrologicalNetwork and National Topographic Data Base). This snapshot approach not only excludes the long-term hydrological dynamics of these features, but it also misses fine-scale features such as cryptic wetlands and rivulets because these optical sensors cannot penetrate the canopy (Bishop et al., 2008). Moreover, the hydrological status of uplands is not captured at all.

Landscape-scale hydrological mapping can be vastly improved by analyzing the movement of water across the land surface using digital elevation models (DEMs). Digital terrain analysis of fine spatial resolution (<5 m) DEMs derived using laser altimetry is highly effective at determining small or canopy covered features like headwater wetlands and permanent or ephemeral streams, and it is not dependent on manual assessment of hydrological proxies (i.e., methods that may introduce errors or compound uncertainty) (Lindsay et al., 2004; Creed et al., 2008a; Creed and Beall, 2009; Jenkins and Frazier, 2010; Creed and Sass, 2011). However, in landscapes where the position of the water table is influenced by geological fractures and surficial geology as well as topography – such as the sub-humid Boreal Plains of Alberta, Canada – topographic methods may fail to detect wet areas (Devito et al., 2005). Furthermore, topographic methods suffer from the same static limitations as photo-interpretation methods by failing to capture the spatial and temporal dynamics of hydrological features on the landscape, whose extent fluctuates with climatic variability.

Remote sensing analysis of Synthetic Aperture Radar (SAR) images offers a superior alternative to both photo-interpretation and digital terrain approaches for the mapping of forested wet areas (saturated or inundated areas) for the following reasons: (1) the microwaves transmitted by the SAR sensors have the potential to penetrate vegetation; (2) they are sensitive to soil moisture content at the top of the soil column; and (3) the acquired SAR images offer spatially and temporally extensive datasets that can capture hydro-dynamics across multiple watersheds and over multiple decades (>20 years imagery are available). SAR images record radar backscatter intensity, which is controlled by both dielectric and structural properties of the surface related to soil moisture, surface roughness, vegetation, and topography (Kasischke et al., 1997). In addition, SAR sensors offer the advantage of all-weather and day–night acquisition.

Recent papers have documented the utility of SAR systems at deriving hydrologically relevant information from archived imagery in forested landscapes (e.g., Creed et al., 2008b; Lang and Kasischke, 2008; Sass and Creed, 2008; Clark et al., 2009). The power of these studies lies in the fact that they use multiple images capturing the same area over many time periods that represent different hydrological conditions. As in time-lapse photography, the object of the imaging "comes alive"; in our case the expansion and contraction of surface hydrological features is captured. For example, Sass and Creed (2008) used a 15-year period to analyze between-year variability in wet areas (a combination of saturated and inundated parts of the landscape) derived from ERS-1 and ERS-2 satellite imagery. They found that ERS derived wet area dynamics were closely correlated to climatic forcing in the boreal forest dominated by an open-canopy wetland environment.

The goal of this paper was to explore the links between surface-hydrological dynamics and vascular-plant species richness using remote sensing methods. We applied the wet-area mapping technique of Sass and Creed (2008) to capture spatial and temporal hydrological dynamics of our study area and use these data to map vascular-plant species richness based on field collections. The resulting species richness map is thus based on a key ecological process, long-term soil moisture condition mapped by satellite-based sensors.

Specifically, we addressed the following objectives: (1) map the probability of wet area formation using SAR-based imagery collected over a representative range of the natural variation in climate; (2) assess vascular-plant species richness along the mapped wetness gradient; (3) generate predictive models linking wetness and vascular-plant species richness; and (4) map and assess vascular-plant species richness as a component of protected-areas management and selection. We demonstrate this eco-hydrological approach in the region of Sundance Provincial Park located in west-central Alberta, Canada. This region is a relevant study site given that there are examples of hydrologically driven biodiversity hotspots, and there is a strong desire to conserve the area's biological diversity given the high pace of industrial development in the region.

2. Materials and methods

A large multi-year SAR image dataset was used to map the probability of surface saturation and inundation for Sundance Provincial Park and the surrounding landscape following the techniques described in Clark et al. (2009) and Sass and Creed (2008). These methods emphasized the selection of images along the full range of the hydro-climatic conditions, the classification of geometrically and radiometrically rectified individual images into three hydrological classes (unsaturated, saturated, inundated), and the combination of individual images into a probability-of-wetness map after combining the saturated and inundated classes into one 'wet area' class. The probability-of-wetness map was then used as a basis for the selection of sites for the assessment of related biodiversity.

2.1. Study region description

The study region is located approximately 20 km west of the Town of Edson, Alberta, Canada (53.57°N, 116.45°W), encompassing the Sundance Creek watershed. Sundance Creek watershed lies in the western area of the Boreal Plain Ecozone, spanning the Lower and Upper Foothills Natural Subregions (Natural Regions Committee, 2006) and drains a 250 km^2 area characterized by moderate relief from 930 m to 1400 m above sea level into the Athabasca River basin (Fig. 1). Sundance Creek drains three small lakes in its headwaters, flows along a deeply incised valley with common occurrence of freshwater springs that form small waterfalls, and drains wetland complexes en route, including a florally diverse marl fen.

Vegetation is composed of a mix of coniferous, deciduous, and mixed-wood forests ranging in age between recently harvested to older forests (≈150 years). Upland sites are composed of trembling aspen (*Populus tremuloides*), lodgepole pine (*Pinus contorta*), and white spruce (*Picea glauca*) stands (including mixed-wood stands of these species), with more lowland, wet sites composed of larch tamarack (*Larix laricina*) and black spruce (*Picea mariana*) dominated fens and bogs. Detailed descriptions of the forest community types and understory compositions for these stands are described elsewhere (Beckingham and Archibald, 1996; Wheatley et al., 2002, 2005; Wheatley, 2007a, 2007b). The major natural disturbance in this area is fire, but currently this is dwarfed by the relatively extensive anthropogenic disturbances of commercial forest harvesting, seismic-based gas exploration lines, oil-drilling pads, and roads.

Sundance Provincial Park was created as part of the Special Places 2000 initiative (Government of Alberta, 1995) that added 81 new parks (including Sundance Provincial Park) and expanded 13 others, adding approximately two-million hectares to Alberta's protected area land base (Fig. 1). The park is 37 km^2 and is elongated following the flow-path of its primary water feature, Sundance Creek. Soon after its establishment, it became evident that ecologically-based management of the park would require consideration of the park's hydrological features and their connections to the adjacent industrial lands and Sundance Creek, a requirement that is articulated in the park's current management plan (Government of Alberta, 2006). Along its length, the park's width varies from 400 m at its narrowest to 1.6 km at its widest. Due to this long, narrow shape, the park's perimeter-to-area ratio is relatively large, making it vulnerable to a myriad of potential effects from adjacent development, primarily gas extraction and timber harvesting. As such, there is a regionally unique 500 m special management zone established around the park (Fig. 1), within which development activities are carefully monitored or mitigated, such that a fixed-width transition zone exists between the working landscape and the provincial park.

2.2. Image selection

Altogether, 29 archived ERS-1 and ERS-2 images were acquired during the snow-free period from May to September from the Alaska Satellite Facility (ASF) spanning the years from 1994 to 2004. The satellite images were selected to reflect the natural range of hydrological conditions in the park as characterized by daily stream discharge over a 30-year period (1979–2008) measured at

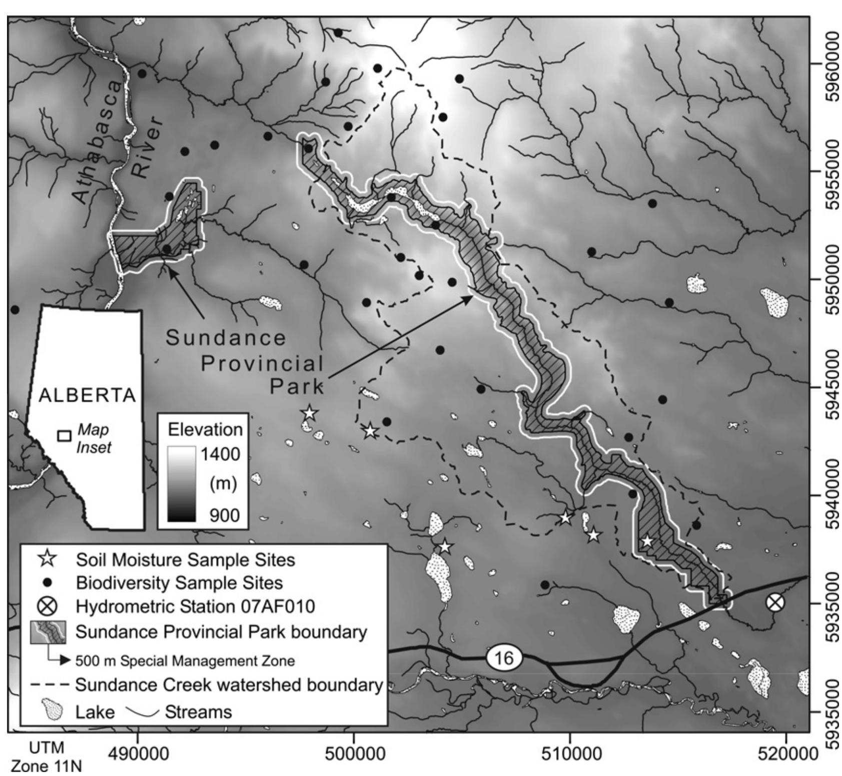

Fig. 1. Study area showing location of Sundance Provincial Park, the 500-m special management zone and soil moisture and biodiversity sampling sites. The boundary of the topographically defined Sundance Creek watershed is shown as extracted from an outlet matching the southernmost point of Sundance Provincial Park. The background layer is topography.

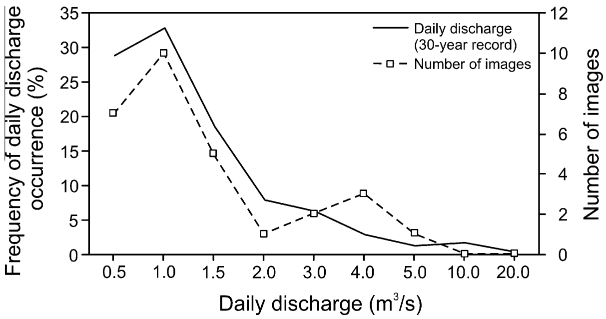

Fig. 2. Frequency distributions of daily discharge measurement as represented by daily discharge data from a 30-year record at Sundance Creek (#07AF010) and by daily discharge data recorded the day of the overpass for the 29 ERS images processed for this study.

a Water Survey of Canada hydrometric station (#07AF010) located on Sundance Creek just downstream of the park boundary (Fig. 1). The SAR images captured the natural range of hydrological conditions; only exceptionally high daily discharges were not sampled by the imagery (Fig. 2). No significant difference was observed between the distribution of daily discharge taken over a 30-year period and the distribution of daily discharge observed on the days that satellite images were acquired based on a Kolmogorov–Smirnov test of distributions ($Z = 0.72$, $p = 0.69$). Statistical analysis was conducted using SigmaStat 2.03 (SPSS Inc.).

2.3. Image pre-processing

ERS images were radiometrically converted to backscatter coefficient and corrected for systematic offsets using ASF MapReady 2.0 software (ASF, 2009). Radiometrically standardized images were corrected concurrently for layover and foreshortening terrain effects and orthorectified using a Canadian Digital Elevation Data (CDED) – DEM resampled from 0.75 arc sec resolution to 25 m grid resolution. During orthorectification, the images were resampled from 12.5 m to 25 m pixel spacing using a bilinear resampling algorithm to average some of the inherent variability of radar backscatter. Georegistration was refined using an additional 30–40 ground control points collected from georeferenced hydrography and road network layers. A 3×3 gamma filter was applied in two iterations to further reduce image speckle while maintaining linear features in the scenes (Lopes et al., 1993). Image analyses were conducted using Geomatica 10.2 (PCI Geomatics).

2.4. Image-based mapping of hydrological boundaries

In order to assess the sensitivity of SAR images to changes in volumetric soil moisture content (VSM), ground measurements of VSM were taken coincident with overpass of one ERS-2 image collected in 2008 and processed according to the methods described in Section 2.3. VSM was sampled using a theta probe (Delta-T Devices, 1999) at pre-selected sites within a large area ($\geqslant$1 ha) of homogeneous vegetation cover to minimize the problem of mixed pixels and speckle (Griffiths and Wooding, 1996). At each site, soil impedance was measured using the theta probe across a 50 m^2 sampling grid where sampling nodes were separated by 10 m. At each node, theta probe readings were made in each of the cardinal directions, with a set of four readings in non-wetland sites and two sets of four readings (one in a hollow and one in a hummock) in wetland sites. Soil impedance was converted to VSM by an exponential regression model provided by Sass and Creed (2008). For each ground based sampling site, homogeneous polygons (in terms of vegetation cover) with a minimum area of 1 ha were defined over which the backscatter coefficient was extracted. These polygons were hand-digitized on the computer screen with the aid of geographic coordinates of sampling sites and a vegetation cover map derived from a recent LANDSAT TM image. The aim was to define the largest possible homogenous polygon for each site to reduce the noise introduced by mixed pixels and speckle (Griffiths and Wooding, 1996). An average backscatter coefficient for each polygon was then calculated. A regression model was generated to relate ground-based VSM measurements to SAR backscatter coefficients.

SAR-based prediction of VSM as a continuous variable is prone to uncertainty due to unwanted signal introduced by topography, surface roughness, and vegetation cover (Sass and Creed, 2008). In addition, inherent random signal noise (speckle) and signal variability in mixed pixels resulting from the close proximities of hydrological features and gradients in a heterogeneous environment create further uncertainty in the interpretation of the backscatter signal. For these reasons, images were classified into three hydrologically relevant classes representing unsaturated, saturated and inundated conditions. Fuzzy classification was used to account for the uncertainty in selecting class boundaries (cf., Clark et al., 2009).

Pixels for each SAR image were assigned membership to each hydrological class (i.e., unsaturated, saturated and inundated) using sigmoidal membership functions based on the assumption that uncertainty of pixel values are normally distributed in the transition zones. Control points representing membership likelihood equal to 0.5 for each hydrological class and the width of the transition zone between hydrological classes were estimated either from the regression model in the case of unsaturated and saturated classes or from transects digitized on SAR images crossing zones of inundation. Permanent open water features (with a 25 m interior buffer to account for shoreline variations) were overlaid on the classified results to negate the impact of wind-induced backscatter increase over inundated areas. After classification of each image into membership probabilities of the three classes, a single crisp classification was created using a fuzzy combine operation. The images were converted into a binary classification by combining the saturated with the inundated classes (herein referred to as wet area class). The long-term probability of wet area occurrence was computed by averaging all 29 images for each individual pixel (Creed et al., 2008b).

2.5. Vascular-plant biodiversity assessment

2.5.1. Site selection

Vegetation sampling sites were selected along the full range of wetness as defined by the long-term probability of wet area occurence map. We stratified the probability map into three classes: dry (probability 0–0.33), mesic (probability 0.33–0.66) and wet (probability 0.66–1). Within each of these three classes and irrespective of dominant forest cover, we randomly selected 11 vegetation-sampling sites in areas that had not been disturbed by human activity, for a total of 33 plots. All sampling plots occurred within mature, pyrogenic forests >70 years of age (or since disturbance) (Fig. 1). Three of the sites were excluded, as the dense canopy cover in these sites prevented sampling according to the pre-selected protocols.

2.5.2. Vegetation sampling

We used existing vascular-plant and tree-sampling protocols developed by the Alberta Biodiversity Monitoring Institute to quantify species richness. At each site, we established two 50 m $\times$ 50 m plots, one directed NE and one directed SW of the site's UTM location. Within each 50 m $\times$ 50 m plot, we conducted one-person 30-min vascular-plant searches to determine the presence of as many species as possible. For each 30-min search, an individual observer spent the first 10 min at plot center recording all observable vascular plant species, then spent 20 min systematically walking throughout the plot recording the presence of as many vascular plants as possible. To maintain consistency among observers, we started the searches at the site's UTM location (site center, and the inner corner of each 50 m × 50 m plot), then moved to within 5 m of the plot's center and searched in a clockwise direction around the plot staying approximately 5–10 m from the plot's edge. When unknown species were encountered, they were collected as voucher specimens for identification after the 30-min search was completed. This ensured that the 20 min in each plot were spent looking for species rather than identifying plants. Voucher specimens were identified in a laboratory setting by experts when they could not be identified in the field. Species lists for both 50 m × 50 m plots were pooled for each sampling site.

2.6. Data analysis

For each vegetation sampling site, we generated metrics of wetness and species richness and built linear regression models to explore the correlation and the predictive potential of biodiversity from wetness. We generated different species richness metrics by grouping vegetation data into species counts for the following groups: total vascular species, total tree species, total shrub species, total herb species, total graminoid species, total ferns and fern allies, and the Shannon Diversity Index for each vegetation site. These metrics were then plotted against the averaged probability of wetness derived from an area of 100 m × 100 m centered on each sampling site. For the vascular plant model, significantly different classes of species richness (i.e., low, moderate and high) corresponding to the different probability classes (dry, mesic, wet) were identified and then used to delineate a biodiversity-based management map.

3. Results

As a first step in mapping the distribution of wet areas within the landscape surrounding Sundance Provincial Park, we examined the relationship between volumetric soil moisture (VSM) and backscatter coefficient. We observed a positive relationship between VSM and backscatter coefficient; a linear regression model explained 64% of the variation in backscatter coefficient ($p = 0.055$; $SE = 0.69$ $\ln[\text{m}^3/\text{m}^3]$) (Fig. 3). Due to limitations of weather (it rained the day of and day after the image acquisition days) we were able to use only six ground-truth sites in our linear regression model. We compared this model to a linear regression model published by Sass and Creed (2008) based on data collected

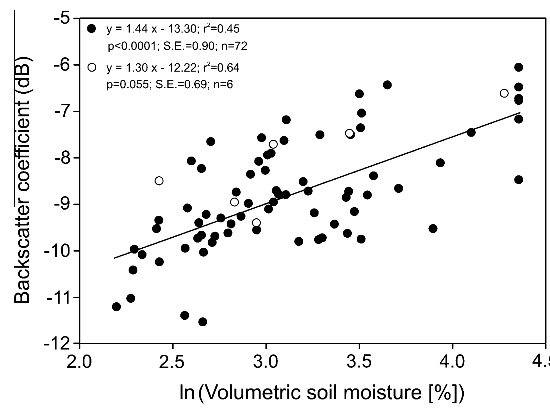

Fig. 3. Relationship between volumetric soil moisture (VSM) and backscatter coefficient (dB) for this study (open circles, dashed line) and for Sass and Creed (2008) (solid circles, solid line), both studies conducted in northern Alberta. ANCOVA showed no statistical difference between the slopes or intercepts of the two models.

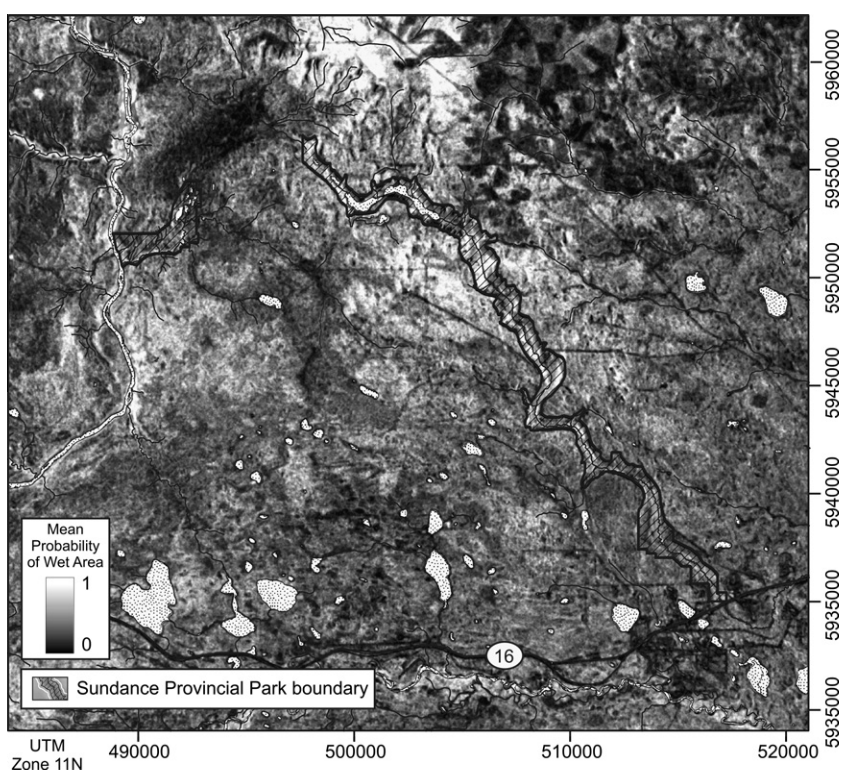

Fig. 4. Spatial distribution of long-term mean probability of wet areas (saturated and inundated areas).

Table 1
Linear regression models for predicting species richness from wetness. Note: WET = probability of wet area occurrence.

Model	r^2	*SE*	Model equation	*P*
# of herb spp.	0.74	5.48	HERB = 17.60 + (30.04 ∗ WET)	<0.001
# of fern spp.	0.49	3.02	FERN = 1.39 + (9.54 ∗ WET)	<0.001
# of graminoid spp.	0.00	3.23	GRAM = 4.76 + (0.91 ∗ WET)	0.646
# of shrub spp.	0.12	4.94	SHRUB = 24.73 + (5.75 ∗ WET)	0.066
# of tree spp.	0.02	1.39	TREE = 5.83 − (0.67 ∗ WET)	0.435
# of vascular spp.	0.70	9.41	VASC = 54.47 + (46.03 ∗ WET)	<0.001
# of fern and herb spp.	0.80	6.11	FERN + HERB = 18.99 + (39.58 ∗ WET)	<0.001
Shannon diversity	0.52	0.08	SHANNON = 0.52 + (0.265 ∗ WET)	<0.001

in a similar landscape about 200 km north of Sundance Provincial Park. An ANCOVA confirmed similarity between the regression models: both the slope and intercept were not statistically different ($p = 0.83$ for slope and $p = 0.07$ for intercept), suggesting that the two landscapes behaved similarly in terms of their interactions with microwave radiation. The standard error ($SE = 0.69$ ln[m^3/m^3]) of the linear regression model makes the predictive power low if used to predict VSM on a continuous scale. After applying the hydrological classification to the 29 individual images and combining them into a wet-area probability map, the pattern of the long-term mean condition for the distribution of wet areas (saturation and inundation combined) emerged across the study landscape (Fig. 4). The map shows distinctive hydrological patterns where wet areas occur along riverine corridors as well as around many of the lakes of the region.

One hundred eighty-nine (189) vascular plant species were recorded from the study area, of which one is non-native and two are considered to be rare (Alberta Conservation Information Management System, 2011). All species found are typical of mesic habitats in the boreal forest of Alberta. We observed statistically significant positive relationships between probability of wet area occurrence and vascular-plant species richness; simple relationships amenable to most management contexts (Table 1). When separated based on broad vegetation types, the total number of herb ($r^2 = 0.74$, $p < 0.001$, $SE = 5.5$) and total number of fern species ($r^2 = 0.49$, $p < 0.001$, $SE = 3.0$) were most successfully predicted by the metric of wetness. Combining herbs and ferns increased the variance explained to 80% ($p < 0.001$, $SE = 6.0$). Probability of wet area occurrence was also successful in explaining the variance in the more inclusive group of total vascular species ($r^2 = 0.70$, $p < 0.001$, $SE = 9.37$) (Fig. 5A).

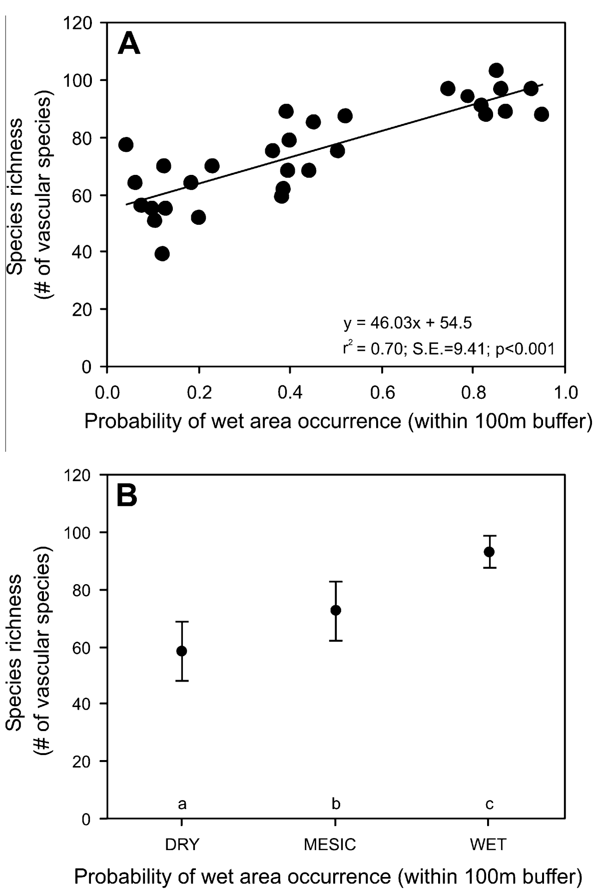

Fig. 5. Modelling species richness from wetness: (A) linear regression model where total number of vascular species is regressed on probability of wet area occurrence and (B) simplified boxplot model of *A* showing mean and standard deviation of total number of vascular species classified by wetness.

We used this more general model to spatially predict vascular-plant species richness. The standard error of the model suggested that there were no more than three separable classes and therefore our final map is classified into low, medium and high vascular-plant species richness classes. This separation was further supported by the results of a one-way ANOVA, which showed statistical differences between vascular-plant species richness when grouped into three wetness classes (all pairwise Tukey comparisons significant at $p < 0.05$) (Fig. 5B).

4. Discussion

In this study, we were able to map a metric of wetness at a regional scale, assess vascular-plant species richness in this area, and then generate models to link wetness to species richness, which can be incorporated into the selection and management of protected-areas.

4.1. Mapping the probability of wet area occurrence at regional scales

Comparing the regression model generated in this study with that of Sass and Creed (2008) justifies the prediction of two hydrologically meaningful classes: saturated and unsaturated. As demonstrated in Sass and Creed (2008), this simple hydrological classification produced an accuracy of 88% when tested across 16 sites of varying vegetation cover. The high standard error of the regression models was not surprising given that the imagery were not corrected for vegetation, surface roughness or topographic effects. Correcting for such effects at the landscape scale is difficult given the lack of spatially distributed data of these surface properties. We argue that the use of uncorrected imagery is still useful to map regional patterns that can inform land use planning, which otherwise would have no such evidence-based hydrological information.

It is interesting to observe that the middle and upper part of Sundance Provincial Park show the largest contiguous zones of wet areas. We suspect this is due to the many springs found in the area (Vogwill, 1983). Whereas, the deeper geological materials are fractured, creating the conditions for the many springs, the surficial materials have low permeability, and as a result large saturated areas form surrounding the spring heads (Vogwill, 1983). Another interesting feature of the probability map is that wet areas, like some of the lakes, may straddle topographic divides. This is not unusual for this relatively low-relief area with deep, fractured geologic deposits that have the ability to route water through subsurface pathways to topographic highs where it may flow across gentle topographic gradients on these upland plateaus (Sass and Creed, 2008).

Vegetation, topography and surface roughness also partially influence the backscatter coefficient from SAR; however, compared to soil moisture, their influence is fairly constant between images, given that the images are chosen in the same season (leaf-off or leaf-on). While open canopies allow the penetration of microwaves, closed canopy forests with high biomass significantly reduce the ability to detect soil moisture. The effect of vegetation on the backscatter coefficient becomes important at around 1.0 kg m^{-2} (Dobson et al., 1992); however, significant correlation between soil moisture and backscatter has been found in forests with much higher biomass (Harrell et al., 1995; Lang and Kasischke, 2008). In fact, Lang and Kasischke (2008) reported that soil moisture was a statistical significant predictor of backscatter under both leaf-off and leaf-on conditions and using either VV or HH polarization in upland and wetland forests of Maryland's Coastal Plain. Much of the topographic effects are ameliorated by the fact that the ERS images were taken from both ascending and descending directions, thereby canceling out the topographic effects. Given that the focus of this study is the delineation of hydrologically relevant ecological boundaries at regional scales, local topography effects will have minimal impact on the mapping.

4.2. Biodiversity as a function of wetness

Given that hydrological flow is a major determinant of physical habitat in streams and wetlands (Bunn and Arthington, 2002), the relationships between species richness and wetness are not entirely surprising. The high coefficient of determination and reasonable standard errors made it possible to apply the linear regression models and generate spatial biodiversity maps (Fig. 6). Because we found similar backscatter–moisture relationships between two disparate study areas (Fig. 3), we would feel confident applying these models throughout the foothills natural region, particularly where general forest-community types are relatively similar. Extrapolations beyond this would require further field calibration; however, each calibration would conceivably generate a species richness map covering several thousand square kilometers, at least in Alberta. The simple nature of the relationship lends itself well to application in a resource management where conservation decisions need to be made over large areas with limited resources.

4.3. Integrating hydrological information into protected areas management

How are these results relevant to protected areas management? We envision four main applications of this work in a management framework.

4.3.1. Evidence-based tool

The hydrologically-derived biodiversity map provides an accessible evidence-based tool to assist in mitigating impacts to ecological integrity for long-term park management. Conservation trade-offs are always costly. Without science-based tools to quantify and prioritize trade-offs, it is difficult (if not impossible) to effect any sort of positive landscape change or mitigate potential impacts to protected areas, especially if scientific results remain in overly technical form and change implies a cost to industry. Arguably, maps are the most powerful and easiest to interpret of these tools. They conceptualize the ecological science into something usable by anyone, especially non-ecologists. Land managers do not need to

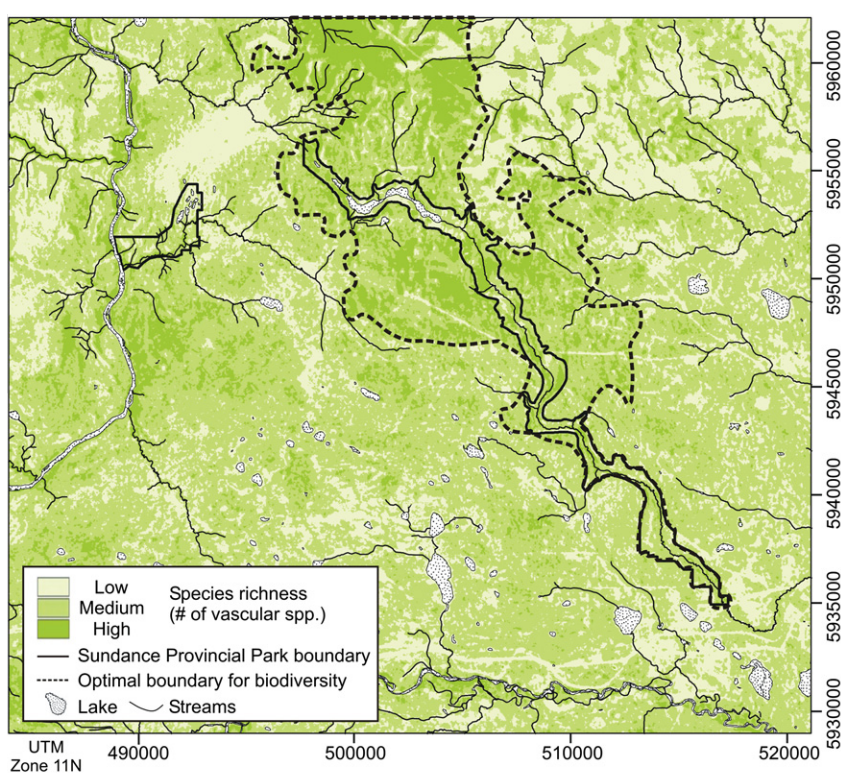

Fig. 6. Map of species richness as modeled from linear regression model depicted in Fig. 5A. Overlain on map is suggested optimal park boundary based on protection of high biodiversity areas.

understand the science of remote sensing and its relationship to biodiversity to use and understand these results (i.e., maps). As such, these results were used (successfully) in recent forest-clearing negotiations with both logging and gas companies operating in proximity to Sundance. If the science is digestible, companies are more willing to attempt impact mitigation. For example, the lower reaches of the Sundance watershed do not have large contiguous wet areas; here the administrative park boundary appears largely sufficient. Thus, conservation focus can be shifted further north in the park. Such maps expedite the processes both of identifying and mitigating potential impacts.

4.3.2. Trans-boundary, systems thinking

The hydrological map clearly highlights the trans-boundary nature of hydro-ecological processes. In a conservation context this cannot be overemphasized to (sometimes non-technical) land managers. Our map reiterates how hydro-ecological processes relate to anthropogenic land partitioning and how they are not always contained within topographical or watershed units. There is a long history of protecting areas based on emotion, popularity, or political agenda, reasons that cannot entirely be discarded as relevant for protecting spiritual sites or traditional-use areas where people can reconnect with nature in some form. However, the long-term management of such areas becomes extremely challenging because trans-boundary hydro-ecological processes are constantly effecting change within the park (e.g., Weaver, 2006; Gaston et al., 2008), yet rarely are these processes clearly identified or understood. There are not only trans-boundary movements of flora and fauna (Weaver, 2006), including movements along corridors to nearby parks or from adjacent vegetated fragments (Zuidema et al., 1996; Armenteras et al., 2003), but there are also matter and energy inputs from outside that are constantly changing due to natural and anthropogenic factors.

Additionally, the hydrological map is based on relatively long-term data from archived imagery, ensuring that it is not considered an ecological snapshot or biased by outlier data. It forces park managers to broaden their perspective from the parochial to the landscape and view the park as a functioning component of a broader system. This systems approach will become keystone for the perpetuity of our parks: our results provide a clear example of why parks are not landscape isolates (Reyers et al., 2010).

4.3.3. Assessing ecological value of parks

Many protected areas are considered environmental benchmarks that indicate reference conditions for undisturbed ecological systems (Arcese and Sinclair, 1997). However, most often, establishing park boundaries is a policy-driven approach, based largely on existing administrative or political borders with little consideration to the ecological value of the resulting area (Svancara et al., 2005). Because of the costs involved in data acquisition, the relative ecological integrity and biological importance (e.g., relative species richness, ecological processes, etc.) of parks are rarely assessed (but see Weaver, 2006), and so a quantified ecological value cannot be placed on most protected areas; this is especially true for smaller regional parks below the spatial extent of most ecological processes. Where efforts have been made to address ecological function, the focus largely has been on terrestrial features such as forest fragmentation (Page et al., 1997; Armenteras et al., 2003) or forested riparian buffer zones (Hannon et al., 2002), all of which require costly a priori ecological assessments to be effective or to understand their potential effectiveness (cf., Parrish et al., 2003).

Our hydrologically-derived biodiversity map can be used to provide a quick assessment of ecological value across large regions. This could be useful for providing a relative "park value" within the larger region or be used to rank parks across a larger jurisdiction such as a province or state. For illustration, based on our diversity mapping, the average vascular-plant species richness of Sundance Provincial Park is 78 species of vascular plants, which is only slightly higher than the average for the surrounding landscape (75 species of vascular plants). Managers can assess numbers like these to understand the relative importance of existing protected areas on the landscape, and inform whether there are sufficient landscape-level set-asides to conserve ecological goods and services (e.g., biodiversity), or whether a landscape is conservation-impoverished. These are powerful tools to bring to round-table stakeholder discussions or government-based land use planning sessions.

4.3.4. Delineation of park boundaries

When faced with opportunities to expand or choose new protected areas, the conservation of both ecological integrity and species richness (biodiversity) should factor high in the site-selection process. The modeled vascular-plant species richness map provides a basis upon which to assess and (given the opportunity) alter an existing park boundary to one more ecologically grounded, based on an integrated hydrological and biodiversity perspective (Fig. 6). For example, in the case of Sundance Provincial Park, the vascular-plant species richness map suggests that increasing the boundaries in the upper reaches of the Sundance watershed would greatly add ecological value to the park.

But perhaps more importantly, our methods generate a valuable data-input layer to be used in concert with other input layers in spatially explicit, GIS-based, systematic conservation planning (sensu Margules and Pressey, 2000). In this context, our maps of species richness can be combined with other data inputs such as rare-species locations and natural-resource cost layers, amongst other factors (e.g., Knight et al., 2006; Ban and Klein, 2009; Rondinini and Chiozza, 2010). Then, through spatial annealing simulations, trade-offs amongst cost effectiveness, fragmentation, and achieving conservation targets (set formally in each simulation) can be assessed objectively generating multiple options for either new conservation area creation, or existing park expansion. This is perhaps the most effective use of the species-richness maps created herein. It should be noted that in this process, one must decouple extensive map coverages like our species-richness maps from, for example, point locations of rare species or community types, for these are usually treated as two separate inputs with different conservation targets; as such, to maintain the general application of this technique, we do not provide herein a species list outlining rare versus common species (for example). Although, one could do this easily with our data, it would form a different (non-technical) input layer into the planning process influencing area selection to the extent set by the user, and is arguably beyond the objectives of this study. It is our experience that species richness maps are often the missing data layer in these GIS-based simulations.

5. Conclusions

Hydrological data derived from radar satellite imagery can be a powerful tool to assist in protected-areas management. Using Sundance Provincial Park as a case study, we have shown that (1) the backscatter coefficient of radar imagery is sensitive to volumetric soil moisture; (2) individual radar images can be classified into a simple hydrological map of unsaturated, saturated and inundated classes, and a series of these types of hydrological maps can be combined into a map of probability of wet area occurrence; (3) indices of vascular-plant species richness are highly correlated to the probability of wet area occurrence; (4) based on these relationships, predictive models of vascular-plant species richness can be mapped; and (5) these maps are effective tools in protected-areas conservation and management, whether used locally or in GIS-based spatial-modeling simulations. Such maps provide an effective tool to assess the relative importance of protected areas within a region, and can inform managers whether or not current parks are sufficient for conservation purposes, or assist modelers develop new conservation strategies.

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Glossary

Backscatter: Microwave radiation reflected back toward a satellite sensor
Digital elevation model (DEM): Records heights above a standard datum (i.e. sea level) of a surface (e.g., the ground surface) on a regular grid
European Remote Sensing (ERS) satellites: Two Earth Observation satellites (ERS-1 and ERS-2) launched during the 1990s by the European Space Agency carrying, amongst other things, an imaging Synthetic Aperture Radar (SAR) instrument
Synthetic Aperture Radar (SAR): Active (generates its own radiation) microwave sensor that uses the flight path of the sensor platform to simulate an aperture much larger than the platform itself, yielding higher resolution radar imagery
Volumetric soil moisture (VSM): A measure of soil moisture by volume (%)