Potential effects of climate change on ecosystem distribution in Alberta

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Abstract: We propose a new and relatively simple modification to extend the utility of bioclimatic envelope models for land-use planning and adaptation under climate change. In our approach, the trajectory of vegetation change is set by a bioclimatic envelope model, but the rate of transition is determined by a disturbance model. We used this new approach to explore potential changes in the distribution of ecosystems in Alberta, Canada, under alternative climate and disturbance scenarios. The disturbance model slowed the rate of ecosystem transition, relative to the raw projections of the bioclimatic envelope model. But even with these transition lags in place, a northward shift of grasslands into much of the existing parkland occurred over the 50 years of our simulation. There was also a conversion of 12%–21% of Alberta's boreal region to parkland. In addition to aspatial projections, our simulations provide testable predictions about where ecosystem changes as a result of climate change are most likely to be initially observed. We also conducted an investigation of model uncertainty that provides an indication of the robustness of our findings and identifies fruitful avenues for future research.

Résumé : Nous proposons une nouvelle modification relativement simple pour étendre l'utilité des modèles d'enveloppe bioclimatique à la planification et à l'adaptation de l'usage des terres dans le contexte des changements climatiques. Dans notre approche, le changement de trajectoire de la végétation est déterminé'par un modèle d'enveloppe bioclimatique mais le taux de changement est déterminé par un modèle de perturbation. Nous avons utilisé cette nouvelle approche pour explorer les changements potentiels dans la distribution des écosystèmes en Alberta, au Canada, en fonction de différents scénarios concernant le climat et les perturbations. Le modèle de perturbation a ralenti le taux de transition des écosystèmes relativement aux projections brutes du modèle d'enveloppe bioclimatique. Mais même avec des périodes de transition plus longues en place, les prairies se sont étendues vers le nord et ont envahi presque toute la forêt-parc actuelle pendant la période de simulation d'une durée de 50 ans. Il y a aussi eu une conversion de 12 % – 21 % de la zone boréale de l'Alberta en forêt-parc. En plus des projections aspatiales, nos simulations génèrent des prédictions qui peuvent être testées quant aux endroits où les changements d'écosystème dus aux changements climatiques ont le plus de chances d'être observés en premier. Nous avons aussi investigué l'incertitude associée au modèle qui fournit une indication de la robustesse de nos résultats et identifie les pistes prometteuses de recherches futures.

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Introduction

A growing body of scientific literature suggests Canada's prairie provinces will undergo significant changes over the course of this century as a consequence of global warming, even if steps are taken to limit future carbon emissions (Lemmen et al. 2004; Sauchyn and Kulshreshtha 2007). By 2050 the mean temperature across the prairies is expected to increase by 2–4 °C, compared with the 1961–1990 reference period, and even greater increases are possible under extreme climate scenarios (Sauchyn and Kulshreshtha 2007). There is less certainty with respect to changes in precipitation, but most models indicate that an increase of up to 15% can be expected, and most of these gains will occur in the winter and spring (Sauchyn and Kulshreshtha 2007). Directional trends observed over the last 50 years have actually been the opposite, however, with an approximately 15% decrease in precipitation observed over the prairies and central mixedwoods of Alberta (Mbogga et al. 2009). Even if precipitation does increase slightly, most regions will become drier during the growing season because of the evaporative effects of increased heating (Schindler and Donahue 2006).

Climate change can be expected to cause changes in the distribution of vegetative communities, given that climate is a key determinant of vegetation patterns at broad geographic scales (Pearson et al. 2002; Walther et al. 2002; Tinner et al. 2008). Such climate shifts occurred in the prairies during the Holocene hypsithermal period (lasting from 9000 to 6000 BP), when North America was approximately 1–3 °C warmer than at present (Strong and Hills 2003). The challenge facing researchers is predicting how and when shifts in vegetation will occur in response to the current round of global warming. Three approaches have been used to address this question: paleoecology, dynamic vegetation modeling, and bioclimatic envelope modeling.

Paleoecological data provide direct insight into how vegetation in the prairies has responded to warmer climate condi-

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tions in the past. For example, whereas Alberta's dominant vegetation type today is boreal forest, pollen-based vegetation mapping shows that most of Alberta was covered by grasslands and aspen parkland during the hypsithermal, with boreal forests restricted to a few high elevation sites (Strong and Hills 2003). However, the rate of climate warming and associated vegetation response during the hypsithermal are poorly defined. Moreover, the rate of warming during this period is unlikely to be comparable to the rapid anthropogenic climate change predicted for the next decades (Weber and Flannigan 1997; Hegerl et al. 2007). While paleoecological data provides valuable information of the possible magnitude of vegetation shifts within centuries, the temporal resolution of these data is too coarse to infer vegetation responses to climate change relevant for land-use planning over decades.

Dynamic vegetation models were originally developed to better understand and simulate short-term changes in vegetated landscapes in response to environmental change, disturbance regimes, and management prescriptions. These models seek to predict the distribution of vegetative assemblages on the basis of mechanistic understanding of plant physiology, population processes, and natural disturbance regimes (Peng 2000; Pearson and Dawson 2003). The ability to predict changes in the distribution of vegetative communities as a result of climate change is an emergent property of these models, to the extent that the influence of climatic variables on plant processes have been characterized. Models of this type are being developed for application to the prairies but have not sufficiently matured to be used in practice (Peng 2000; Luckai and Larocque 2002; Price and Scott 2006).

The third approach for linking climate with the distribution of vegetative communities is bioclimatic envelope modeling (Pearson and Dawson 2003; McKenney et al. 2007). Bioclimatic envelope models (BEMs) statistically associate current vegetation patterns with historical climate data, providing the climatic "envelope" of each vegetation type. With this statistical relationship defined, predictions can be made about how vegetation patterns may shift in response to projected changes in climate (Hamann and Wang 2006). The predicted vegetation type for a given area can be thought of as an equilibrium condition that the site will eventually transition to, given enough time and assuming that the linkage between climate and vegetation type is robust. The strength of the BEM approach lies in its limited ambition: BEMs do not aim to predict any immediate changes or intermediate stages toward the long-term equilibrium. This, however, also limits their usefulness for application in land-use planning, as they provide no indication of how long it will take to reach the predicted equilibrium condition (Woodward and Beerling 1997).

Shifts in the distribution of vegetative communities can be expected to lag behind changes in the bioclimatic envelope. This is primarily because mature individuals of most tree species have substantial tolerance to climatic fluctuations and can persist for extended periods outside of their usual climatic envelope (Hogg 1994). For example, white spruce (*Picea glauca*) trees can be found in farm shelter belts and island populations in many parts of western Canada's grassland ecoregion (Hogg and Schwarz 1997; Chhin and Wang 2002). But while mature spruce trees can persist under these low moisture conditions, their ability to regenerate here is low (Hogg 1994; Hogg and Schwarz 1997; Chhin and Wang 2002). Therefore, the transition to a new vegetation type is likely to be slow as long as mature trees are present but may proceed quickly if the mature trees are killed by a disturbance, leaving a site that is struggling to regenerate and vulnerable to a change in species mix (Schindler 1998; Hogg and Wein 2005). One can expect that these transitions will initially be discontinuous in time and space as they will reflect the distribution of natural disturbances more than changes in the mean climate (Weber and Flannigan 1997; Flannigan et al. 2001; Hogg et al. 2002).

In this paper, we propose a new and relatively simple modification to extend the utility of BEMs for land-use planning and adaptation under climate change. In our approach, the trajectory of vegetation change is set by a BEM model, but the rate of transition is determined by a disturbance model. The working assumption is that changes in the vegetative composition of a given site will not manifest without a disturbance event, which is likely to hold true for at least the next few decades (Hogg 1994; Hogg and Schwarz 1997; Chhin and Wang 2002). We used this new approach to explore potential shifts in vegetation in Alberta, Canada, under alternative climate and disturbance scenarios, and we discuss the implications of these changes for land-use planners and those charged with developing adaptation strategies.

Methods

Bioclimatic envelope modeling

Our study area is the province of Alberta, in western Canada (662 948 km^2). Ecosystems of Alberta comprise grasslands in the south; coniferous, submontane forests in the west; boreal mixedwoods in the north; and various transitional ecosystems. Alberta's land base is classified into a hierarchical system of natural regions, natural subregions, and ecodistricts on the basis of geology, landform, soils, hydrology, climate, vegetation, and wildlife (Alberta Environmental Protection 1994; Strong and Thompson 1995). Ecodistricts are the smallest units in this system of classification, and this unit is the dependent variable in our predictive vegetation model. There are 281 ecodistrict polygons for the province, averaging 2361 km^2 in size, which allows us to subdivide Alberta into climatically homogeneous units. We used a digital version of the ecodistrict map produced by Alberta Sustainable Resource Development in 2003.[3]

Our baseline climate data were derived from the Alberta Climate Model (Alberta Environment 2005), which uses ANUSPLIN software (Hutchinson 2000) to interpolate climate normal data observed at weather stations for the period 1961–1990. These climate variables include monthly average minimum temperature, maximum temperature, and precipitation. Projected climate surfaces were generated by overlaying climate change predictions from general circulation models (GCMs) as anomalies onto the 1961–1990 base-

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[3] Available from the Alberta Sustainable Development, Resource Data Division at www.srd.gov.ab.ca.

Table 1. Transition rules and disturbance types used in the disturbance model.

Ecosystem type	Acceptable transitions	Disturbance type
Boreal	Parkland (forested)	Fire
Foothills	Parkland (forested)	Fire
Alpine	Foothills or Parkland (forested)	Fire
Parkland (forested)	Grassland	Drought
Parkland (cleared)	Grassland	None required[a]
Grassland	None	Not applicable

[a] The transition to grassland does not require a disturbance because the trees have already been cleared.

line dataset (Mitchell et al. 2004). Because of uncertainties in forecasting CO_2 emissions and modeling climate, we used three GCM/emission scenario combinations that represent a median CGCM2-B2, a "pessimistic" hot/dry HADCM3-A1F1, and an "optimistic" cool/moist PCM-B1 scenario for Alberta. The CGCM2 (coupled general circulation model) is run by the Canadian Centre for Climate Modelling and Analysis, the HAD model is run by the Hadley Centre for Climate Prediction and Research in England, and the PCM (parallel climate model) is run by the National Center for Atmospheric Research in the United States. These were chosen from 20 available scenarios that we evaluated for Alberta based on the TYN-SC data sets of the Tyndall Centre for Climate Change Research (Mitchell et al. 2004).

Based on the 36 monthly temperature and precipitation variables, we generated spatial coverages of 15 biologically relevant and interpretable variables, including growing season precipitation and temperature, dryness indices, various degree-days, frost-free period, temperature extremes, and snowfall, according to Wang et al. (2006). For this study, we generated 1 km^2 resolution climate grids in Universal Transverse Mercator projection, covering the study area with approximately 660 000 tiles. We make the data sets and custom software developed for this study available on the University of Alberta Web site.[4]

Our approach to bioclimatic envelope modeling followed the approach described in Hamann and Wang (2006). Canonical discriminant analysis (Hand 1981) was used to identify the most relevant climate variables from among those available. Discriminant analysis maximizes the variance between groups (ecodistricts) and, therefore, indicates which of the original climate variables differentiate ecosystems best. The analysis was implemented through PROC CANDISC and PROC DISCRIM of the SAS Statistical Software Package (SAS Institute Inc. 2001). PROC DISCRIM assigns individual observations (tiles of multivariate climate data) to groups (ecodistricts), according to the Mahalanobis distance between an observation and the mean vector of the closest group, which may not necessarily be the original ecodistrict. Bioclimatic envelope projections were generated for each decade from 2000 to 2050. While we use ecodistricts as dependent variables because they are climatically homogeneous units, it should be kept in mind that they also represent topoedaphic variables, which do not change. We therefore summarize our predictions at a higher hierarchical level as discussed in the next section.

Disturbance modeling

Our starting point for modeling disturbance and associated shifts in vegetation was a base map derived from the provincial natural region map (Fig. 1*a*). The province has six natural regions within which the ecodistricts are nested. We did not use the ecodistrict map itself because robust and meaningful rules for disturbance and vegetation change can only be constructed at a much higher hierarchical level. Much of the parkland zone has been cleared for settlement and agricultural use. So we divided this zone into forested and agriculture categories, based on a MODIS land cover map (Friedl et al. 2002).

The disturbance model tracked the status of ecosystems at a resolution of 1 km^2 (matching the resolution of the BEM) in time steps of 1 year. Cells were initialized with values from the base map. At each time step, the model compared the current ecosystem type of each cell with the ecosystem predicted by the BEM for that time period (recall that the BEM is updated every 10 years). If the two values differed, a disturbance routine was initiated to update the status of the cell.

A set of rules served to limit ecosystem transitions to those felt to be ecologically most plausible and meaningful (Table 1). This primarily entailed restricting transitions to the dominant trajectory, as defined by the BEM and corroborated by paleoecological data and ecological studies (e.g., Hogg and Hurdle 1995; Strong and Hills 2003). If the ecosystem type predicted by the BEM was not permitted by the transitional rules, the cell would proceed to the next time step unchanged. An exception was made when the proposed transition was on a valid trajectory (as per Table 1) but an interim step had been omitted. In these cases the model forced an interim transition (e.g., a jump from boreal to grassland was forced to parkland).

We assumed that a cell would not undergo transition until the mature vegetation in it had been killed by a disturbance (Hogg 1994; Hogg and Schwarz 1997; Weber and Flannigan 1997; Flannigan et al. 2001). Disturbance was probabilistically applied to each cell at every time step, at a user-defined rate (e.g., if the rate was set at 2% per year, a given cell would be disturbed, on average, once every 50 years). Disturbance was applied independently to each cell (i.e., there were no contagion effects), and the probability of disturbance was the same for all cells of a given ecosystem type. If a simulated disturbance event occurred, the status of the cell would be updated as per the transition

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[4] ClimateAB software and databases downloadable at http://tinyurl.com/2l7jdp.

Fig. 1. Bioclimatic envelope maps for Alberta. (*a*) Base map for used for the simulations. (*b*) Model projections using historical climate data from 1960 to 1990. (*c–e*) Bioclimatic envelope model (BEM) projections for 2050 using three climate scenarios. (*f–h*) BEM projections for 2050 with the disturbance model activated (annual rate of fire = 0.14%; annual rate of drought = 4%).

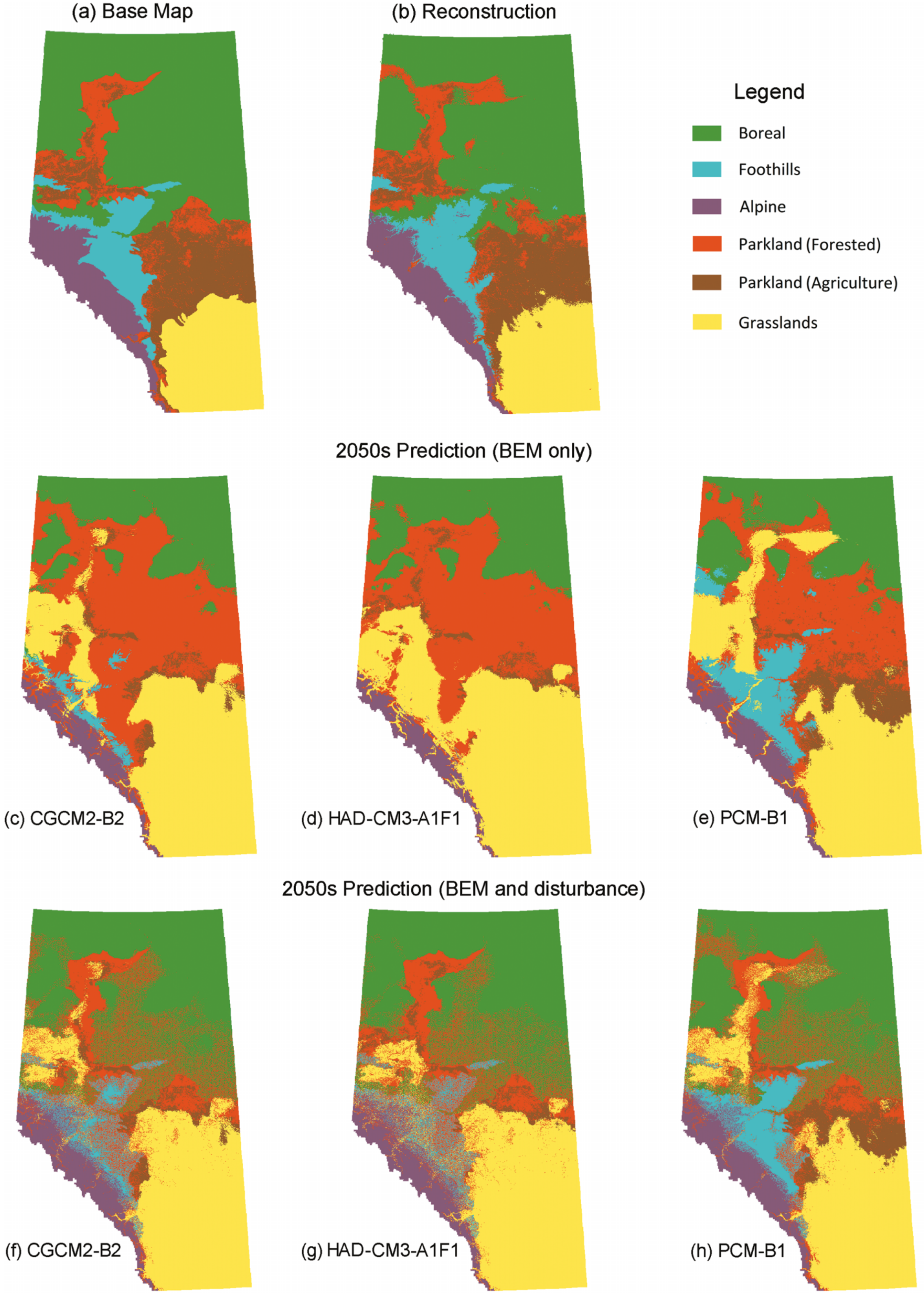

Fig. 2. Distribution of fires in Alberta, 1960–2006, in relation to the boreal and foothills zones.

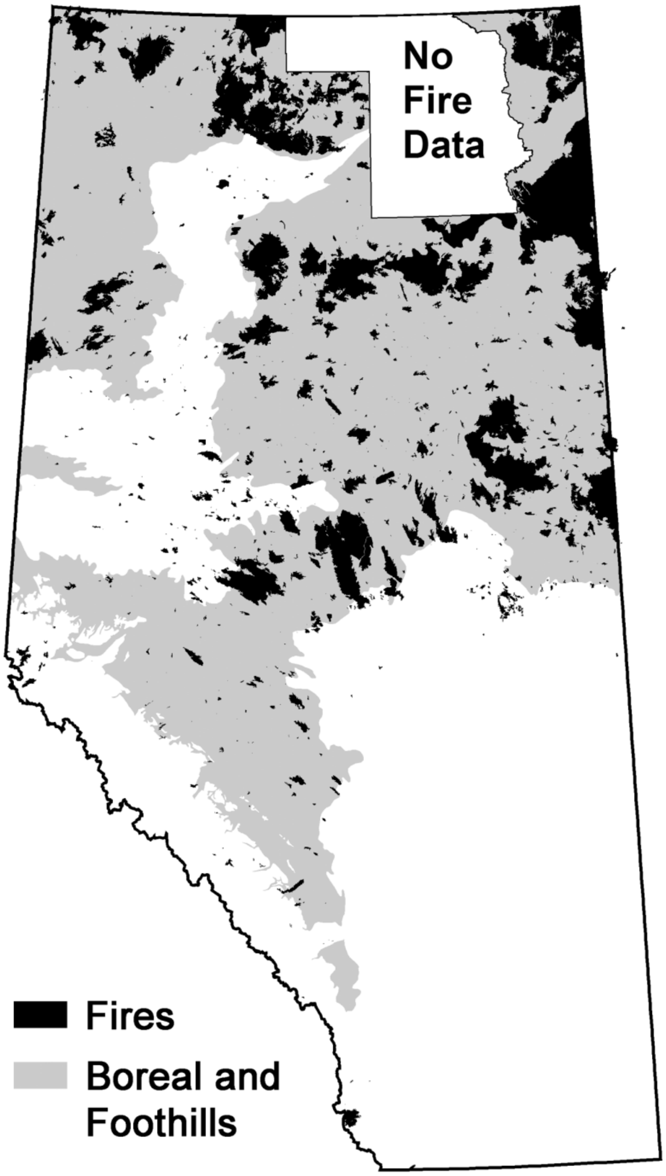

Table 2. Loadings of three canonical discriminant functions: 1961–1990 climate data.

Climate variable	Can 1	Can 2	Can 3
Mean annual temperature	**0.60**	–0.47	**–0.52**
Mean coldest month temperature	**0.97**	0.03	–0.13
Mean warmest month temperature	–0.07	**–0.78**	**–0.44**
Continentality (July–Jan. temp.)	**–0.86**	–0.46	–0.13
Mean annual precipitation (log)	0.17	**0.73**	0.09
Mean summer precipitation (log)	0.18	**0.66**	**–0.43**
Annual heat moisture index (log)	0.17	**–0.72**	–0.39
Summer heat moisture index (log)	–0.18	**–0.81**	0.07

Note: $\log_{10}$-transformed climate variables are designated as (log). Boldface values indicate correlation coefficients >0.90.

rules. Otherwise the cell would proceed to the next time step unchanged.

The disturbance rate parameter was linked to the dominant form of disturbance in each ecosystem type (Table 1). Fire was the obvious disturbance type to use for forested ecosystems in the boreal and foothill regions of Alberta, given the fire history of these regions (Fig. 2). The base rate was set to 0.7% per year, reflecting the provincial fire record for this region over the past 30 years.[5] The rate of fire in Alberta is expected to increase under global warming because of increased frequency of dry conditions, a longer fire season, and increased lightning strikes (Li et al. 2000; Flannigan et al. 2005; Tymstra et al. 2007). Based on fire projections by Flannigan et al. (2005), Tymstra et al. (2007), and Balshi et al. (2008), we also ran the model at a rate of 1.4% fire disturbance per year to bracket the upper extreme of what may occur in the future.

In the parkland region, the disturbance rate parameter was linked to drought. Hogg (1994) determined that the southern limit of the parkland in western Canada is a function of available moisture. The rate of fire in this region has been extremely low (Fig. 2). Published estimates of the rate of aspen (*Populus tremuloides*) mortality in response to drought are unavailable, so we sensitivity tested rates from 1% to 6% per year. The base rate was set at 2%–4% per year, based on unpublished data from T. Hogg.

Results

Three orthogonal canonical discriminant functions are required to account for most of the variance in climate variables among ecosystems in Alberta (Table 2). The first canonical discriminant function accounts for 51% of the total variation and reflects a gradient towards a more continental climate from Alberta's mountain ranges toward the northeast of the province. The northeast has colder winters and a larger difference between January and July temperatures. The second independent canonical discriminant function accounts for 26% of the variation in climate variables and reflects the storm track of the northern jet stream — a precipitation band that stretches across the province at approximately 56°N. The third function accounts for 12% of the variation and is important to reliably separate some ecosystem types that are cool and dry (or warm and dry) in summer from ecosystems that are cool and wet (or warm and wet) in summer. This function shows more complicated geographic patterns, separating the most extreme environments (high alpine, subarctic, and the driest grassland types) from similar but less extreme ecosystems. All other canonical discriminant functions can be dropped without affecting the predictions or misclassification error rates.

We tested the BEM's ability to recreate current ecosystem patterns, using historical climate data from 1960 to 1990 as input (Fig. 1*b*). The model correctly classified 88.3% of the 1 km^2 cells. Almost all of the misclassified cells were located at the interfaces between ecosystem types (compare Fig. 1*b* with Fig. 1*a*).

In the 2000–2050 simulation, large changes in the distribution of bioclimatic envelopes were observed (Figs. 1*c*–1*e*). With the disturbance model activated, much of the

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[5] Alberta fire history database, accessed 11 April 2008 from www.srd.gov.ab.ca/.

Table 3. Size of Alberta ecosystems in 2050 relative to the base map.

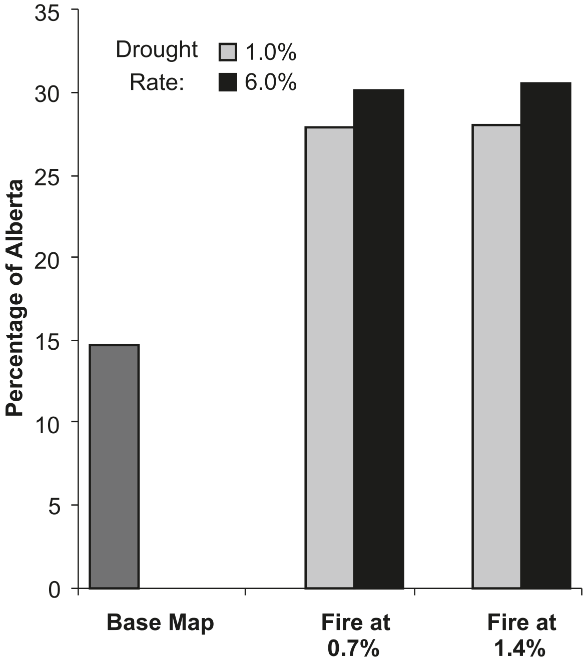
		CGCM slow[a]		CGCM fast[b]		Raw BEM	
Ecosystem type	Base map area (km^2)	Area (km^2)	Change (%)	Area (km^2)	Change (%)	Area (km^2)	Change (%)
Alpine	56 736	49 706	–12.4	44 601	–21.4	28 648	–49.5
Foothills	54 343	44 206	–18.7	36 864	–32.2	12 196	–77.6
Boreal	285 700	250 983	–12.2	225 194	–21.2	131 770	–53.9
Parkland	166 916	126 699	–24.1	156 891	–6.0	260 391	56.0
Grassland	97 121	189 222	94.8	197 266	103.1	227 811	134.6

Note: CGCM, coupled general circulation model; BEM, bioclimatic envelope model.

[a] Annual rate of fire = 0.7% and annual rate of drought = 2%.

[b] Annual rate of fire = 0.14% and annual rate of drought = 4%.

Fig. 3. Extent of grasslands in 2050 based on the CGCM2-B2 climate scenario.



BEM equilibrium pattern was realized by 2050, despite the introduced lags in ecosystem transition (Figs. 1*f*–1*h*). Most notable was an increase in the extent of grassland, which doubled in size under the CGCM2-B2 median climate change scenario (Table 3). The increase in grassland came largely at the expense of parkland, which in turn expanded into the boreal, foothills, and alpine regions, reducing their extent (Figs. 1*f*–1*h*). Changing the rate of disturbance had little effect on the patterns of change but did reduce the extent to which these patterns were realized (Table 3).

The vegetation patterns produced by the three climate scenarios were broadly similar, though some regional differences were apparent (Figs. 1*f*–1*h*). The largest discrepancy was in the foothills region where the PCM-B1 climate change scenario resulted in little change, whereas the CGCM2-B2 and HAD-CM3-A1F1 climate scenarios resulted in extensive conversion of foothills to parkland (Figs. 1*c*–1*e*). The dry/hot HAD-CM3-A1F1 climate projections were also notable in that they indicated the greatest conversion to grassland in the southern half of the province, but the least conversion to grassland in the northern half.

To determine the extent to which the transition rules modified the output maps, relative to the raw BEM projections, we tracked the instances where transitions were blocked or modified because they did not comply with the transition rules. For the 2040–2050 transition, where cumulative error should have been greatest, the rate of rejected transitions among the three climate scenarios ranged from 1.2% to 2.3% of cells. Instances where intermediate steps along valid trajectories were missed ranged from 2.3% to 6.1% of cells. The combined effect of the transition rules and lags introduced by the disturbance model was to reduce the differences among the scenarios, relative to the raw BEM projections (compare Figs. 1*c*–1*e* with Figs. 1*f*–1*h*).

Sensitivity testing of the drought parameter, which determined the rate of conversion of parkland to grassland, indicated that the rate of drought had minimal influence on the modeling results. A sixfold increase in the rate of drought resulted in only 8% more grassland in the CGCM2 model run (Fig. 3).

Discussion

Our findings can be compared with the results of two other studies that used alternative approaches for describing vegetation patterns in Alberta under a warmer climate. One study (Strong and Hills 2003) was based on paleoecological data from the Holocene hypsithermal period (lasting from 9000–6000 BP), when Alberta was 1–3 °C warmer. This study concluded that the current parkland and drier portions of the boreal were dominated by grassland during the hypsithermal. In our study, the parkland and drier portions of the boreal were the areas where conversion to grassland was greatest, suggesting that our trajectories of change were reasonable. However, the degree of change in our study was less than in the hypsithermal. This may have been due to the short duration of our simulation and to differences in the amount of warming.

The other study available for comparison used a Cumulative Moisture Index (CMI; precipitation minus potential evapotranspiration) to project vegetation patterns for the period 2041–2070 (Hogg and Bernier 2005). A CMI of less than –15 cm provides a good empirical fit with the current distribution of grasslands across western Canada, and a CMI between 0 and –15 cm provides a good fit with the current distribution of parkland. These relationships were used to predict the distribution of drought-stressed forests (i.e., CMI < 0) in the future, based on climate data from a GCM model (GCGM2-A21).

The future vegetation patterns generated by Hogg and Bernier (2005) were reasonably similar to ours in the southern Alberta but diverged in the north. Specifically, Hogg and Bernier found that changes in the boreal region occurred primarily in the northern half of the region, whereas we found that the southern half of the boreal to be most affected. It is an open question as to which approach has greater validity. The BEM uses much more of the available data when drawing statistical linkages between climate and vegetation, but the CMI approach incorporates a degree of causality, through documented moisture requirements for forest growth and regeneration (Hogg et al. 2002; Frey et al. 2004).

Our findings are subject to three forms of uncertainty. The first is uncertainty related to the projections of the GCMs. Although the sophistication of these models has increased in recent years, gaps remain in our understanding of long-term climatic patterns and how they will change in response to rising levels of greenhouse gases (Cane et al. 2006) That said, the three diverse GCMs used in this study produced vegetation patterns that were reasonably consistent at the provincial scale. This does not constitute proof that the patterns are robust, since the GCM models could share common flaws, but it provides some assurance that choice of GCM is not a critical factor in discerning large-scale patterns.

The second source of uncertainty is related to the projections of the BEM. The BEM was able to recreate existing vegetation patterns reasonably well using historical climatic data, as evidenced by a low rate of misclassification error. But again, this does not constitute proof that our projections of future vegetation patterns are robust. The statistical relationships forming the basis of the BEM are not the same as causal relationships. They may not hold once climatic and ecological parameters begin to shift in response to global warming (Woodward and Beerling 1997).

The third source of uncertainty in our study is related to the disturbance model used to limit the rate of transition. Uncertainty here did not influence the vegetation patterns per se, but the degree to which they were manifested over the 50 year time frame of the simulation. Though we believe we have captured the essence of the system — transition driven by disturbance — the results should not be considered more than a first approximation.

Most of the uncertainty in the disturbance model can be attributed to the simplifying assumptions that were used. For example, we assumed that direct mortality of mature trees from warming temperatures and related moisture stress could be discounted. The ability of mature trees to survive well outside their usual climatic zone is demonstrated in farm shelter belts and island populations in many parts of the Canadian prairies (Hogg and Schwarz 1997; Chhin and Wang 2002).

We also assumed that sites would regenerate to a new vegetation type if the bioclimatic envelope shifted and the mature trees were killed by a disturbance. We believe it was reasonable to assume that transition would be rapid after disturbance because the bioclimatic envelope changes occurred in landscapes that were, for the most part, already transitional in nature (as evidenced by inventory data). For example, the conversion of boreal forest to parkland occurred primarily in boreal mixedwood forests that contained substantial amounts of aspen. In effect, the transitions in our simulation involved changes in the relative abundance of local species, not the influx of new species through long-distance migration. Schindler (1998) and Hogg and Wein (2005) provide examples of how such transitions might occur in a boreal setting. Beyond 2050, as bioclimatic envelope changes involve landscapes progressively farther from current natural region interfaces, migration lags may become an important factor (Weber and Flannigan 1997).

Because of the high degree of stochasticity associated with disturbance processes, we assumed that the size and location of future disturbances could not be meaningfully predicted (Armstrong 1999). Instead, we pursued a probabilistic approach. We modeled disturbance at a very fine scale to produce maps that, at the provincial scale, would provide a probabilistic indication of where and to what extent shifts in vegetation are likely to occur, reflecting the intersection between changing climatic envelopes and average likelihood of disturbance. The message in our maps is conveyed through the intensity of color change; the status of individual cells is not meant to be discerned at the provincial scale.

The parameterization of the disturbance model also contributed to uncertainty in the results. Though various forms of disturbance exist, we were only able to quantify the dominant form of disturbance for each ecosystem type. We attempted to reduce the uncertainty related to the disturbance rate parameter through sensitivity testing. For example, though the rate of aspen mortality due to drought has not been well quantified, sensitivity testing showed the model was relatively insensitive to this parameter. Most of the parkland, upon which the drought parameter acted, has already been cleared for agriculture and settlement. The rate of fire had a greater influence on model results; however, the range of potential fire rates has been reasonably well bounded (Flannigan et al. 2005; Tymstra et al. 2007; Balshi et al. 2008), and it is reasonable to assume that the future rate has been captured within the range examined in this study. That said, a rate of fire near the high end of our estimates in combination with a high rate of disturbance from other causes could result in an overall rate of disturbance higher than what we modeled.

The foothills stands out as the region with the highest degree of uncertainty. The climate projections for this region were inconsistent, and a potentially major disturbance agent, mountain pine beetle, was not modeled. Although the likelihood of a pine beetle epidemic appears high, the timing of such an epidemic cannot be reliably predicted (Carroll et al. 2006). If a massive beetle outbreak occurs soon, a well-established regenerating forest that is relatively resistant to shifts in bioclimatic envelope may be in place in 2050. However, if the beetle outbreak is delayed for a few decades, to the time that bioclimatic envelopes have begun to shift, the outbreak may result in a rapid transition to a completely new vegetation type. Additional research will be needed before reliable conclusions concerning vegetation changes in the foothills can be made.

Despite the inherent uncertainties, our study provides information of use to land managers. One of the main insights is that even with transition lags in place, the potential exists for large shifts in the distribution of Alberta's biomes over the next 50 years. A northward shift of grasslands into much of the existing parkland appears particularly likely. There is a high degree of consistency among the climate models on this point. The change won't be a transition in biome per se, because most of the parkland has already been cleared for settlements and agricultural use. Instead, it will involve a management shift to agricultural crops and practices suitable for summers that are significantly hotter and drier than is currently the case, with an increased frequency of drought conditions (Marchildon et al. 2008).

It is also likely that a substantial portion of Alberta's boreal region will be converted to parkland over the next 50 years (12%–21%, Table 3). Today, large burns still regenerate to mixedwood forest, through a combination of natural regeneration of hardwoods and the planting of conifer seedlings in softwood stands that have been salvaged. Once bioclimatic envelopes begin to shift, however, regeneration failures are likely to occur (Hogg 1994; Hogg and Schwarz 1997; Hogg and Wein 2005), setting the stage for a change to parkland.

Increased fire suppression and augmented forest regeneration efforts could slow or possibly halt the shift to parkland. But the business case for additional industry expenditures is currently weak, given that rising energy and labour costs are threatening the forestry sector's ability to generate income in Alberta (Alberta Forest Products Association 2008). There may also come a time when it becomes too expensive for the government to continue suppressing fires everywhere in the province. Over the past 5 years an average of Can$225 million/year was spent on wildfire management in Alberta. To put this into perspective, average provincial revenues from timber royalties and fees were Can$88 million/year during this period (Alberta Sustainable Resource Development 2008).

According to our simulations, boreal transitions will initially occur in the forests adjacent to the current parkland and spread outwards from there. These findings represent a general tendency, in that the stochastic nature of fire was averaged out in our simulations. The actual pattern that manifests in coming years will depend on the size and distribution of future fires. In Alberta, fires over 100 km^2 account for approximately 72% of the area burned, providing some indication of the anticipated size of transition patches.[6] Because of variability in local site conditions and local climate, it is unlikely that transitions will be homogeneous within the areas that are burned.

Knowledge of general trends can be of value to land managers in various ways. Setting land-use priorities, designating protected areas, and developing species recovery plans are a few examples where an understanding of how the landscape may potentially change in coming decades would be useful (Noss 2001; Scott and Lemieux 2005). Though our projections are subject to uncertainty, they can help managers bound the range of possible futures in the context of scenario modeling (e.g., Schneider et al. 2003).

Our study also serves to identify fruitful avenues of future research. Our approach of using a disturbance model to quantify transition lags appears to be a workable solution to addressing some of the inherent deficiencies of BEMs (Pearson and Dawson 2003). This approach could be refined by improving the model's transition processes and reducing the uncertainty in parameter estimates. Attention should also be directed towards improving our understanding of how vegetative communities will respond to changes in climate. A study aimed at documenting the distribution and characteristics of early transitions would be of great value. This information would be useful for validating and improving the transition model and would also serve to test the statistical associations underlying the BEM model. If early transitions to parkland are dispersed across the boreal or are concentrated in the north as predicted by Hogg and Bernier (2005), it would call our BEM into question. Continued research along these lines will also aid the development of dynamic vegetation models, which should eventually become powerful tools for land-use planning and climate adaptation efforts (Peng 2000).

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