

## Moving beyond bioclimatic envelope models

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# **Moving beyond bioclimatic envelope models: integrating upland forest and peatland processes to predict ecosystem transitions under climate change in the western Canadian boreal plain**

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## **ABSTRACT**

By the end of this century, much of the climate space of western Canada's boreal forest is expected to shift northwards and be replaced by climates that are currently associated with aspen forest, parkland, and grassland ecosystems. In this study we review the various processes that will mediate ecological responses to these projected changes in climate. We conclude that ecological transitions are unlikely to involve a gradual wave-like shift in ecotonal boundaries. Instead, we predict that ecological changes will lag substantially behind changes in climate and that individual ecosystem components will respond at different rates. In particular, if precipitation inputs are maintained as expected, then peatlands should exhibit considerable resilience to climate change and remain a dominant feature on the landscape in 2100. Because peatlands retain large amounts of water on the landscape their continued presence may in turn slow the rate of forest loss, especially the aspen component. Thus, ecological response to climate change in the western boreal region may involve a transition to a novel ecosystem that includes peatlands and aspen as dominant features — unlike anything that exists today. Moreover, this interim stage may remain in place well into the next century, potentially providing additional time for forest-dependent species to adapt.

## **KEY WORDS**

Boreal forest; peatlands; bioclimatic envelope models; climate change

## INTRODUCTION

In recent years there has been a growing interest in understanding how regional ecosystems will respond to global warming (Price *et al.*, 2013; Iversen *et al.*, 2015). A common approach involves the use of bioclimatic envelope models (BEMs), which statistically characterize the climate space, or “envelope”, of individual ecosystems based on current eco-climatic associations (Pearson and Dawson, 2003; Hamann and Wang, 2006). Future ecosystem distributions can be predicted by using BEMs in conjunction with climate projections from General Circulation Models (GCMs). The working assumption is that ecosystems will track their climate envelopes as the envelopes shift across the landscape in coming decades.

BEM projections represent long-term equilibrium outcomes, and do not take into account the time it will take for ecosystems to respond to climatic change, or the processes involved. Ecological lags are expected to occur, and are an important consideration in the context of adaptation (Gray and Hamann, 2013). If the ecological transitions are rapid the intrinsic capacity of species for adapting to change may be exceeded (Malcolm *et al.*, 2002; Loarie *et al.*, 2009). This in turn is likely to influence the nature of management responses (Loarie *et al.*, 2009; Hamann and Aitken, 2013). For example, under a scenario of rapid change, assisted migration may be needed to ensure that species with low intrinsic mobility are able keep pace with shifts in their preferred environmental conditions (Hewitt *et al.*, 2011).

In boreal forest systems, much of the research into climate-related ecological transitions and lag effects has been compartmentalized, focusing on either the upland forest or peatland component. In upland forests, natural disturbances such as fire and insect outbreaks have been identified as critical factors influencing the rate of change (Malmstrom and Raffa, 2000; Dale *et al.*, 2001; Woods *et al.*, 2010). In peatlands, negative feedback processes that minimize water losses during dry conditions have been highlighted (Kettridge and Waddington, 2014; Waddington *et al.*, 2015). In this study we explore how these processes may work in concert to modify the predicted state of the boreal ecosystem in 2100 relative to baseline BEM projections. More generally, we hope to illustrate the need for future collaboration among terrestrial and peatland research groups in order to fully understand boreal responses to climate change.

## BIOCLIMATIC ENVELOPE MODELLING

Our study area is limited to the plains of Alberta, Canada, from the Dry Grassland ecosystem in the south to the Boreal Plain ecosystem in the north (Fig. 1). Level to gently undulating plains are the dominant landforms across the study area, with a minor inclusion of hummocky landscapes. Surficial geology is primarily glacial in origin, including clay-rich till, fine-grained lacustrine deposits, and coarse-grained glaciofluvial and glaciolacustrine deposits (Fenton *et al.*, 1994).

Our primary interest is the Boreal Plain ecosystem (~168,000 km<sup>2</sup>), and the potential changes in composition that may result from climate change (Fig. 1; Boreal Plain is synonymous with the

Boreal Central Mixedwood Natural Subregion in the provincial ecosystem classification). Upland areas, comprising 52% of the Boreal Plain, are characterized by a mosaic of pure and mixed stands of aspen, white spruce, and jack pine (Natural Regions Committee, 2006). Extensive wetlands, primarily fen and bog peatlands, are found in the poorly drained areas that comprise 45% of the region. A variable woody cover of black spruce and/or tamarack is common within the peatlands. Open water accounts for the remaining 3%. Wildfire is the dominant form of natural disturbance, burning approximately 0.5 percent of the Boreal Plain per year, on average over the last 50 years.<sup>1</sup> Linear disturbances, including roads, seismic lines, and pipelines, are found in moderate to high densities across much of the region (Schneider *et al.*, 2003).

Our approach to bioclimatic envelope modelling was based on methodology developed by Hamann *et al.* (2006) and extended in subsequent studies (Schneider *et al.*, 2009; Mbogga *et al.*, 2010; Roberts and Hamann, 2012). We obtained the historical climate data for construction of the BEM, downscaled to a 1 km<sup>2</sup> resolution, from the ClimateWNA model, for the 1961-1990 period (Wang *et al.*, 2012). The available climate variables included mean annual temperature, mean annual precipitation, mean warmest monthly temperature, mean coldest monthly temperature, seasonality, growing degree-days above 5 °C, and a measure of available moisture. Of these variables, available moisture has the greatest effect on determining current vegetation patterns (Hogg, 1994; Hogg and Bernier, 2005).

We statistically associated the climate variables with the following ordered set of ecosystem types (i.e., the dependent variables in our model), using an ordinal regression model in the R programming language (R Core Team, 2012): Dry Grassland → Fescue Grassland → Parkland → Aspen Forest → Boreal Plain. Ordinal regression was used because in western Canada, boreal forest, parkland, and grassland ecosystems constitute an ecocline that is responding to a climate gradient (Hogg, 1994; Hogg and Hurdle, 1995). The ecosystem types and their spatial distribution were derived from the provincial Natural Subregion classification, with some lumping of smaller units (Fig. 1; ecosystem names are simplified from the original to improve readability).

To predict equilibrium ecosystem distributions in future periods we used our BEM in combination with climate projections from the GCMs and CO<sub>2</sub> emission scenarios in the Intergovernmental Panel on Climate Change Fifth Assessment (Van Vuuren *et al.*, 2011). These data were again obtained from the ClimateWNA model at a 1 km<sup>2</sup> resolution (Wang *et al.*, 2012). The projected increase in mean annual temperature for the Boreal Plain in the 2080s, averaged across all GCMs, is 3.7 °C. The mean change in annual precipitation over the same period, across all GCMs, is an increase of 6.7%. We present here, as a representative example, the BEM projections for the ensemble mean of all available GCMs running a median (RCP 4.5) CO<sub>2</sub> emission scenario (Van Vuuren *et al.*, 2011; Wang *et al.*, 2012). BEM projections for individual GCMs are available in Schneider (2013).

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<sup>1</sup> Based on data in the Alberta Historical Wildfire Database: [www.wildfire.alberta.ca](http://www.wildfire.alberta.ca)

The 2080s BEM projection for the GCM ensemble mean indicates that virtually all of the climate space currently occupied by the Boreal Plain will be replaced by climates currently associated with Parkland and Fescue Grassland ecosystems (Fig. 2). The Boreal Plain climate envelope will in turn move upslope, into the boreal hill system, and northwards. Alberta's Boreal Plain is predisposed to this transition because it is near the climatic tipping point that separates forested from non-forested systems (Hogg, 1994; Hogg and Hurdle, 1995, Iverson *et al.*, 2015). Precipitation inputs are relatively low — similar to those in the Aspen Forest and Parkland — so it is only the low temperature and associated low rate of evapotranspiration of the Boreal Plain that accounts for its forested status. This is supported by paleontological evidence of boreal to parkland transitions during the warm Hypsithermal period, approximately 6,000 years ago (Strong and Hills, 2005; Williams *et al.*, 2009). In addition, it was not until after the Hypsithermal that climatic conditions became suitable for peatland initiation in the southern portion of our study area (Zoltai and Vitt, 1990).

## PEATLAND DYNAMICS

Moving from south to north in Alberta, there is a gradual increase in the proportion of wetlands, reflecting lower temperatures, lower rates of evapotranspiration, and a small increase in precipitation (Fig. 3). Transitioning to the Boreal Plain, climate variables continue their linear trend, but there is an abrupt threefold increase in the occurrence of wetlands (Fig. 3). This nonlinear pattern of change can be attributed to the special properties and feedback processes exhibited by peatlands, which comprise most of the wetlands in the Boreal Plain (Larson, 1995; Devito *et al.*, 2012; Waddington *et al.*, 2015). Thus, predicting boreal wetland responses to climate change, including the potential for ecological lag effects, requires an understanding of the special nature of peatland dynamics.

Within peatlands the rate of production normally exceeds the rate of decomposition, resulting in the slow accumulation of biomass over time (Bauer *et al.*, 2003). The large accumulated mass of peat acts like a sponge, storing water accumulated from melting snow and summer rains. Water movement within the compressed lower layers of slowly decomposing peat is very low, limiting the loss of water into the underlying ground (Devito *et al.*, 2005; Waddington *et al.*, 2015). In addition, under dry conditions, the surface layer of peat desiccates, and functions like a layer of mulch to reduce water losses from evapotranspiration (Price *et al.*, 2009; Kettridge and Waddington, 2014; Waddington *et al.*, 2015). As a result of these negative feedback processes peatlands retain large volumes of water on the landscape, and they are inherently resilient to climatic fluctuations (Devito *et al.*, 2012; Waddington *et al.*, 2015).

The effect that rising temperatures will have on peatlands in coming decades will depend in large part on their level of water saturation. If the local water table declines for a prolonged period, oxygen is introduced into the upper layers of peat, permitting efficient aerobic decomposition to occur in place of the slow anaerobic process that normally predominates (Bhatti *et al.*, 2006; Sulman *et al.*, 2009; Fenner and Freeman, 2011). Furthermore, the combination of drought and high temperatures can result in reduced productivity or mortality

of the upper living layer of moss, limiting the replacement of peat lost to decomposition (Gignac *et al.*, 2000; Bragazza, 2008). Conversely, if a peatland remains well hydrated, then any increase in the rate of decomposition resulting from higher temperatures is likely to be offset by an increase in the rate of production (Moore, 2002; Charman *et al.*, 2012; Loisel and Yu, 2013). Furthermore, several studies have concluded that decomposition within deeper layers of peat is limited by the accumulation of phenolic breakdown products, not necessarily low temperature (Beer and Blodau, 2007; Beer *et al.*, 2008; Morris and Waddington, 2011).

Rising temperatures are also expected to result in an increased rate of fire in the Boreal Plain (Balshi *et al.*, 2009; de Groot *et al.*, 2013). The impact on peatlands that remain well hydrated may be limited, because fires typically consume only the top few centimetres of peat under such conditions (Zoltai *et al.*, 1998; Benschoter *et al.*, 2011). Moreover, cumulative losses will be offset, to a degree, by vegetation regrowth over the fire cycle (Wieder *et al.*, 2009; Thompson and Waddington, 2013). In contrast, deep drying of peat can result in substantial losses from wildfire, because smouldering combustion is capable of consuming all of the peat layers that are not fully saturated (Zoltai *et al.*, 1998; Benschoter *et al.*, 2011; Turetsky *et al.*, 2011). Metre-deep fires under such conditions have been reported anecdotally, especially in small isolated peatlands systems (Zoltai *et al.*, 1998; Lukenbach *et al.*, 2015).

Climate change may also lead to a contraction of peatlands through peripheral ingress of terrestrial vegetation (Berg *et al.*, 2009; Murphy *et al.*, 2009). Shrub invasion occurs when there is a drop in the water table, providing conditions suitable for the establishment of non-aquatic species (Gignac and Vitt, 1994; Turetsky *et al.*, 2011). Increased evapotranspiration from these plants can result in a cycle of ever-faster peatland drying (Heijmans *et al.*, 2013). In cases where the drop in the water table is prolonged, the rate of change can be quite rapid — peatlands in Alaska's Kenai Peninsula have contracted by up to 11% per decade as a result of shrub ingress since the 1950s (Berg *et al.*, 2009). In other cases, the ingress of shrubs may be disrupted by periodic flooding, because of the vulnerability of terrestrial vegetation to inundation.

The black spruce trees that already exist within many peatlands also need to be considered. Under current conditions, the trees are generally stunted and usually provide less than 25% cover. Moreover, low oxygen in the rooting zone tends to suppress photosynthesis and the transpiration rate of black spruce trees, such that they are a minor contributor to the water balance (Dang *et al.*, 1991; Thompson *et al.*, 2014). The critical question is, as temperatures rise, will the aforementioned negative feedbacks within peatlands keep the water table high enough to maintain low transpiration in the black spruce? Or will the water table drop enough that productivity and transpiration of black spruce increase, resulting in further drying and a strong positive feedback cycle (Dang and Lieffers, 1989; Choi *et al.*, 2007)? This, together with the future impact of fire within black spruce systems, remain key uncertainties at present.

The dependence of the aforementioned processes on the state of peatland hydration implies that peatland responses to climate change will largely be mediated by changes in the water table (Turetsky *et al.*, 2011; Gong *et al.*, 2012; Sherwood *et al.*, 2013; Kettridge *et al.*, 2015).

The response may be rapid if the change in climate involves a direct reduction in precipitation, as has been the case in Alaska (Klein *et al.*, 2005; Berg *et al.*, 2009). Substantially longer lag times can be expected in areas such as Alberta, where precipitation inputs are expected to remain stable or slightly increase, partially offsetting increased evapotranspiration from rising temperatures.

In Alberta, changes in the water table are likely to be complex, reflecting both spatial and temporal variability in the net water balance. The Boreal Plain is large, and precipitation inputs in the future will vary across the region, as they do now. Furthermore, potential evapotranspiration varies among ecosystem components. The high resistance to evapotranspiration losses exhibited by peatlands is not shared by upland forests or open water (Petrone *et al.*; 2007, Wang *et al.*, 2014). With warmer temperatures and a longer ice-free period, these areas will export increasing amounts of water into the atmosphere, depleting groundwater storage and, to a varying degree, drawing water from adjacent peatlands (Ferone and Devito, 2004; Petrone *et al.*, 2008; Devito *et al.*, 2012; Wang *et al.*, 2014). Within peatlands themselves, water table responses are also likely to vary with peatland type, amount of tree cover, surficial geology, and topographic location (Devito *et al.*, 2005).

Temporal variability also needs to be taken into account, especially with respect to precipitation. Precipitation in the Boreal Plain is characterized by multi-year periods of below average precipitation interspersed with short periods of greatly increased precipitation (Mwale *et al.*, 2009). This interannual variability is expected to increase under global warming (Mladjic *et al.*, 2011; Cohen *et al.*, 2014). Thus, in addition to a long-term trend of increased water loss from increased evapotranspiration, there exists the potential for multi-year droughts, some of which are likely to be severe (Michaelian *et al.*, 2011; Wang *et al.*, 2014). Water table declines within peatlands could be substantial during prolonged droughts, resulting in substantial losses of peat through multiple mechanisms (Ise *et al.*, 2008; Zhaojun *et al.*, 2011).

## **UPLAND FOREST DYNAMICS**

The dominant vegetation of Parkland and Fescue Grassland ecosystems includes aspen and various grass species. The proportion of aspen is a function of available moisture, and varies from closed forests in moist areas to isolated groves in dry areas (Natural Regions Committee, 2006). White spruce is uncommon and is limited to moist sites, such as north-facing riparian slopes. Jack pine can be found in areas with sandy soils, but is also generally uncommon.

Given these vegetation patterns, equilibration of Alberta's Boreal Plain with a Parkland or Fescue Grassland climate would entail the loss of most of the white spruce component of upland forests (Hogg, 1994; Hogg and Hurdle, 1995). Patches of grassland would also appear and enlarge over time, largely at the expense of aspen forest, with the amount of change depending on the degree of warming that occurs. The ability of Jack pine to grow on xeric sites suggests that it might persist on sandy soils for an extended period.



As with peatlands, ecological transitions within upland forests are expected to lag behind changes in climate, though different processes are involved (Schneider *et al.*, 2009; Gray and Hamann, 2013; Price *et al.*, 2013). In forests, resistance to change is largely a consequence of the intrinsic tolerance of mature trees to variability in climatic conditions (Hogg and Schwarz, 1997; Lieffers *et al.*, 2001; Gray and Hamann, 2013). Disturbances that result in widespread mortality of mature trees, such as wildfire, are therefore a key factor influencing the rate of ecological transition (Hogg and Wein, 2005; Schneider *et al.*, 2009). The high spatial and temporal variability in the distribution of large fires and other natural disturbances implies that ecological transitions will be patchy in nature, rather than occurring as a progressive wave of change (Schneider *et al.*, 2009).

In white spruce, vulnerability to transition is highest during the regeneration phase because hot and dry conditions can result in poor seed germination and high seedling mortality (Nilsson and Orlander, 1995; Hogg and Schwarz, 1997; Hogg and Wein, 2005). Also, if the rate of fire is very high, spruce trees may not be able to mature fast enough to produce cones before being killed in a subsequent event.

Aspen is less vulnerable than spruce to stand transitions following fire because the root system generally remains viable and is capable of regenerating the stand through suckering, even when climatic conditions are suboptimal (Lieffers *et al.*, 2001). Transitions within aspen stands are therefore more likely to occur as result of clonal death from drought, augmented by attack from insects and disease (Frey *et al.*, 2004; Hogg *et al.*, 2008; Michaelian *et al.*, 2011).

The spread of grasses in the Boreal Plain will require little more than reduced competition from trees. Small natural grasslands are already present along the Peace River lowlands, from west-central Alberta all the way to Wood Buffalo National Park in the northeast (Moss, 1952; Schwarz and Wein, 1997). In addition, grass (often agronomic species) has been planted along the thousands of kilometres of roadsides, pipeline right-of-ways, and seismic lines that permeate the Boreal Plain (Schneider *et al.*, 2003). These grasses have been unable to invade intact forest stands under current conditions, but grass expansion is expected once climatic conditions tip the competitive balance in their favour (Sumners and Archibold, 2007; Thorpe, 2011).

## **ALBERTA'S BOREAL FOREST IN 2100: A NOVEL ECOSYSTEM**

A limitation of BEMs is that ecosystems must be treated as discrete units. This means that ecological responses to climate change are portrayed simply as spatial shifts in ecotone boundaries. For example, the leading edge of the Aspen Forest ecosystem is projected to move like a wave across the entire Boreal Plain, followed by a second wave of change involving the Parkland ecosystem (Fig. 2).

Our review of ecosystem dynamics and transitional processes suggests that ecosystem responses to climate warming will be far more complex than the simple 'wave' model

portrayed by BEMs. In particular, we expect that ecological responses will lag significantly behind climatic changes and that individual ecosystem components will respond to warming in different ways and at different rates. As a result, Alberta's Boreal Plain in 2100 is unlikely to be comprised of a mosaic of Parkland, Aspen Forest, and remnant Boreal Plain ecosystems, as might be expected on the basis of BEM projections. Rather, most of the region is likely to be represented by a novel ecosystem displaying a combination of individual ecological elements not found in any existing ecosystem.

We hypothesize that the dominant features of this novel ecosystem will be peatlands and aspen forest. The rapid contraction of peatlands that has occurred in Alaska over the last 50 years cannot be extrapolated to Alberta because the Alaska declines have largely been caused by a long-term reduction in precipitation (Klein *et al.*, 2005; Berg *et al.*, 2009). Precipitation inputs in Alberta are expected to remain stable or increase, and under these conditions the feedback mechanisms intrinsic to peatlands will promote water retention and stability (Gong *et al.*, 2012; Kettridge and Waddington, 2014; Waddington *et al.*, 2015). Furthermore, the legacy of several thousands of years of peat accumulation (Halsey *et al.*, 1998), covering 45% of Alberta's Boreal Plain, cannot quickly be unwound. Nevertheless, some decline in their extent can be anticipated by 2100 because the ability of peatlands to resist change and remain isolated from the progressive drying of other ecosystem components is not absolute.

The peatland changes that do occur will be concentrated where susceptibility is highest. At the local scale, changes will likely be most pronounced at the peatland-forest interface, including swamp wetlands and ephemeral draws. The conditions for maintaining peat are least favourable here, because peat thickness and water table depth are low relative to the centre of the peatland, and the potential for the ingress of terrestrial vegetation is high (Bhatti *et al.*, 2006; Berg *et al.*, 2009). Initially, these transitions may be periodically reset by cyclical wet periods, which can cause inundation and mortality of ingressing terrestrial vegetation.

At the landscape scale, variability in susceptibility among wetland types is an important factor (Bauer *et al.*, 2003; Devito *et al.*, 2012). Ponds and lakes without underlying peat are likely to contract relatively quickly as a result of rising evapotranspiration. Fens dependent on groundwater inflow could also contract, and conversion to bogs may occur (Camill *et al.*, 2009; Granath *et al.*, 2010). Location relative to local topography will also play a role; peatlands in perched or isolated basins will have greater susceptibility to drying than peatlands in depressions or those with the potential for external water inflows, even if only periodic (Devito *et al.*, 2005). At the regional scale, peatland transitions may be influenced by the distribution of future fires and droughts. The high stochasticity of these events, especially the large ones, may result in a somewhat patchy and random pattern of change. That said, overall cumulative risk of transition will be higher in areas of the Boreal Plain that experience a Parkland climate at an early stage, relative to areas that are affected later.

Our expectation that aspen will also remain a dominant ecological feature of the Boreal Plain in 2100 is based on the demonstrated tolerance of aspen for Parkland climates and the ability of aspen to regenerate through asexual suckering after fire (Lieffers *et al.*, 2001). The grassland

climatic conditions that would likely cause significant declines in aspen are not anticipated to occur until late in this century, and are unlikely to affect all parts of the Boreal Plain, at least under the median CO<sub>2</sub> emission scenario we examined (Fig 2). As with peatlands, spatial variability in susceptibility to transition can be expected, linked to site-specific factors such as slope, aspect, and soil type.

Another factor promoting the retention of aspen is the interrelationship between aspen stands and peatlands. If peatlands persist under a Parkland climate as expected they may serve to support aspen by providing supplemental water, particularly during periods of drought. Aspen is a clonal species and exhibits a highly interconnected root system that permits the sharing of water and other resources (DesRochers and Lieffers, 2001; Snedden, 2013). Devito *et al.* (2012) propose that this 'root pipeline' can move water from peatlands into aspen stands, sustaining the forest through climatic cycles. The implication is that, under higher levels of warming, aspen stands adjacent to peatlands may persist, whereas stands in areas of low moisture, such as hilltops and south facing slopes, may decline and eventually convert to grass. The stands that persist may come to serve as refugia for forest biota, warranting extra attention in the context of proactive conservation planning.

In contrast to aspen, we expect that white spruce will steadily decline once the Boreal Plain begins to experience a Parkland climate, largely as a consequence of vulnerabilities associated with regeneration (Nilsson and Orlander, 1995; Hogg and Schwarz, 1997; Hogg and Wein, 2005). Given the tolerance of mature spruce trees to climatic variability, the rate of white spruce decline will largely be a function of the rate of fire and other causes of adult mortality (Hogg and Wein, 2005; Schneider *et al.*, 2009). As a rough approximation, if we assume that the average rate of burning will double (Balshi *et al.*, 2009) to 1% per year, and that white spruce transitions will only occur after exposure to a Parkland climate (as per our sequential BEM projections), then we could expect almost half of the existing spruce to be lost by 2100.<sup>2</sup> This may be an underestimation because mortality from insects, disease, and drought are not included in this calculation. On the other hand, it is unlikely that regeneration will consistently fail under a Parkland climate, given natural interannual variability in precipitation patterns.

The long-term trajectory of white spruce will also be affected by silvicultural activities. Hand planting, and replanting when necessary, minimizes the vulnerability of spruce to regeneration failure. Assisted migration of genetic stock from warmer regions may also facilitate persistence under warmer and drier conditions (Pedlar *et al.*, 2011). The problem is that the rate of artificial planting may not be sufficient to make much of a difference at the regional scale (Nelson *et al.*, 2011; Zielke *et al.*, 2012). The infrastructure for providing planting stock is designed to provide a steady supply of seedlings at a rate matched to the local rate of forest harvesting. The system is not designed to deal with replanting needs associated with massive fires, which occur only sporadically but account for most of the area burned. Economic constraints may also become a limiting factor, if the business case underlying planting is

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<sup>2</sup> Numerical calculation assuming that fires are random independent events; historical fire occurrence derived from the Alberta Historical Wildfire Database: [www.wildfire.alberta.ca](http://www.wildfire.alberta.ca)

jeopardized by an increased risk of fire, drought and insect damage (Johnston and Williamson, 2005; Woods *et al.*, 2010).

In conclusion, our review of transitional processes suggests that the outcomes predicted by BEMs for Alberta's Boreal Plain ecosystem are unlikely to be realized until well beyond 2100. Furthermore, rather than a gradual wave-like transition from Boreal Plain to Parkland/Grassland, we propose that a novel ecosystem (i.e., having no analogue among existing Natural Subregions) will develop as an interim stage and remain in place for an extended period (Hobbs *et al.*, 2009). Because peatlands retain large amounts of water on the landscape, and because they are resistant to change, peatlands may play an important role in slowing the rate of forest loss. Nevertheless, the high structural and species diversity characteristic of Alberta's boreal mixedwood forest (Stelfox *et al.*, 1995) will inevitably decline as white spruce is progressively lost from the system. A prolonged transition phase will at least provide forest-dependent species additional time to shift their ranges and otherwise adapt to changing climatic conditions.

The state of Alberta's Boreal Plain past 2100 is difficult to predict. If CO<sub>2</sub> emissions follow the median trajectory that we have explored in this report, then the eventual transition to a Parkland or Grassland system appears inevitable. However, this fate is not sealed. If ecological transitions are as slow as we anticipate, there may still be time to forestall major losses of forest through aggressive efforts to control CO<sub>2</sub> emissions. The fundamental question then is whether human society will respond to the effects global warming faster than the peatlands and forests of the Boreal Plain.

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## FIGURE CAPTIONS

**Fig. 1.** Study area, showing the major ecosystem types used in the analysis. Derived from the provincial Natural Subregion classification (Natural Regions Committee, 2006), with some categories combined and with simplified labels.

**Fig. 2.** BEM projections for the Alberta plains based on the ensemble mean of all available GCMs running a median (RCP 4.5) CO<sub>2</sub> emission scenario. Panel A =the 2050s; Panel B = the 2080s.

**Fig. 3.** Current wetland proportion (blue), mean annual temperature (red), and mean annual precipitation (black) for selected Alberta ecosystems arranged in a south to north orientation.

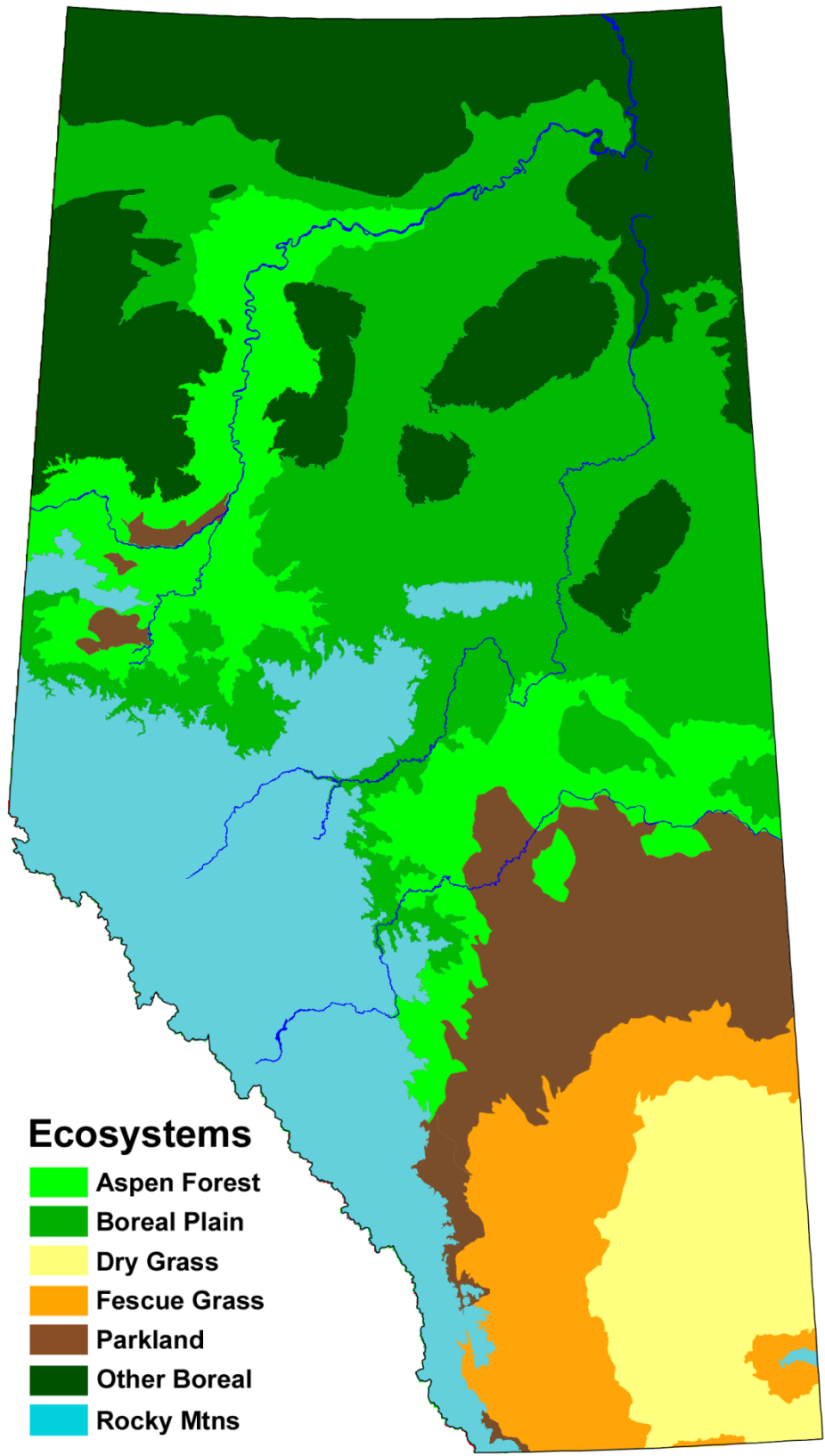


Fig. 1

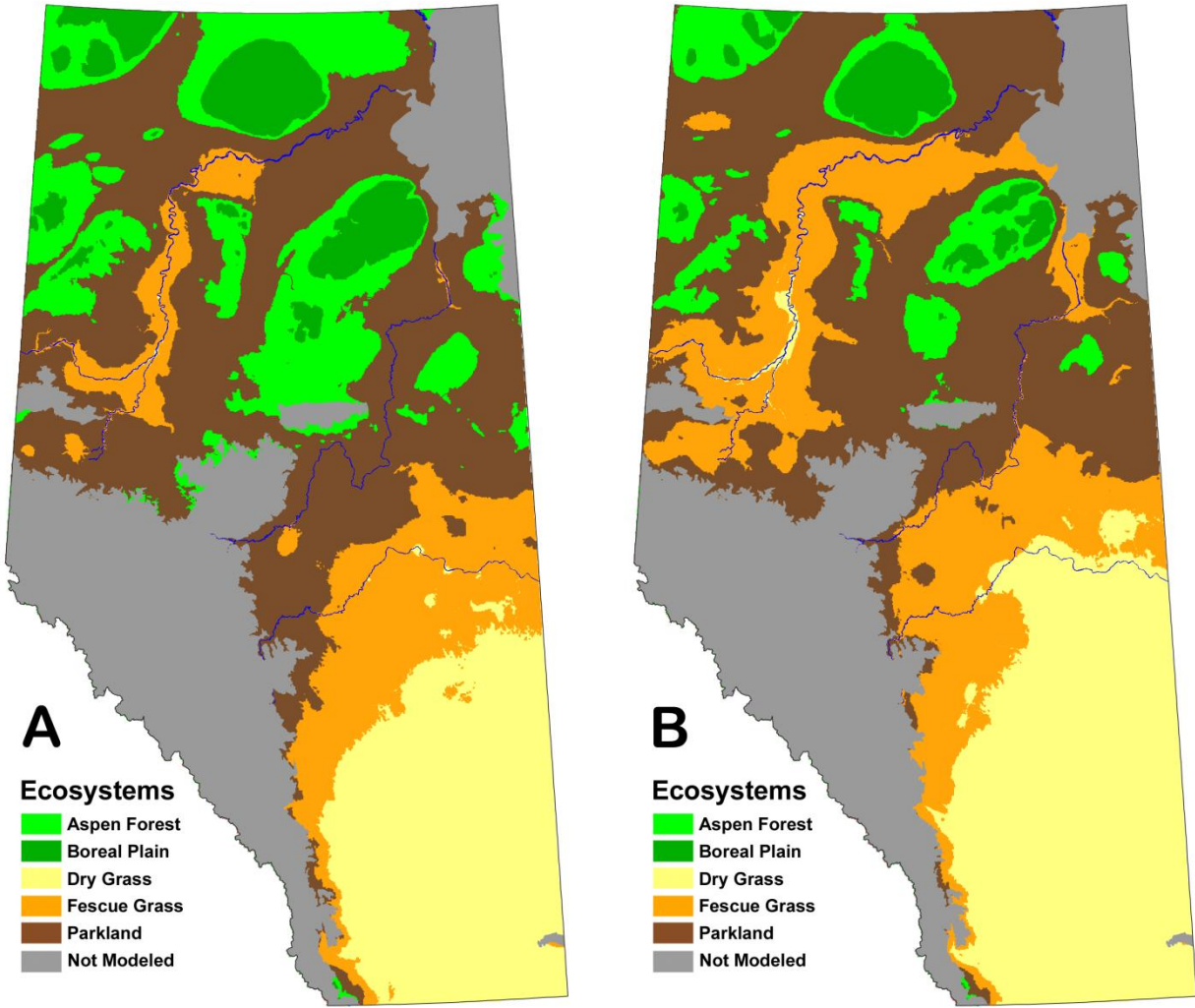


Fig. 2

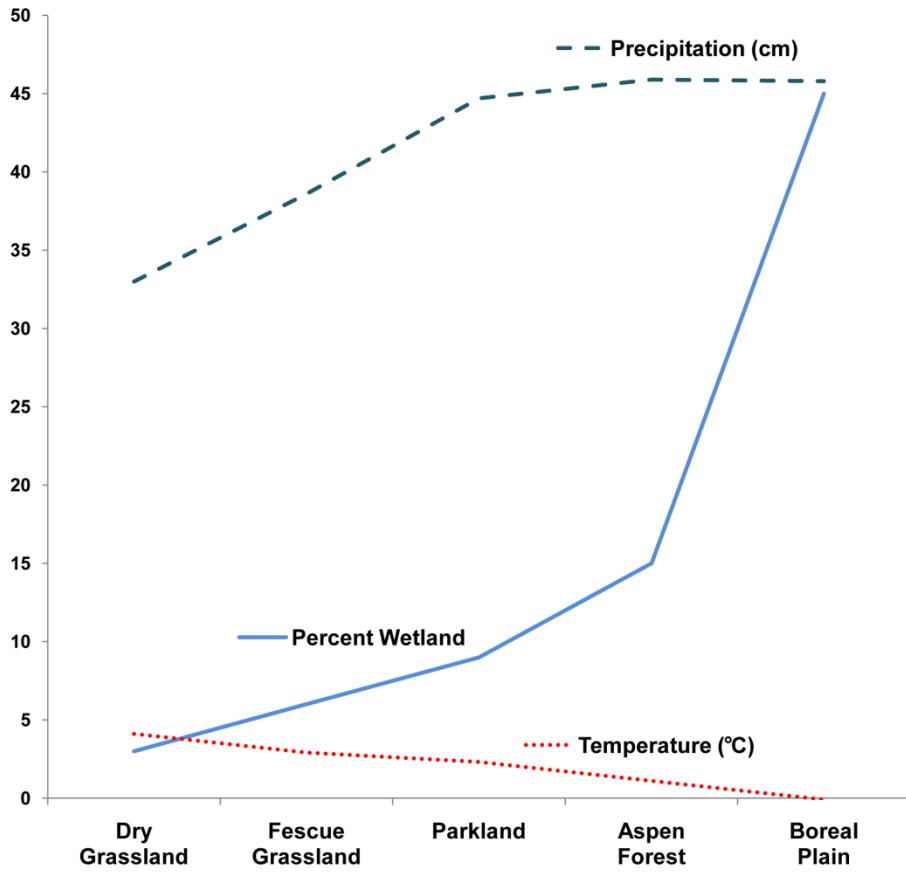


Fig. 3