



## Article

# Rapid Ecosystem Change at the Southern Limit of the Canadian Arctic, Torngat Mountains National Park

Emma L. Davis <sup>1,\*</sup>, Andrew J. Trant <sup>1</sup>, Robert G. Way <sup>2</sup>, Luise Hermanutz <sup>3</sup> and Darroch Whitaker <sup>4</sup>

<sup>1</sup> School of Environment, Resources and Sustainability, University of Waterloo, 200 University Avenue West, Waterloo, ON N2L 3G1, Canada; atrant@uwaterloo.ca

<sup>2</sup> Northern Environmental Geoscience Laboratory, Department of Geography and Planning, Queen's University, Kingston, ON K7L 3N6, Canada; robert.way@queensu.ca

<sup>3</sup> Department of Biology, Memorial University, St. John's, NL A1B 3X9, Canada; lhermanu@mun.ca

<sup>4</sup> Western Newfoundland and Labrador Field Unit, Parks Canada, Rocky Harbour, NL A0K 4N0, Canada; darroch.whitaker@canada.ca

\* Correspondence: emma.davis@uwaterloo.ca or emmalaureldavis@gmail.com

**Abstract:** Northern protected areas guard against habitat and species loss but are themselves highly vulnerable to environmental change due to their fixed spatial boundaries. In the low Arctic, Torngat Mountains National Park (TMNP) of Canada, widespread greening has recently occurred alongside warming temperatures and regional declines in caribou. Little is known, however, about how biophysical controls mediate plant responses to climate warming, and available observational data are limited in temporal and spatial scope. In this study, we investigated the drivers of land cover change for the 9700 km<sup>2</sup> extent of the park using satellite remote sensing and geostatistical modelling. Random forest classification was used to hindcast and simulate land cover change for four different land cover types from 1985 to 2019 with topographic and surface reflectance imagery (Landsat archive). The resulting land cover maps, in addition to topographic and biotic variables, were then used to predict where future shrub expansion is likely to occur using a binomial regression framework. Land cover hindcasts showed a 235% increase in shrub and a 105% increase in wet vegetation cover from 1985/89 to 2015/19. Shrub cover was highly persistent and displaced wet vegetation in southern, low-elevation areas, whereas wet vegetation expanded to formerly dry, mid-elevations. The predictive model identified both biotic (initial cover class, number of surrounding shrub neighbors), and topographic variables (elevation, latitude, and distance to the coast) as strong predictors of future shrub expansion. A further 51% increase in shrub cover is expected by 2039/43 relative to 2014 reference data. Establishing long-term monitoring plots within TMNP in areas where rapid vegetation change is predicted to occur will help to validate remote sensing observations and will improve our understanding of the consequences of change for biotic and abiotic components of the tundra ecosystem, including important cultural keystone species.

**Keywords:** tundra vegetation; remote sensing; climate change; shrubification; land cover; tundra ecotone; eastern Arctic



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## 1. Introduction

Arctic plant communities are experiencing significant modification due to recent climate change [1,2], which is altering the composition, distribution, and productivity of many tundra plants [3–5]. Because of the close ties between tundra vegetation and numerous biological [6,7], geophysical [8–10], and cultural systems [11,12], studying changes in plant community dynamics across a range of Arctic environments is critical for predicting future impacts of environmental change. Identifying the drivers of past changes in plant communities is also essential for guiding long-term monitoring, habitat vulnerability assessments, and the management of cultural keystone species in northern protected areas.

Perhaps the most striking recent change in Arctic plant communities has been the increase in shrub dominance that has occurred in many tundra landscapes. Shrubification, referring to increases in the abundance, cover, and biomass of shrubs [13], has been reported through Inuit knowledge sharing [14,15], remote sensing studies [16–18], and field-based observations [19,20]. Warming air temperatures can be viewed as a top-down driver of shrubification [13,21], with mediating factors such as herbivory, soil moisture, nutrient availability, and landscape position contributing to variability in the response of plant communities [21–24]. Whereas shrubification is an obvious indicator of the realized impacts of climate change, shifts in the distribution and abundance of tundra plants that are more visually subtle (i.e., low statured plants and ground cover) can also have wide-reaching effects over large areas. One approach to gaining a temporally and spatially rich perspective on tundra vegetation change is to combine remote sensing techniques with in situ data collection. Doing so can help to overcome our limited ability to understand mechanistic processes from remote sensing imagery [25] and the logistical challenges of conducting extensive field campaigns in remote areas of the North [26].

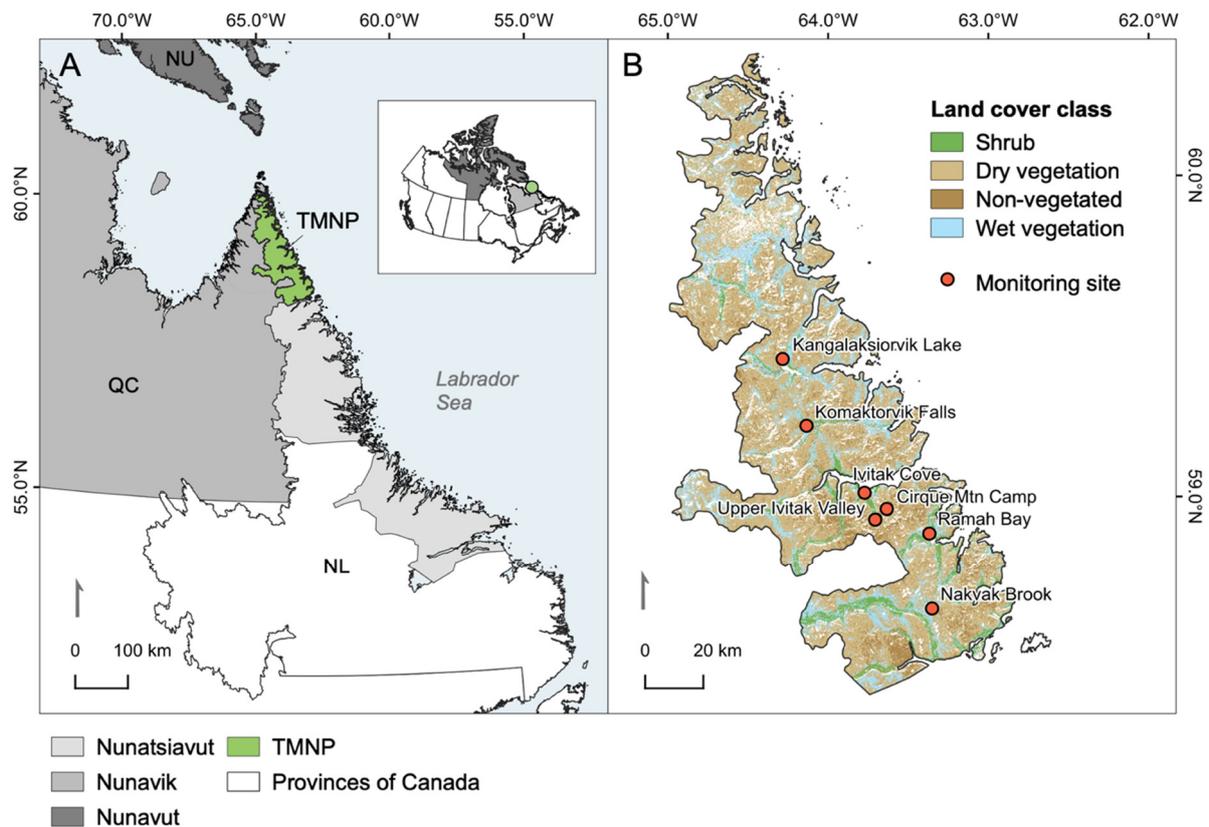
In the eastern Canadian low Arctic, the Torngat Mountains of Nunatsiavut and Nunavik have undergone widespread environmental change in recent years, including changing permafrost [27] and glacier thinning [28,29], declines in caribou populations [30], and increased shrub establishment and growth [16,31]. In 2005, Torngat Mountains National Park Reserve (Tongait KakKasuangita SilakKijapvinga; designated Torngat Mountains National Park in 2008; TMNP) was officially established with the intent of protecting the unique natural history and culture of the region [32]. Northern protected areas, such as TMNP, hold significant ecological value by offering protection against human drivers of habitat and species loss, and are expected to serve as important refuges for plants and animals moving northwards and to higher elevations due the impacts of climate change [33,34]. Due to their fixed spatial boundaries, however, protected areas are themselves highly vulnerable to changing environmental conditions and many will see shifts in biome or land cover types in the coming decades [35,36]. Ecological inventories developed through field-based data collection and remote sensing are an essential tool for managing protected areas and culturally important species, and provide information about baseline conditions against which to measure on-going change [37]. Valuable plot-scale information about vegetation change is available in southern TMNP [31]; however, this field site includes only a small portion of the range of conditions represented across the park's topographically diverse 9700 km<sup>2</sup>, and little is known about contemporary plant community dynamics in central and northern TMNP.

The goal of this research is to extend our knowledge of vegetation dynamics into the past and across a larger spatial expanse to support future research and management activities in TMNP and surrounding areas, and to generate baseline information about critical habitat for Torngat Mountain caribou. The approach used here has wide applicability for monitoring land cover change in Arctic tundra environments elsewhere.

## 2. Methods

### 2.1. Study Area

Torngat Mountains National Park of Canada (TMNP; Tongait KakKasuangita SilakKijapvinga in Inuttitit) is located in an overlapping area of the Nunatsiavut and Nunavik Inuit land claims in northern Labrador (Figure 1). It was established as a National Park Reserve when the Labrador Inuit Land Claim was enacted in 2005 and became a National Park in 2008 when the Nunavik Inuit Land Claim came into effect; the park is intended to represent and protect the Northern Labrador Mountains natural region [32]. Spanning over 9700 km<sup>2</sup>, the park is cooperatively managed with Inuit from Nunatsiavut and Nunavik [38], and the area has been a homeland to modern Inuit and their predecessors for millennia.



**Figure 1.** (A) Location of Torngat Mountains National Park (TMNP) in northern Nunatsiavut and northeastern Nunavik. The green circle on the inset map of Canada shows the location of the park. (B) Land cover classes (2014) and locations of existing temperature monitoring sites in TMNP.

The Torngat Mountains are the southernmost extent of the Arctic Cordillera in Canada. The area is topographically complex and the highest elevations are encountered in the Selamiut Range in the central portion of the park, where the tallest peak, Mount Caubvick, reaches 1652 m a.s.l. The park spans more than 200 km of latitude and relief declines to the north; low elevation areas are characterized by fiords that extend far inland from the coast, deep river valleys, and a complex coastline that includes an archipelago of hundreds of nearshore islands. The regional climate is strongly influenced by the Labrador current [39], which induces foggy and cool maritime conditions throughout the summer. Formation of landfast sea ice along the entire northern Labrador coastline results in cold, continental conditions during winter. The harsh climate and slow soil development have resulted in sparse vegetation cover throughout much of the park. TMNP is situated just north and east of the treeline; however, evergreen and deciduous broadleaf shrubs are common at low elevation on sheltered slopes [40] and along riparian channels [41].

The broader Nunatsiavut and Nunavik regions (Figure 1A) have undergone appreciable environmental change in recent decades. Mean winter and summer temperatures have increased by 2.0 °C/decade and 0.5 °C/decade respectively (1987–2016), and annual temperatures are expected to warm by a further 2–8 °C by the end of the century [42]. Changes in precipitation are difficult to detect, but summer precipitation has generally increased whereas annual snowfall has declined throughout northern Nunatsiavut and Nunavik (1980–2014). In the future, a wetter climate is anticipated as a result of increased rainfall [42]. Climate change in the region has and will continue to have various direct and indirect impacts for permafrost [27], animal and plant populations [41,43,44], and vegetation structure and cover [16,31,45].

## 2.2. General Approach

This study employed remote sensing and statistical modelling to: 1) detect land cover change in TMNP from 1985 to 2019, and 2) predict where shrub expansion is likely to occur in the future (2039/43 time period). Relationships between contemporary vegetation classes and surface reflectance/topographic variables were modelled and applied to past Landsat composites to hindcast land cover changes through time. Spatial and temporal changes in vegetation type and abundance were then analyzed in the context of biogeophysical controls. Finally, the probabilities of pixels transitioning to shrub dominance were forecast for the 2039/43 time period by extending a model of past (1985/89 to 2014) transitions to shrub dominance into the future.

## 2.3. Preparation of Spatial Data

### 2.3.1. Land Cover Map

The reference dataset of contemporary (2014) land cover for TMNP was a 19 class ecotype map (5 m resolution) generated by Ponomarenko and Quirouette [46]. The ecotype map was created from a combination SPOT5 imagery collected in 2008 and 2009, and field validations in 2008. Although the 2008 ecotype map [46] predates the 2014 Landsat composites/indices (described below) by six years, 2014 was the nearest period with sufficient Landsat coverage for modelling due to persistent summer cloud cover. The map was reclassified from 19 ecotypes into four generalized land cover classes including shrub, dry vegetation, non-vegetated, and wet vegetation, with all other ecotypes excluded from analysis (Table 1). Combining ecotypes reduces the level of detail available for different vegetation types; however, the generalized classes were necessary for reducing model complexity in later steps of our analyses. We note that reclassified pixels in the shrub class, which were of particular interest, predominantly represent the low-medium shrub category (58.7% of shrub ecotype pixels; Table 1; including *Salix* spp., *Betula glandulosa*, *Rhododendron tomentosum*), with dwarf shrubs (including *Vaccinium vitis-idaea*, *V. uliginosum*, *Empetrum nigrum*, *Rhododendron lapponicum*, dwarf species of *Salix*, *Betula glandulosa*) and medium-tall shrubs (including *Salix* spp., *Betula glandulosa*, *Alnus alnobetula*) making up smaller proportions of the total (see Table 1). The reclassified ecotype map (hereafter the “land cover raster”) was reprojected (“warp” tool in QGIS) and aggregated (“r.resamp.stats” in GRASS) to match the projection and 30 m resolution of Landsat imagery. The mode of the 36 pixels (5 m res.) within each 30 m grid cell was used during aggregation and the “align” tool (nearest neighbor interpolation) was applied to ensure alignment with corresponding Landsat pixels.

The dry vegetation class was the dominant cover type in the reclassified 2014 land cover raster (61.3% of the total pixels). Wet and non-vegetated pixels were the next most abundant at 18.8% and 13.5%, respectively. The shrub cover class was the least common with 6.5% of the total pixels.

### 2.3.2. Landsat Imagery

Annual composite images of the study area were generated for years 1985 to 2019 from USGS Landsat Surface Reflectance products (Tier 1; Landsat 5 ETM, 7 ETM+, and 8 OLI). Due to the large latitudinal, climatic, and phenological variations across TMNP, it was expected that differences in growing season may alter the peak greening period in southern versus northern portions of the park. Annual duration of peak greening (seasonal changes in the normalized difference vegetation index; NDVI) was thus evaluated for southern, central, and northern TMNP using time series of 250 m resolution Terra MODIS data from 2014 (MOD13Q1.006; AppEEARS web application; [47]). Peak annual greening ranged from three months in the south (approx. June 15 to September 15) to only one month in the north (approx. July 15 to August 15). The period from July 1 to September 1 was therefore retained for acquiring Landsat imagery for land cover hindcasting and model evaluation.

**Table 1.** Reclassification and percent composition of the original 19 ecotypes in four representative land cover classes.

Original Ecotype (Ponomarenko and Quirouette 2014)	Reclassified Land Cover (This Study)	Ecotype Pixels per Land Cover Class
Dwarf shrub	Shrub	28.5%
Low-medium shrub	Shrub	58.7%
Medium-tall shrub	Shrub	12.8%
Herb-lichen tundra	Dry vegetation	21.7%
Mesic <i>Racomitrium lanuginosum</i>	Dry vegetation	23.3%
Rock-lichen	Dry vegetation	55.1%
Alluvial non-vegetated	Non-vegetated	6.0%
Rock	Non-vegetated	90.9%
Sparsely vegetated beach	Non-vegetated	3.0%
Brackish fen	Wet vegetation	0.4%
Fen	Wet vegetation	41.9%
Moist sedge	Wet vegetation	57.7%
Cloud	NA (Masked)	NA (Masked)
Ocean	NA (Masked)	NA (Masked)
Shadow	NA (Masked)	NA (Masked)
Shallow water	NA (Masked)	NA (Masked)
Snow	NA (Masked)	NA (Masked)
Snowbank	NA (Masked)	NA (Masked)
Water	NA (Masked)	NA (Masked)

Landsat image collections for the time period of interest (1 July to 1 September 1985 to 2019) were queried in the Google Earth Engine platform [48] and underwent an automated quality assurance procedure that screened pixels for sensor saturation (likely indicative of snow or water), cloud cover (>80%), and atmospheric haze and opacity. Images from Landsat 5 and 8 were cross-calibrated to a Landsat 7 equivalent using the coefficients and methods described in Pironkova et al. [49]. Composites containing red, green, blue color (RGB), shortwave infrared (SWIR), and near infrared (NIR) bands were generated for each year using the median values of the remaining unmasked pixels. The normalized difference vegetation index ( $NDVI = NIR - R / NIR + R$ ) and normalized difference water index ( $NDWI = NIR - SWIR / NIR + SWIR$ ) were later calculated for each composite image in R [50]. Inspection of the annual composites showed persistent data gaps due to cloud cover and/or late lying snow within individual summers, therefore composites were binned into 5-year averages (1985/89, 1990/94, etc.) for analysis. Data availability within the park boundary was >80% in all binned composites (mean 86.3%; SD = 4.9%) and exceeded 90% in the 2010/14 and 2015/19 bins (coincident with the launch Landsat 8 in 2013).

### 2.3.3. Topographic Rasters

A series of additional rasters representing topographic characteristics were prepared to inform the classification of land cover types. Slope, aspect, and latitude rasters were generated in QGIS from the Government of Canada's Canadian Digital Elevation Model [51] which was reprojected and aggregated to a 30 m resolution (bilinear interpolation). Aspect was folded about the northeast–southwest line to rescale values from 0° to 180° and address the circularity of compass measurements [52]. In the resulting raster, large values indicate

the warmest (southwest-facing) and low values represent the coolest aspect (northeast-facing). A distance to the coast raster was also generated from a rasterized shapefile of the coastal boundaries of Québec and Labrador.

#### *2.4. Hindcasting (1985/2014) and Simulating (2015/2019) Land Cover Using Random Forest Classification*

Land cover rasters were generated using summer 2014 Landsat composites/indices (red, green, blue, SWIR, NIR, NDVI, NDWI) and topographic variables (aspect, slope, latitude, elevation, distance to coast) in a pixel-based random forest classification algorithm (see workflow in Figure S1). The random forest algorithm was developed using the “ranger” option from the “train” function (“caret” package; [53]) in R. Preliminary analyses on a subset of the data showed that an “mtry” value (predictors sampled for splitting at each node) of 2 and minimum node size of 1 produced the best models, and that classification errors plateaued rapidly when random forest iterations contained >100 trees. These parameters (mtry = 2, min.node.size = 1, num.trees = 300) were thus adopted for modelling throughout TMNP. The random forest algorithm was trained on a random subset of 50% of the data with the remaining 50% used for model evaluation. Predictor variable influence on model accuracy was qualitatively assessed with a variable importance plot. Overall classification accuracy was determined from test-set predictions using a confusion matrix and receiver operating characteristic (ROC) curves.

Hindcasting (1985/89 to 2010/14) and simulating (2015/19) land cover classes was achieved by applying the random forest algorithm to the binned Landsat composites (topographic variables remained constant), resulting in seven land cover intervals (1985/89 to 2015/19). These served as the basis for detecting land cover changes through time.

#### *2.5. Change Detection*

Changes in land cover classes were first assessed by comparing the output rasters from 1985/89 and 2015/19. Missing pixels were masked across rasters to ensure comparisons occurred over a common area. Next, the frequencies of different transitions in land cover classes over time (e.g., the proportion of 1985/89 pixels that transitioned to shrub, dry vegetation, wet vegetation, or non-vegetated pixels from each of four initial land cover types; 16 possible outcomes) were calculated. Temporal trends in the proportional cover of each land class were assessed by calculating the proportion of each land cover type in the five-year bins relative to the 1985/89 baseline, with missing pixels masked across all seven land cover layers to ensure comparability.

To investigate whether shrubs have shifted to higher elevations or latitudes over time, the elevations and latitudes of “new” shrub pixels were determined for rasters from 1990/94 to 2015/19. Shrub pixels from the binned land cover rasters were isolated from other cover types, and pixels corresponding to shrubs in the previous five-year bin were masked (thus, data from the earliest bin, 1985/89, are not available). The median, and 25th and 75th quantiles of elevation and latitude, were then determined for the new shrub pixels in each raster.

Finally, proportional changes in land cover class were evaluated for seven sites where hourly ground surface and air temperatures are currently being monitored, providing local-scale snap-shots of vegetation change (Figure 1B). The proportion of each land cover class was determined from three binned land cover rasters (1985/89, 2000/05, 2015/19) for the 10 km<sup>2</sup> surrounding the sites. These sites have the potential to be used for validating remotely sensed data and for establishing long-term vegetation monitoring in the future.

#### *2.6. Predicting Future Shrub Expansion*

To identify potential hotspots of future shrub expansion, a set of candidate binomial models were assessed to predict the likelihoods of non-shrub pixels transitioning to shrub cover in the future (see workflow schematic in Figure S2). The approach follows that of Lemay et al. [54] who used both binomial and multinomial regression to predict areas of shrub change in Nunavik, Québec. Their results showed that although a binomial shrub

change model ignores other potential transition types, its predictive ability was similar to that of more complex multinomial models. Our approach similarly used topographic and biotic variables, derived from the 1985/89, “time<sub>1</sub>”, land cover raster, to parameterize the model and predict shrub dominance in 2014, “time<sub>2</sub>”. Potential topographic predictors included elevation (m above sea level), folded aspect, a quadratic term for slope (to allow for the possibility that moderate slopes could be more favorable than very steep or shallow slopes), the additive and interactive effects of slope and aspect, latitude, and the distance to the coast. The candidate biotic predictors included rasters for land cover class (categorical; land cover class of a pixel at time<sub>1</sub>), neighboring shrubs (numeric; number of shrub class pixels within a 24-pixel search window of the focal pixel at time<sub>1</sub>), and shrub edge (binary; whether the focal pixel was adjacent to a shrub pixel at time<sub>1</sub>, determined from an 8-pixel search window). From an ecological perspective, it was expected that land cover types would differ in their likelihood of becoming shrub dominant at time<sub>2</sub>. We also expected that pixels adjacent to existing shrubs would be more likely to transition to shrub dominance in the future due to higher potential for seed production and dispersal or clonal growth from proximal shrubs. Sites in the vicinity of extant shrubs could also have an inherently higher likelihood of being suitable shrub habitat due to physical changes associated with shrub structure (e.g., [55]). The 2014 (time<sub>2</sub>) shrub cover raster was split into a training set containing 50% of the data to parameterize the candidate models and a test set to evaluate the accuracy of the final model. The potential influence of spatial autocorrelation on model parameterization was addressed by selecting every other pixel for the training set and the remainder for the test set. As in Lemay et al. [54], pixels that were themselves shrub dominant at time<sub>1</sub> were excluded from the analysis because it was assumed that they would still be shrub dominant at time<sub>2</sub> (transition probability of 1); this tendency was confirmed by our analyses of land cover transitions that showed shrub pixels to be highly persistent over time (see Results for more details).

Multimodel inference was employed to select the most suitable predictive model from a suite of candidate models of varying complexity. Several constraints were imposed on the possible combinations of predictor variables to limit the number of comparisons [54]. Land cover class was included in all models, whereas either shrub edge or the number of shrub neighbors was included. Latitude and distance to the coast were always included together in candidate models, and slope and aspect (in the form of “slope + aspect + slope × aspect + slope<sup>2</sup>”) were likewise grouped. These conditions resulted in 24 candidate models (see full list of models in Table S1) that were evaluated using an information theoretic approach (“MuMIn” package in R; [56]). The most suitable model was selected using Akaike’s Information Criterion (lowest AIC); model averaging using Akaike’s weight can be a means of increasing the robustness of the predictions by weighting the parameters of the top performing models [57], but this was not required because the top ranked model had an Akaike weight of 1 (i.e., little additional information would be gained by model averaging). The selected model was then applied to the test-set to evaluate model accuracy against the distribution of shrub class pixels from the 2014 land cover raster; shrub dominant pixels at time<sub>2</sub> were considered those with a probability  $\geq 0.50$  of becoming shrub dominant given their associated topographic and biotic characteristics.

In the final step, the parameterized binomial model was extended to determine the likelihood of pixels transitioning to shrub dominance one time step into the future (2039/43; time<sub>3</sub>). The temporal span of predictions (2014 to 2039/43; 27 years) was determined from the length of time between the midpoint of the earliest available Landsat composites (1985/89) and the reference land cover imagery (2014). Beyond these practical considerations, the timespan has the benefit of informing near-term research planning and park management and is relevant to the lifetimes of people using the area today. Rasters representing the same biotic variables as for time<sub>1</sub> were generated for time<sub>2</sub> from the 2014 land cover raster, whereas topographic variables were assumed to remain constant. Once again, pixels in the shrub cover class (now for time<sub>2</sub>) were masked from the predictor rasters. A raster depicting shrub transition probabilities was created from the model output.

### 3. Results

#### 3.1. Random Forest Algorithm

The overall classification accuracy of the random forest algorithm (test-set data of 2014 land cover classes) was 81.1% and the area under the ROC curves was 0.77 (average of the four classes). NDVI, NIR, and NDWI had the highest relative importance for improving the classification accuracy of the random forest algorithm, whereas SWIR, slope, and aspect were of lowest importance (Figure S3). The balanced classification accuracies by land cover type varied from 74.2% for non-vegetated pixels to 85.4% for shrub pixels (Table 2).

**Table 2.** Confusion matrix showing the number of actual vs. predicted pixels in each vegetation class for the test-set data. The bolded numbers indicate the number of correctly classified pixels in each class. The bottom row reports the balanced classification accuracy for each land cover type.

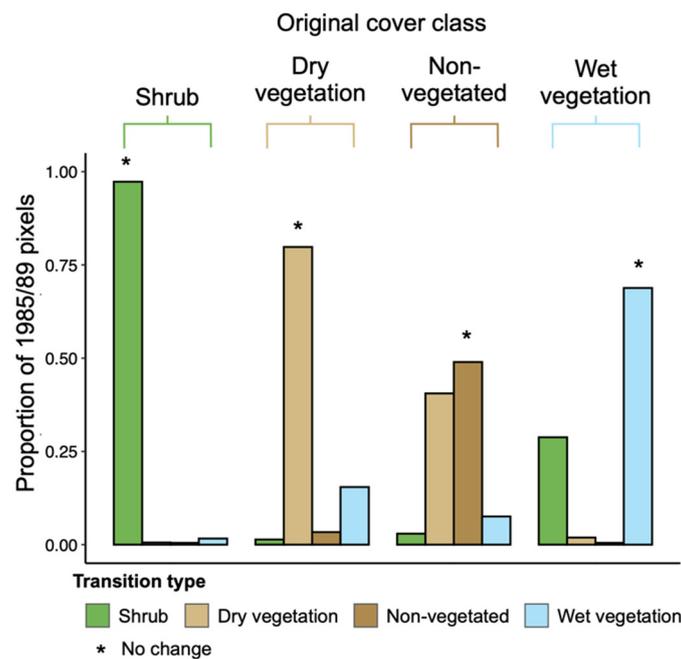
	<b>Shrub (Actual)</b>	<b>Dry veg. (Actual)</b>	<b>Non-veg. (Actual)</b>	<b>Wet veg. (Actual)</b>	<b>Sum (Predicted)</b>
Shrub (predicted)	197,908	8211	6030	36,526	248,675
Dry veg. (predicted)	16,160	2,328,700	206,718	210,150	2,761,728
Non-veg. (predicted)	2882	68,008	237,276	7230	315,396
Wet veg. (predicted)	57,619	137,502	19,207	578,506	792,834
Sum (actual)	274,569	2,542,421	469,231	832,412	-
Balanced classification accuracy	85.38%	82.06%	74.21%	81.49%	-

#### 3.2. Temporal Changes in Land Cover Class

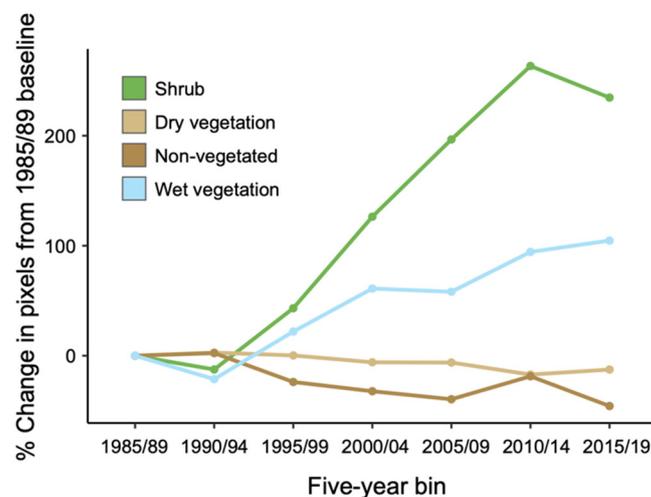
Here, temporal changes in land cover are described by cover type. Point-in-time changes were determined from the frequency of class transitions between the 1985/89 binned raster and the 2014 reference land cover raster. Temporal trends were determined by comparing the number of pixels of each cover type in each 5-year bin relative to the 1985/89 baseline raster.

##### 3.2.1. Shrub Cover Class

The vast majority (97.3%) of shrub-dominant pixels in the 1985/89 raster remained so in 2015/19, suggesting that transitions from shrub dominance to other vegetation types rarely occurred (<3% of the time; Figure 2). Notably, 28.8% of pixels in the wet vegetation class in 1985/89 transitioned to shrub cover in 2015/19, compared to only 1.4% and 3.0% of dry and non-vegetated pixels. When common pixels were compared across the seven binned composites (Figure 3), shrub cover underwent the largest proportional change over time, with a 234.7% increase in the number of shrub-dominant pixels by 2015/19 relative to the 1985/89 composite. We note that shrubs represent the smallest of the four cover classes, and the change represents an increase from 2.2% to 7.1% of the total land cover pixels with available data. The increase in shrub cover occurred rapidly from 1990/94 onwards (Figure 3).



**Figure 2.** Proportion of pixels transitioning between land cover classes from 1985/89 to 2015/19. Bars are grouped by the original (1985/89) cover class and transitions are indicated by colors; ‘\*’ indicates the case where pixels remained in the same cover class between time periods.



**Figure 3.** Percent change in the abundance of pixels in each land cover class relative to 1985/89 values.

### 3.2.2. Dry Vegetation Class

Most pixels identified as dry vegetation in 1985/89 (79.8%) remained in that cover class in 2015/19 (Figure 2). The largest between-class transition was to the wet vegetation class (15.5% of pixels). Many non-vegetated pixels in 1985/89 became dominated by dry vegetation in 2015/19 (40.6% of pixels). There was little temporal change in the percentage of dry vegetation cover (Figure 3) due to the balance of pixels gained from the non-vegetated category and pixels lost to the wet vegetation category, and the large number of pixels in the dry vegetation category (i.e., a larger absolute change in the number of pixels required to cause a change in percent cover).

### 3.2.3. Non-Vegetated Class

About half of the non-vegetated pixels in 1985/89 remained so in 2015/19 (49.0%). As described above, many non-vegetated pixels in the earlier time period became dominated by dry vegetation by 2015/19 (40.6%). There was a general decline in the abundance of non-vegetated pixels over time, with 45.7% fewer in the 2015/19 raster relative to 1985/89.

### 3.2.4. Wet Vegetation Class

The wet vegetation class saw 28.8% of pixels transition to shrub cover by 2015/19 while 69.8% remained unchanged. Transitions to other land cover types represented only a small portion of the total outcomes. By 2015/19, wet vegetation had increased by 104.7% relative to 1985/89, with most of the gains coming from the dry vegetation class. As with changes in shrub cover, the increase in wet vegetation primarily occurred after 1990/94.

## 3.3. Spatial Changes in Land Cover Class

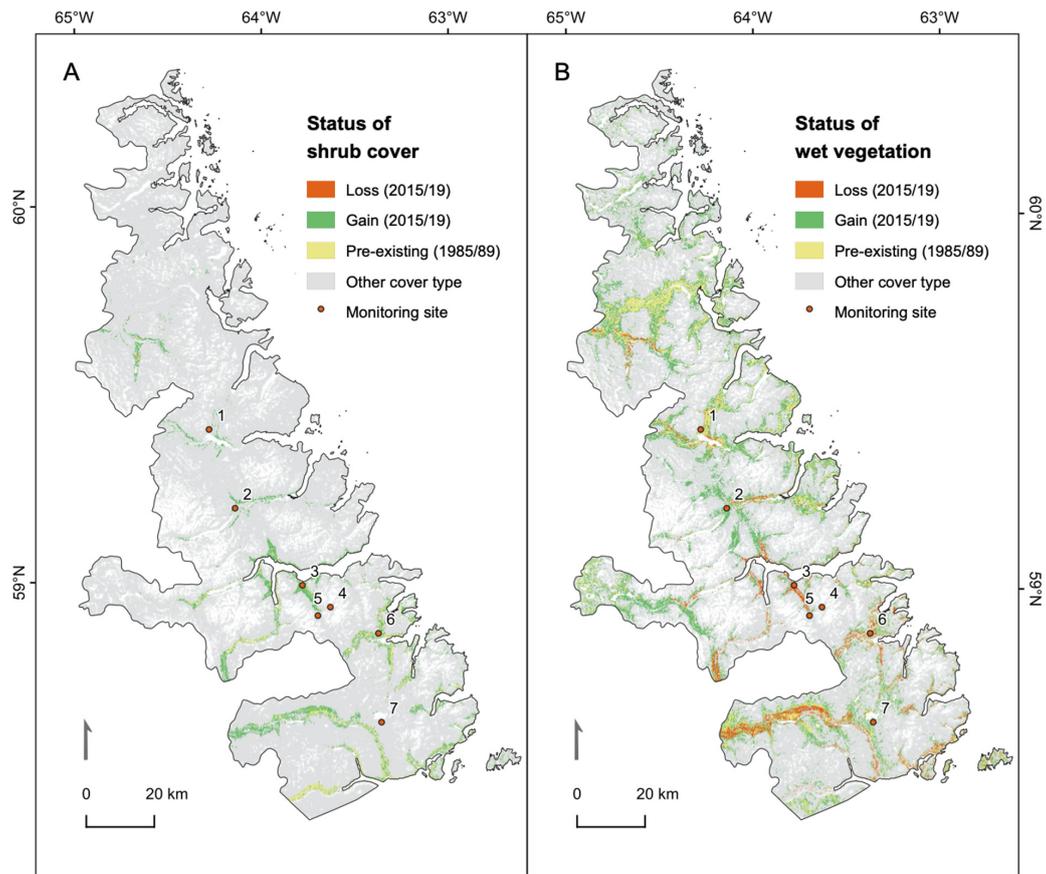
Comparisons of the distribution of shrub-dominant pixels between 1985/89 and 2015/19 showed that shrubs generally tended to expand in areas adjacent to existing shrub patches, particularly in drainage slopes and valley bottoms (Figure 4A). Most of the shrub gain in 2015/19 occurred in the southern half of the park, and relatively little expansion occurred in the top third. Although there was significant variability across elevations and latitudes, the median elevation of new shrub establishment appears to be increasing over time (Figure 5A). The median latitudes of new shrub pixels showed no temporal trend but the extreme limit of new shrub pixels showed a clear latitudinal increase from 2005 to 2019 (Figure 5B).

In general, gains in wet vegetation tended to occur at mid-elevations near and adjacent to drainage areas. In the southern portion of the park, extensive loss of wet vegetation occurred between the two time steps (Figure 4). By comparing the area of wet vegetation loss (red in Figure 4B) to that of shrub gain (green in Figure 4A), it is clear that most of this loss represents a transition to shrub dominance in 2015/19. As with gains in shrub cover, transitions to dry vegetation (mainly from the non-vegetated class) tended to occur in the south of the park. Maps of gains and losses of dry and non-vegetated pixels can be viewed in Figure S4.

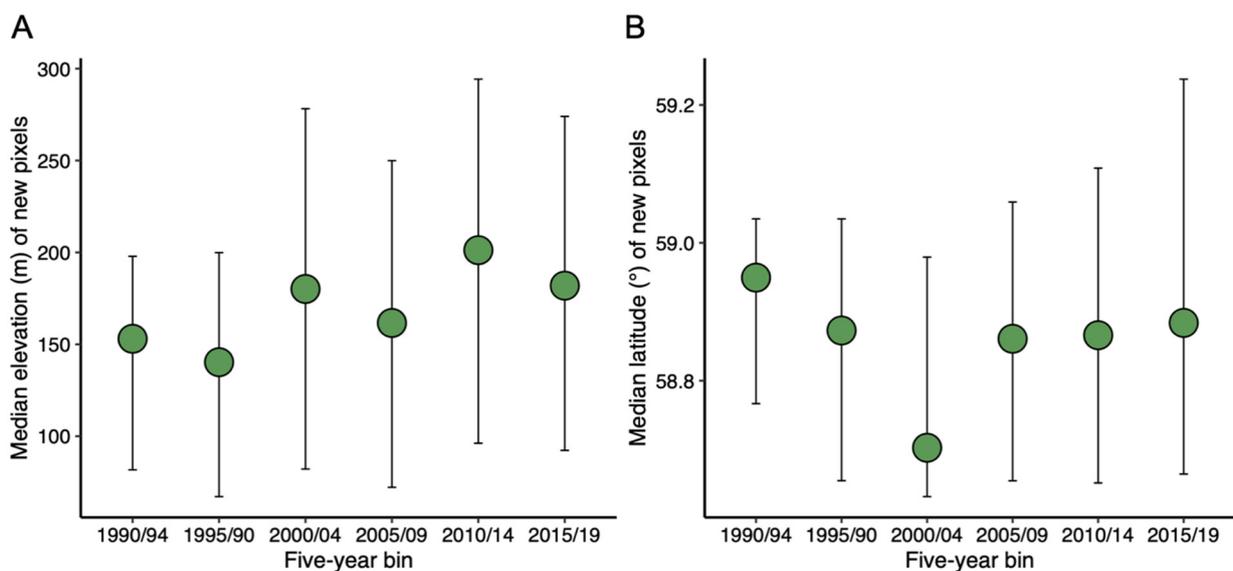
## 3.4. Predicting Changes in Shrub Cover

Model comparisons revealed the model containing all predictor variables except shrub edge to be the best model for predicting shrub probabilities (Tables S1 and S2). When applied to the test set, the model accuracy was 95.6% and the true positive rate of the shrub class was 71.3%. The very high overall model accuracy in part reflects the large number of non-shrub pixels in the dataset that were correctly classified. The AUC of the test-set ROC curve was 0.93.

The model output showed elevation, latitude, wet vegetation, and distance from the coast to be the most influential predictors (Table 3). The probability of a pixel transitioning to shrub dominance decreased with elevation and latitude (Figure 6) and was higher inland than near the coast. Shrub expansion was also more likely for wet than for non-vegetated or dry vegetation pixels (Table 3; Figure 6). The number of shrub pixels in the neighborhood had a positive effect on the likelihood of a focal pixel also being a shrub. The relatively smaller parameter estimates and z-values for slope and aspect indicate that the influence of these topographic variables on shrub probabilities was more subtle.



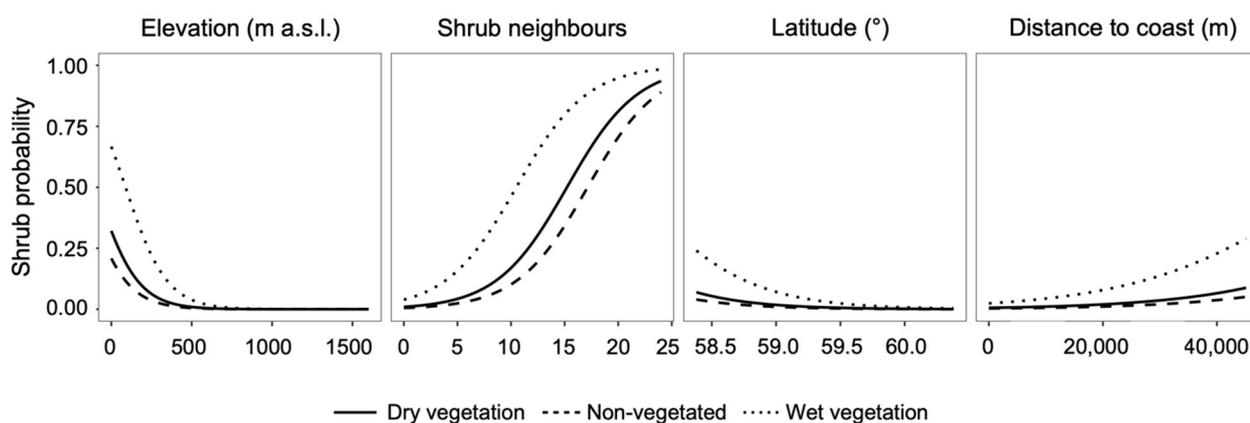
**Figure 4.** Maps depicting changes in shrub (A) and wet vegetation (B) land cover classes between 1985/89 and 2015/19. In (B), most pixels in the “loss” category correspond to areas of shrub gain in (A). Monitoring sites: 1—Kangalaksiorvik Lake; 2—Komaktorvik Falls; 3—Ivitak Cove; 4—Cirque Mtn. Camp; 5—Upper Ivitak Valley; 6—Ramah Bay; 7—Nakvak Brook.



**Figure 5.** Median (green dot) and 25th and 75th percentiles (bars) of the elevation (A) and latitude (B) of new shrub pixels in binned land cover rasters. New shrub pixels were determined by masking (removing) the shrub pixels present in the previous five-year raster (thus, 1985/89 not shown).

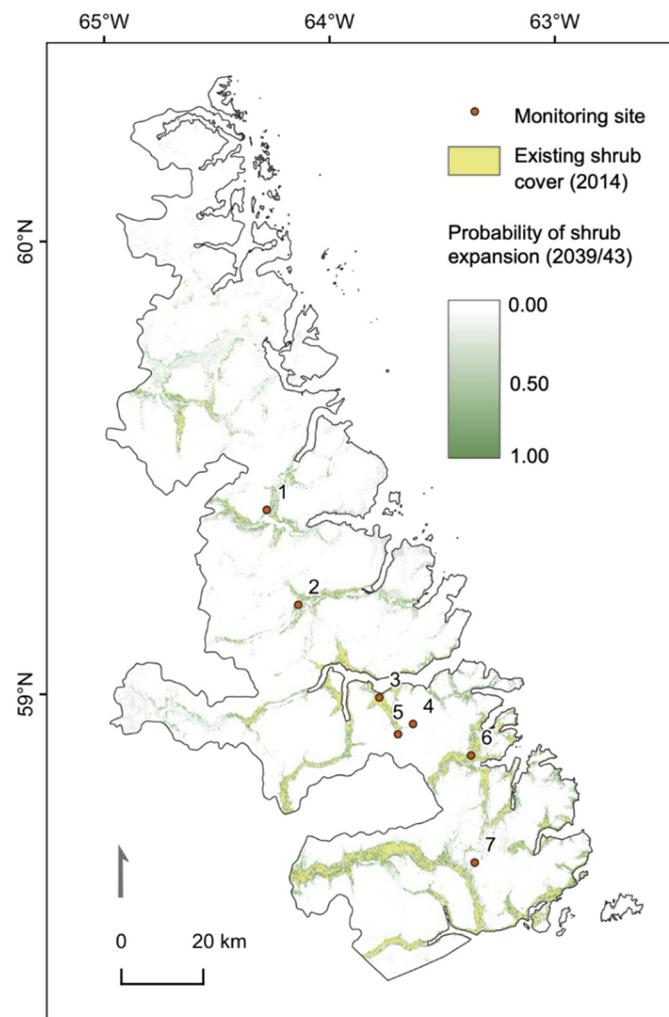
**Table 3.** Model output of the selected binomial regression model in which the probability of a pixel being shrub dominant at time<sub>2</sub> (2014) was estimated from topographic and biotic variables at time<sub>1</sub> (1985/89).

Variable	Estimate	Std. Error	z Value	p value
(Intercept)	133.00	0.44	302.04	<0.0001
Land cover: No veg	−0.59	0.01	−57.77	<0.0001
Land cover: Wet veg	1.44	0.01	228.47	<0.0001
Neighbour shrubs	0.30	0.00	177.88	<0.0001
Elevation	−0.01	0.00	−354.22	<0.0001
Latitude	−2.28	0.01	−308.48	<0.0001
Distance to coast	$6.21 \times 10^{-5}$	0.00	193.50	<0.0001
Aspect	0.11	0.00	21.89	<0.0001
Slope	2.97	0.05	54.77	<0.0001
Slope <sup>2</sup>	−3.67	0.07	−55.94	<0.0001
Aspect × slope	0.18	0.02	10.31	<0.0001



**Figure 6.** Modelled probabilities of dry vegetation, non-vegetated, and wet vegetation pixels undergoing a state transition to the shrub cover class based on changes in the four most influential variables in the binomial model. In each panel, the relationship between shrub probabilities and the predictor variable is shown with other model variables set to their average values. Model confidence intervals are not visible at this scale because they are very narrow, likely due to the large size of the dataset.

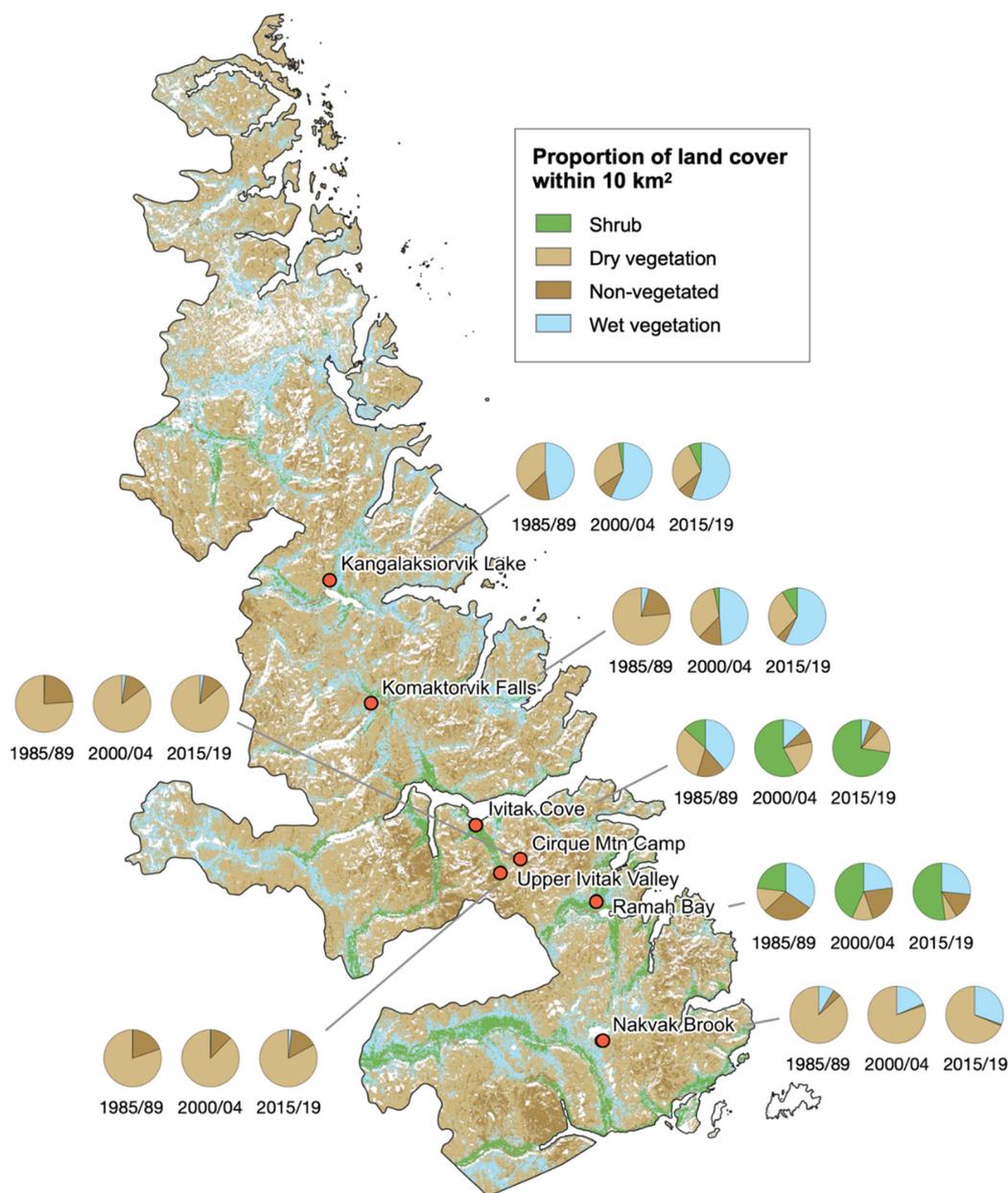
The shrub probability estimates showed that a 50.9% increase in the extent of shrub cover is expected by 2039/41 relative to 2014. Expansion is more likely to occur near areas currently occupied by shrub patches (Figure 7), suggesting that local habitat infilling will be a dominant mode of shrub expansion. The higher likelihood of wet vegetation transitioning to shrub cover in the binomial model and the presence of extensive shrub patches in river valleys indicates that soil moisture has historically been an important determinant for shrub colonization. Predictions for 2039/43 show that latitudinally, shrub expansion is more likely to occur in the southern two-thirds of the park than in the north. In the northern third, new shrub colonization is most probable in low elevation areas currently occupied by wet vegetation, but at lower probabilities overall.



**Figure 7.** Extent of shrub class pixels in 2014 (time<sub>2</sub>; yellow) and probability of pixels becoming shrub dominant by 2039/43 (time<sub>3</sub>; green scale). Monitoring sites: 1—Kangalaksiorvik Lake; 2—Komaktorvik Falls; 3—Ivitak Cove; 4—Cirque Mtn. Camp; 5—Upper Ivitak Valley; 6—Ramah Bay; 7—Nakvak Brook.

### 3.5. Vegetation Change at Existing Monitoring Sites

Temperature monitoring stations are currently located in the southern two-thirds of the park (Figures 1B and 8) due to the higher cost and logistical challenges accessing the northern portion of the park. Temporal changes in the proportions of land cover classes were variable across the seven monitoring sites (Figure 8). A large increase in the proportion of shrub cover occurred at Ivitak Cove from 1985/89 to 2015/19, whereas nearby Upper Ivitak Valley and Cirque Mountain Camp have seen an increase in dry vegetation but not shrub cover. These findings highlight the importance of elevation and wet vegetation cover as predictors of shrub expansion; Ivitak Cove is a low valley site (36 m a.s.l.) with ample wet vegetation cover in the 1985/89 raster. In contrast, Upper Ivitak Valley and Cirque Mountain Camp are both located >450 m a.s.l. and were initially dominated by dry and non-vegetated cover (i.e., less susceptible to shrub expansion). Full location details of monitoring sites are available in Table S3.



**Figure 8.** Changes in the proportions of the four land cover classes within 10 km<sup>2</sup> of existing monitoring sites in Torngat Mountains National Park. Basemap shows the distribution of land cover classes in the 2014 reference layer.

Shrub expansion has also been substantial further south at Ramah Bay. About a quarter of pixels within the monitoring area were shrub dominant in 1985/89, increasing to about one half by 2015/19. This change is supported by a series of repeat photographs taken near the monitoring site (Figure S5). Finally, the southernmost site at Nakvak Brook, located on a plateau at 506 m a.s.l., has shown little increase in shrub cover, in contrast with the nearby river valley. The proportion of wet vegetation cover has increased over time, however, indicating that the area could become more suitable for shrub expansion in the future.

#### 4. Discussion

##### 4.1. Temporal and Spatial Changes in Land Cover

The shrub and wet vegetation classes saw the most significant gains in spatial extent from 1985/89 to 2015/19. Increased shrub cover and growth have previously been

documented in the region [14,16,31,58,59] and have generally been attributed to regional warming [16,31,60]. The timing and spatial scope of the changes observed here similarly suggest that climate has been an important control on vegetation change throughout TMNP; widespread shrub expansion began in the mid-1990s, corresponding to a period of renewed warming and increased precipitation in the Torngat Mountains [31,42]. As would be expected under a warming climate scenario, shrub expansion also appears to be accelerating at higher elevations and (to a lesser degree) higher latitudes in recent decades. It is important to note, however, that a substantial decline in the size of the George River caribou population, whose migratory range overlaps TMNP, occurred over the same period as temperature warming [30]. Indeed, the interacting effects of an increasingly favorable climate and reduced herbivory may, at least in part, explain the relatively high rate of shrub expansion that has occurred in the Torngat Mountains relative to other areas of the North American Arctic [16,45].

In addition to increasing spatial coverage, shrub class pixels were also highly persistent on the landscape, with few from the early time period transitioning to other land cover types (<3%). Lemay et al. [54] similarly observed that only a small portion (14%) of shrub dominant pixels underwent a change in cover type from 1990/94 to 2010 in Nunavik, Canada. Examining the life history traits of the major shrub taxa (*Salix*, *Alnus*, and *Betula*) reveals several possible explanations for their persistence. Many shrub species are relatively long-lived compared to other arctic plants (decades to centuries; [61] and references therein) prolonging their lifespan by surviving underground and spreading asexually through layering during poor conditions [62]. With their comparatively large stature and ability to quickly respond to improved growing conditions through shoot extension, low to tall shrubs are also positioned to be strong competitors for space and resources while altering microhabitat conditions such as light availability [63], snow distribution [64], and soil conditions [65–67]. The rapid increase in shrub cover and persistence of pixels suggests that shrub dominant areas in TMNP are unlikely to transition to other cover types in the short term, barring major disturbances (e.g., via wildfire, a strong increase in herbivory, or flooding).

A notable finding of the transition analysis was that most of the gain in the shrub class took place in low-elevation riparian areas from pixels previously classified as wet vegetation. These changes are likely driven by expansion of the low-medium shrub class given their relatively large abundance (e.g., compared to dwarf shrubs) and tendency to occupy valley bottoms in the original ecotype raster. Increased shrub cover in riparian and floodplain habitats has also occurred in western North America [68–70] as these areas seem to possess favorable hydrological characteristics. The role of soil moisture as a mediator of shrub expansion [23,68], productivity [24], and growth [9,20] has been widely reported, with the positive effects of warming temperatures often being greater where moisture is non-limiting. At the same time, highly saturated soils can also be unfavorable for some shrubs [70,71], indicating that expansion is more likely where moisture requirements are met but not greatly exceeded. It is thus possible that the conversion of wet vegetation to shrub cover observed in TMNP was initially facilitated by surface drying, which can promote shrub expansion in wet areas [72]. Surface drying can be caused by increased evaporation due to warming, soil drainage from permafrost thaw, or reduced soil moisture caused by earlier snow melt. Such changes can negatively impact waterfowl species whose wetland habitat can depend on the presence of frozen ground [73]. Mid-summer soil moisture in wet areas in the south of TMNP has been observed to decline in recent years [31], although the underlying mechanisms and the associated impacts on plant communities require further study.

The consequences of shrub expansion have justifiably received significant attention in the literature in recent years; however, transitions from dry to wet land cover are also a noteworthy climate change impact because they signal a change in hydrological processes with broader implications for ecosystem functioning [74] and wildlife [73]. Our analysis of historical land cover change showed a net increase in the extent of wet vegetation over

time, with most of the gains coming from the dry vegetation class at mid-elevations. Recent climate change impacts in TMNP have included increased summer precipitation [42], permafrost degradation [27], and glacial melt [28,29], which together may have contributed to the expansion of wet vegetation (moist sedge and fen ecotypes) at higher elevations. Transitions to thermokarst/wetland habitat can arise when active layer deepening occurs atop poor subsurface drainage (e.g., impermeable ground; presence of permafrost at depth) and causes ponding [75]. This has been observed in western Canada where forest loss has resulted from waterlogging caused by permafrost thaw and associated ground surface subsidence [76]. Field validations are needed to determine whether similar processes are occurring in TMNP; however, in the only field-based observations of permafrost conditions available, Davis et al. [31] found active layer depth to be greater in areas of high surface wetness and a reduced probability of permafrost under pooled surface water relative to dry sites.

Changes in dry and non-vegetated cover were proportionally smaller than for the shrub or wet vegetation classes, possibly due to the dry, shallow soils, rocky substrates, and/or more extreme environments found in these cover classes that would restrict many plant species. The gains in dry vegetation that did occur tended to originate from non-vegetated pixels in the southern half of the park, and importantly may be contributing to the general greening trend observed over the past three decades [16]. Future field validations would help to clarify the dominant life forms that have become establishing in these areas.

#### 4.2. Future Shrub Expansion in TMNP

The predictive model revealed shrub expansion to be regulated by a combination of biotic and physical variables. The positive effect of the shrub neighbor variable in the model signals that expansion is more likely to occur near existing shrub patches. This tendency has been observed in tundra environments elsewhere through repeat photography [77] and statistical modelling [54], and could be caused by the higher reproductive potential and/or improved growing conditions near areas that are already shrub-dominant. Spread from existing shrub patches may also be enhanced through a positive feedback with snow cover [13,78], because snow that becomes trapped in shrub branches provides protection from harsh winter conditions and moderates soil moisture and temperature regimes [79]. Spatial correlation in properties such as microclimate and soil conditions could also mean that the environment is more likely to be favorable for shrub expansion near existing shrub habitat.

Topographic and geographic variables have also been shown to moderate changes in shrub cover (e.g., [21,80,81]), because landscape position serves as a proxy for factors that influence plant species composition. In TMNP, we found that elevation and latitude were inversely related to the probability of shrub expansion, whereas probabilities increased with distance from the coast. Low-lying, continental areas in the south of the park likely offer warmer and more sheltered environments as temperatures decline and wind speeds increase at higher elevations and to the north. Peak greening in the northern portion of TMNP was found to last only about one month, which could be related to enhanced exposure to cold, coastal conditions at the tip of Nunavik and Nunatsiavut, and may constrain further shrub expansion in the area. The frequent transition of pixels from the wet vegetation to the shrub cover class in TMNP also suggests that soil moisture in low lying areas may provide the conditions necessary for further expansion, as has been observed in northwestern Canada [81]. Given the ties between shrub cover and snow, elevational or latitudinal differences in the amount and duration of snow cover may also be expected to influence shrub distributions.

We note that the accuracy of the predictive shrub change model relies on several assumptions, foremost of which is that relationships between land cover transitions between time steps (i.e., 1985/89 and 2014), and rates of change, remain constant into the future. Furthermore, information about local geomorphology, the influence of climate change,

and species/lifeform-specific data are not explicitly incorporated into our predictions. Finally, our detection of land cover transitions using post-classification change detection has several advantages and disadvantages [82]. The technique was beneficial in allowing us to identify the frequency of transitions across all land cover classes, but the accuracy of the final product is inherently tied to the initial reclassification of the ecotype map, which has not yet been validated in the field. Nonetheless, although we caution against interpreting meaning from individual pixels, the models do provide insight into general areas and cover classes that are more or less likely to undergo future change.

#### 4.3. Implications for Wildlife, Protected Areas Management, and People

Newly protected areas such as TMNP can benefit from research that extends the state of knowledge deeper in time because in situ monitoring programs will have only been recently established. We have shown that substantial changes in land cover have occurred in TMNP from 1985 to the present, and that further increases in shrub cover can be expected in the future. The timing of land cover change in TMNP exemplifies the value of combining remote sensing data with in situ data collection to monitor environmental change in protected areas. Our analysis showed that substantial changes in land cover had already occurred prior to the establishment of the park and the initiation of associated monitoring programs in 2005. Data suggest that vegetation monitoring in low elevation/latitude areas (<500 m a.s.l. and south of 59.0°N) with adequate soil moisture and nearby shrub patches will be suitable for early detection of shrub expansion. It would be beneficial to establish additional monitoring plots along naturally occurring gradients of soil moisture, elevation, latitude, and continentality to improve our understanding of the moderating influence of these factors on shrub expansion in the region. In regard to other land cover changes, wet vegetation appears to have expanded the most in the vicinity of Komaktorvik River. This monitoring site could be well-suited for more detailed studies of the mechanisms underlying the conversion from dry to wet vegetation and from wet vegetation to shrub cover. Finally, although transitions to dry vegetation were less frequently observed, this class occupies the greatest area such that even small proportional changes could have larger cumulative impacts if they occur across the entirety of the park.

The expansion of shrub cover that has already been observed and is predicted to continue in TMNP has implications for several ecosystem processes. The presence of shrub canopies has been associated with a decline in the productivity of culturally significant berry plants through shading [12,15,83], and a reduced abundance of lichen [17,84], an important winter forage for caribou. Caribou may also be impacted by the displacement of wet vegetation and expansion of shrubs in low valleys because caribou browse in sedge meadows early in the growing season. In contrast, animals that make use of shrubs, such as some species of migratory birds, may see their habitats expand into new areas with changes in land cover [41,85]. The winter survival of subnivean animals may also benefit from shrub expansion as snow trapping in dense branch canopies insulates the ground and offers protection against harsh conditions [73]. On a larger scale, and because shrub cover has a role in many landscape-level feedbacks [13], shrub expansion will likely impact rates of carbon and nutrient cycling, and modify hydrological processes. For example, shrub expansion may alter ground temperatures and associated permafrost conditions [86] through interactions with the accumulation and duration of snow cover [10,64,87]. Shrub expansion from 2014 to 2039/43 is expected to be proportionally more modest at 50.9% than what occurred from 1985/89 to 2015/19 (234.7%), however, implying that even under improved climate conditions, other controlling factors will ultimately slow spread and limit the area that can plausibly be converted to shrub cover.

Finally, the past and future land cover change we have documented could also affect Inuit use of the land and park visitation. For example, expansion of shrub thickets along river valleys can make travel by foot or snowmobile more difficult along these important corridors, while also providing cover for black bears (*Ursus americanus*) and polar bears (*U. maritimus*), which presents challenges for human safety. Vegetation change

will also affect the availability of important country foods such as caribou, ptarmigan, and berries, with consequences for food security and the connection of Inuit to the land. Understanding how these trends may unfold in future could help park managers and Inuit in the region better adapt and minimize the cultural and socioeconomic consequences of environmental change.

## 5. Conclusions

Our research has demonstrated that substantial changes in land cover have occurred in Torngat Mountains National Park over the past four decades. The most notable changes include a large increase in the extent of shrub cover in low-latitude valley bottoms, as shrubs have displaced wet vegetation, and a shift from dry to wet vegetation cover at mid-elevations. The nature and extent of these changes indicate that climate change, in addition to biotic (e.g., land cover type, plant–herbivore interactions) and topographic (e.g., elevation, latitude, continentality) controls, has been the main underlying driver of ecosystem change. In the future, a further increase in shrub cover is expected to occur with implications for management of protected areas.

**Supplementary Materials:** The following are available online at <https://www.mdpi.com/article/10.3390/rs13112085/s1>, Figure S1. Depiction of workflow used to generate four-class land cover rasters (1985/89 to 2015/19) from Landsat composites and topographic data for TMNP. Figure S2. Depiction of the workflow used to predict transitions to shrub dominance from biotic and topographic variables using a binomial regression framework. Figure S3. Relative variable importance plot of the random forest algorithm. Predictors with higher/lower overall importance values contribute more/less to the accuracy of the classification algorithm. Figure S4. Maps showing changes in dry vegetation (A) and non-vegetated cover (B) from 1985/89 to 2015/19 in Torngat Mountains National Park. Figure S5. Shrub change near Ramah Bay shown through (A) repeat photography and (B) predicted land cover maps of the park during a similar time period. The yellow dot in (B) is the approximate location of the photographs in (A). Table S1. Variables included in the 24 candidate binomial regression models used in an information theoretic approach to select the most suitable model for predicting shrub cover change. Table S2. Summary statistics of the comparison between 24 candidate binomial regression models of shrub change between 1985/89 and 2014, in order of rank (Model—name of model corresponding to Table 1; df—model degrees of freedom; logLik—model log-likelihood; AIC—Akaike’s Information Criterion;  $\Delta$ AIC—change in AIC between ranked models; Model weight—Akaike weight). Table S3. Locations of existing ground and air temperature monitoring sites in Torngat Mountains National Park.

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**Data Availability Statement:** The spatial data and code used in this study are available by contacting the corresponding author.

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