CLIMATE CHANGE IMPACTS ON THE ISLAND FORESTS OF THE GREAT PLAINS AND THE IMPLICATIONS FOR NATURE CONSERVATION POLICY:

THE OUTLOOK FOR SWEET GRASS HILLS (MONTANA), CYPRESS HILLS (ALBERTA – SASKATCHEWAN), MOOSE MOUNTAIN (SASKATCHEWAN), SPRUCE WOODS (MANITOBA) AND TURTLE MOUNTAIN (MANITOBA – NORTH DAKOTA)

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SUMMARY

This study investigates future climate change impacts on ecosystems, with a focus on trees, in 5 island forest sites in the northern Great Plains ecoregion: Sweet Grass Hills (Montana), Cypress Hills (Alberta-Saskatchewan), Moose Mountain (Saskatchewan), Spruce Woods (Manitoba) and Turtle Mountain (Manitoba-North Dakota). The sites are relatively small forests, isolated from other woodlands by intervening grassland. They have high nature conservation, recreational and cultural value. Their smallness, isolation, restricted number of keystone species and ecotone nature make the island forests very vulnerable to climate change.

Using 3 different global climate models (GCMs) incorporating the latest emissions scenarios we construct climate scenarios for the 2020s, 2050s and 2080s according to standard Intergovernmental Panel on Climate Change (IPCC) guidelines. From these scenarios we derive climate moisture indices (CMIs) based on projected precipitation, temperature and evapotranspiration to model available moisture for vegetation growth. All GCM scenarios indicated declines in moisture levels over time. As compared to the climate normals (the baseline climate) of 1961-1990, the predicted decline in moisture availability at the 5 island forests is approximately 10 cm by the 2020s, 21 cm by the 2050s, and 32 cm by the 2080s. As moisture availability is a critical determinant of forest structure and health at Plains forest sites, the loss of such substantial amounts of moisture is expected to have severe impacts, including the conversion of large areas of forest from trees to scrub or grass cover, the possible extirpation of some tree species, and negative impacts on biodiversity, landscape diversity, and recreational and cultural values. Landscape change may be sudden and dramatic, via vectors such as wildfire, insect attack or severe drought. Traditional minimal-intervention management will not prevent loss of diversity and risks catastrophic and permanent landscape change. Management that aims simply to retain existing vegetation, or to restore historical vegetation distributions and ecosystems, will fail as the climate steadily moves farther away from recent and current norms. A realistic biodiversity strategy must take into account that climate, and therefore flora, fauna, hydrology and soils, will not be static over this century. In a world of climate change, selection of protected areas may need to focus on site heterogeneity and habitat diversity (as these provide some buffer against climate change) rather than on representativeness. As well, preserving some elements of biodiversity will require increasing management counter-intervention across the landscape. Climate change is not currently considered within management plans at any of our study sites.

Given the island forests' vulnerability and the magnitude of probable climate change impacts, we recommend an interim strategy of "managed retreat," incorporating active, anticipatory management, as the best risk management approach. Elements of this "no regrets" strategy include aggressively controlling wildfires and other disturbances, maintaining or creating successional stand diversity, maintaining or increasing landscape diversity, and aiding regeneration of key extant species. It may also be necessary to introduce more drought-tolerant varieties of existing species and, *in extremis*, to introduce entirely new species should a key extant species prove unsustainable as the climate shifts. As the forests are isolated, in-migration of tree species or varieties better adapted to the changing climate would have to be provided by active management. Provenance trials should be undertaken to test new tree varieties and species to provide us with maximum options for ecological salvage. Zoning within each island forest will be a valuable technique allowing a differentiated response to climate change. As the management response to climate change may have to be radical compared with traditional North American nature conservation practice, extensive public consultation will be required. A binational biophysical monitoring program that looks at the entire Plains island forest archipelago collectively, rather than at each site in isolation, is strongly recommended. Also recommended is the expansion of this study north and south to encompass all island forests within the Great Plains ecoregion to determine the bounds of probable climate change, to determine island forests' individual and collective vulnerability, and to foster knowledge transfer of best monitoring, management and consultations practice.

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1. INTRODUCTION

In the midst of the Great Plains, scattered from central Alberta to Texas, are island forests, refugia of trees and tree-dependent species isolated in a sea of grass (for a visual overview see Figures B1 and B2 in appendix B). This study examines the impacts climate change will have on a subset of these unique and valuable forests and makes practical recommendations for the management of these impacts.

Plains island forests are of 4 basic types: highland, sand dune, riparian or scarp. Highland forests owe their existence to increased effective moisture resulting from an orographic precipitation effect and from cooler high-elevation temperatures (with a concomitant reduction in evapotranspiration). Sand dune forests owe their existence to high water tables that result from the rapid infiltration of moisture down through the sand. This infiltration shifts the competitive advantage away from grasses and to deeper-rooted shrubs and trees. Riparian forests are based on a flow of river water whose origin typically lies in a distant, more humid, environment. Sometimes these riparian forests spread out across the nearby Plains; more often they are confined to river valley slopes and to a valley's tributary ravines. Scarp woodlands, like riparian forests, are typically linear in shape and may stretch in long narrow corridors across the Plains landscape. An escarpment may generate a little more precipitation and may be the site of springs. A dissected scarp face will retain pockets of moisture and north or east-facing slopes may be a little cooler. Escarpments also often act as effective firebreaks, which encourages the development of some types of woodland (Wells 1970).

In this study we select 5 island forests as study sites, all of which are relatively close to the U.S. – Canada international boundary. All are highland forests, except for Spruce Woods in southwestern Manitoba, which is a combination riparian and sand dune forest. Turtle Mountain forest straddles the North Dakota – Manitoba international boundary. Cypress Hills forest straddles the Saskatchewan-Alberta interprovincial border. Moose Mountain forest is in southeastern Saskatchewan. Sweet Grass Hills forest is in north-central Montana.

There is good reason to suppose that Plains island forests are at significant risk from climate change. They are marginal or ecotone systems, borderline between grassland and forest ecosystems, and therefore sensitive to relatively small changes in environmental conditions. Saporta et al. (1998) note various scientific papers confirming the common sense expectation that climate change can be expected to affect species populations at their range limits most profoundly. With regard to the boreal forest at its southern boundary with the Plains, IPCC Working Group II (1996) noted that a warming of average annual temperature of as little as 1°C (in the absence of increased moisture) could be enough to shift the forest northward, while the southern limit of the boreal forest converts to grassland or other non-boreal species (Wheaton 1997). Hogg and Hurdle (1995), Lenihan and Neilson (1995) and Price and Apps (1996) all discuss potential zonation shifts and vegetation community changes in the boreal forest.

By focusing on trees as a proxy for forest ecosystems we are adopting a "keystone" species approach. The assumption is that within a given ecosystem, whilst all species and individual organisms are ultimately interrelated by a network of structures and processes such as food webs or energy flows, a few particular species may play a critical role in supporting the existence of many others (Paine 1980) and one is therefore justified in paying particular attention to these species' habitat needs. Anderson et al. (1998), for example, note the importance of keystone species in climate change modelling. Trees are clearly keystone species in island forests and, as we are able to model their prospects, it is reasonable to do so. This does not, however, preclude that other species may have key roles affected by climate change in these island forests that we do not here address in detail.

As a management response to climate change Webb (1992) and Dudley et al. (1996) agree that we need protected natural areas to allow and assist the migration of species to areas of more suitable habitat. Yet even in large contiguous forests natural migration rates may not be nearly fast enough to cope with climate change. Scott and Suffling (2000), for example, suggest white spruce can only move at the rate of 3 to 4 km every 40 years. A more plausible scenario is that pre-existing minor arboreal elements of an ecosystem will expand as the environment changes to their advantage. In any case, ecological rights-of-ways for trees across grassland from island forest to island forest are not feasible and a key characteristic of island forests is that, by definition, the trees within them have nowhere else to migrate to. Further, as they are relatively small ecosystems, island forests may not be as ecologically diverse or as resilient to climate change as larger systems. They may exhibit reduced genetic diversity and greater vulnerability to catastrophic disturbance, such as wildfire, insect attack or severe drought. Should a virulent pest or invader reach an isolated forest, it may be some time, if ever, before an appropriate natural control also reaches the forest. As we discuss in chapter 11, periodic major disturbances are the norm, not the exception, in the greater Plains environment in which the island forests are embedded. This increases the global vulnerability of these isolated habitats.

While vulnerable, island forests are also valuable and contested landscapes. They typically contain important species and ecosystem outliers at the very edge of their natural range, making them of conservation importance. Highland forests often supply valuable water to the surrounding plains. As trees are valued on the Plains, all of the forests are important for tourism and recreation. Island forests sometimes contain small lakes and ponds that are valuable for a great variety of wildlife, especially waterfowl. They are often of cultural and spiritual importance to local Native North American peoples. Despite their value and vulnerability, Plains island forests (in contrast, for example, to the commercial boreal forest of North America) have been little studied with regard to future climate change (although paleoclimatic records are sometimes available).

Typically island forests are not managed for timber harvest, but primarily for recreation and nature conservation purposes and for sustainable multi-resource use. This makes management for climate change impacts more difficult than in managed-for-harvest forests (Cohen and Miller 2001), since forest managers with a wood-production focus can simply continue, with new species and varieties, traditional timber-maximisation strategies. A further complication in the island forests is that normally there are pre-existing management or use stresses on island forests (such as grazing, mining, hunting, or oil and gas pressures). How to manage a forest undergoing significant climate change for nature conservation purposes is particularly unclear.

We can summarise the key characteristics of the island forests as follows:

- they are landscapes on the edge of constant change, with a (pre)history of frequent forest expansion and contraction;
- there is little possibility of natural migration of tree species as is sometimes postulated for large contiguous forests as climate change progresses;
- island forests may lack the resiliency of larger forest areas;
- they are likely to be affected early, and possibly dramatically, by climate change;
- they harbour valued plant and wildlife diversity;
- they contain outlier species populations;
- they have been largely preserved from fires for 100 or more years;
- in detail they are individualistic forests with individualistic histories;
- highland forests are waterheads to surrounding arid regions;
- they are valued landscapes surrounded and impacted by intense and competing stakeholder interests;
- they are very valuable for scientific research owing to the fact they are ecotone landscapes, sensitive to climate change, are in a more natural state than surrounding agricultural lands and, if parks, may be specifically mandated to encourage research;
- prime management concerns are nature conservation and recreation, not commercial timber production;
- they are small enough and accessible enough to be actively managed at relatively low cost, should policy so dictate; and,
- government, as a major or majority landholder in many island forests, has a responsibility to act proactively, in consultation with stakeholders and the broad public, as responsible trustee for sustainable forests.

Critically, although climate change is correctly viewed as a long-term issue, managers need advice on how to manage the island forests today. Climate change is already measurably underway, and today's sapling may have to cope, as a mature tree, with a significantly different climate 50 or 100 years from now. We also now need workable management models to address such issues as the current and future risk of catastrophic landscape change. For example, should a major wildfire sweep one of the island forests, do we know what the prospects for natural regeneration are? Would we know what, if anything, to replant?

Solomon (1994) and Halpin (1997) both note that managers of nature reserves and parks need to consider the possible impacts of rapid climate change within their management strategies. But it is not enough to consider each reserve or park in isolation. As important is the need to evaluate the effectiveness of a system or set of reserves in a world of climate change. The climate future of one island forest will affect our management objectives in another, and individual species or ecosystems under threat in one forest may find a useful conservation role in another. We need to view these forests as members of a great and scattered forest archipelago. This study is the first effort, to our knowledge, to consider the island forests collectively, rather than in isolation from each other, and to view the Plains island forests as related entities, rather than as isolated systems to be managed individually.

Peters and Darling (1985) argue that in serious cases of population decline, conservation management has to become increasingly intrusive if it is to have any hope of success. Our climate scenarios suggest that the island forests will suffer serious challenges to ecosystem integrity that may demand much more active management than is currently practised. "Salvage ecology" or ecological triage could involve relatively radical, and to many North Americans unpalatable, management techniques. The experimental introduction of exotics, or the undertaking of breeding programs to create varieties more adapted to a new climate, may be advisable. Traditional hands-off wilderness or wildlands conservation may be an inadequate response.

We address the following questions in this study:

- What has been the vegetation prehistory and history of selected island forests on the northern Plains?
- What is the range of probable future climates for these island forests?
- What impacts would these climates likely have on the key tree species of these existing forests?
- What do these impacts imply for nature conservation management, in general, and at these sites in particular?
- Does current forest management at these sites take into account climate change?
- What are the immediate and long-term management options available to us in light of probable climate change impacts?
- How should we choose amongst these options and which options seem most plausible?

The study structure is as follows. First we present up-to-date climate change scenarios for the northern Plains area centred on the 49th parallel international boundary. The scenarios are constructed according to International Panel on Climate Change (IPCC) recommendations. From these scenarios we generate climate moisture indices (CMIs) both for the entire northern Plains study area (Figure 1) and for the specific island forest sites. We discuss how to understand and interpret these CMI scenarios in terms of vegetation impacts. We then present the vegetation and management history and prehistory of each of our study sites, including current vegetation management policy and issues. In general, we find climate change is not incorporated into current vegetation management strategies. We provide forecasts of likely forest futures for each of the study sites individually, and then for the island forests collectively, in the light of climate change. We then present possible management model options to adapt to climate change. Several of these options imply a radical, and certainly contentious, shift in North American conservation philosophy. A shift away from the traditional conservation consensus – centred around wilderness preservation and low-intervention management – and towards a much more interventionist conservation paradigm, may be forced upon us by climate change. Finally, we present a series of practical recommendations to adapt to climate change both within each study site and within the entire island forest archipelago.

2. STUDY SITE SELECTION

We selected study sites on the basis of the following:

- the existence of unresolved vegetation management issues;
- the presence of locally unique and possibly threatened ecosystems;
- the presence of species of particular interest;
- an assessment of the level of government and public interest;
- site managers' interest in climate change input into their management decisions;
- our desire for an international, inter-state and inter-provincial sweep of climate and vegetation issues;
- the existence of ecological linkages and comparability between sites;
- site modelability; and,
- manageability of study site numbers.

Although more laborious in many ways (even weather instrumentation varies between Canada and the United States), we have deliberately included study sites from 5 jurisdictions in 2 countries. As climate change affects the entire Great Plains ecoregion, it makes sense to consider climate, ecology and policy on both sides of the international boundary. Because of climate, vegetation and physiography similarities Turtle Mountain and Moose Mountain make a logical pair for comparison of site ecology and management, as do the Cypress Hills and the Sweet Grass Hills. Spruce Woods is included as it is an ecologically unique site with numerous management issues and is an interesting example of a riparian and sand dunes island forest. The Sweet Grass Hills contain several disjunct blocks of forest separated by grassland and for some purposes we break that site into 2 forested subsites, West Butte and East Butte. The same is true of the Cypress Hills, which we sometimes analyse in terms of 2 subsites, West Block and Centre Block.

Table 1 summarises, for reference purposes, the native tree species present within our 5 study sites. Detail about each individual site's forest and management history, and about current vegetation issues, is found in the relevant site chapter.

Table 1: Summary of native tree species within the study sites: SGH – Sweet Grass Hills; CH – Cypress Hills; MM – Moose Mountain; TM – Turtle Mountain; SW – Spruce Woods. An "x" signifies the presence of the species; "xx" signifies the species is dominant or co-dominant; "?" indicates the species' presence is hypothetical.

Common name	Scientific name	SGH	CH	MM	TM	SW
Whitebark pine	*Pinus albicaluis*	X				
Lodgepole pine	*Pinus contorta*	XX	XX			
Limber pine	*Pinus flexilis*	X				
Tamarack	*Larix laricina*					X
White spruce	*Picea glauca*		X			X
White x Engelmann spruce	*Picea glauca* x *engelmannii*	X				
Black spruce	*Picea mariana*					X
Colorado spruce	*Picea pungens*	X				
Subalpine fir	*Abies lasiocarpa*	X				
Douglas-fir	*Pseudotsuga menziesii*	XX				
Rocky Mountain juniper	*Juniperus scopulorum*	X				
Aspen poplar	*Populus tremuloides*	XX	XX	XX	XX	XX
Balsam poplar	*Populus balsamifera*		X	X	X	X
Black cottonwood	*Populus trichocarpa*	X	?			
Eastern cottonwood	*Populus deltoides*	X	X	?	?	X
Narrowleaf cottonwood	*Populus angustifolia*		X			
Willows	*Salix* spp.	X	X	X	X	X
White birch	*Betula papyrifera*	X	X	X	X	X
Bur oak	*Quercus macrocarpa*			?	X	X
American elm	*Ulmus americana*				X	X
Hackberry	*Celtis occidentalis*				?	?
Greene mountain-ash	*Sorbus scopulina*		X			
Douglas maple	*Acer glabrum*	X				
Manitoba maple (boxelder)	*Acer negundo*	X	X	X	X	X
Basswood	*Tilia americana*					X
Green ash	*Fraxinus pennsylvanica*			X	X	X
Black ash	*Fraxinus nigra*					X

3. STUDY REGION BOUNDARIES AND CURRENT CLIMATE

The island forests considered in this study are located in the southern Canadian and northern U.S. Great Plains within the region defined by 47.5°N to 51.0°N and 115°W to 95°W (see Figure 1). This is a region of cold winters, hot summers, an annual temperature range exceeding 80°C, and large climate variability (Lemmen et al. 1997). Winters are generally dry, while summer is the wettest season. 20% to 30% of annual precipitation falls as snow, while 70% to 80% falls as rain. June is generally the wettest month. The western part of this region is in the rain shadow of the Rocky Mountains, which form an effective barrier to the maritime influence of the Pacific Ocean. The eastern part has greater exposure to southerly flows of warm, moist air from the central U.S. and the Gulf of Mexico. Precipitation amounts, therefore, tend to increase from west to east. Despite summer being the wettest season, strong sunshine, low humidity and drying winds lead to large evaporative water losses in this season. Lowest mean temperatures generally occur in January, with the warmest winter temperatures located in the south-west of the region and coldest temperatures in the north-east. In July, generally the hottest month, the pattern is similar, although the difference between the cooler north-east and the warmer south-west is not as marked as in winter.

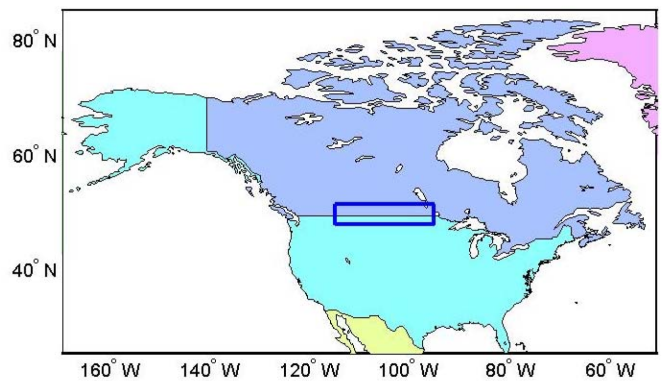

Figure 1: Boxed area indicates the region within which the island forests considered in this study are located.

Joyce et al. (2001) note that annual precipitation has decreased by 10% in eastern Montana and western and central North Dakota over the past 100 years, while Bootsma (1994) notes there is no historical precipitation trend apparent for the Canadian prairie region. Statistically significant warming of 0.9°C from the late 1800s to the 1980s is evident over this region (Lemmen et al. 1998).

4. SELECTION AND CONSTRUCTION OF CLIMATE SCENARIOS

Climate change scenarios for the island forests were constructed in accordance with the *Guidelines on the Use of Scenario Data for Climate Impact and Adaptation Assessment* recommended by the Intergovernmental Panel on Climate Change Task Group on Scenarios for Climate Impact Assessment (IPCC-TGCIA 1999). These guidelines outline the criteria (detailed in appendix G) to which climate change scenarios should conform if a meaningful assessment of the impacts of climate change is to be made. The IPCC-TGCIA recommendations indicate that global climate models (GCMs) – three-dimensional computer models which mathematically model the physical processes of the atmosphere, oceans, cryosphere and land surfaces, and the relationships between these systems – are currently the only available means of constructing scenarios of climate change which are physically plausible and which can encompass the range of possible future climates.

For construction of the climate change scenarios used in this study we selected the most recent GCMs (which are generally more sophisticated than earlier versions) and recent climate change experiments undertaken with *Special Report on Emissions Scenarios* (SRES) emissions scenarios. SRES emissions scenarios were commissioned by the IPCC for its Third Assessment Report (IPCC 2001) and are described in detail in Nakicenovic et al. (2000). They provide detailed scenarios of atmospheric composition (i.e. concentrations of greenhouse gases and aerosols) which are used within GCMs to determine the effect on the radiation balance of the Earth-atmosphere-ocean system.

SRES emissions scenarios have only recently become available and it takes several months to complete a single climate change experiment within a given GCM. For these reasons a complete suite of SRES emission scenarios experimental results from all IPCC-recommended GCMs was not available at the time of this study. However, results from 4 emissions scenarios and 3 different GCMs were available for scenario construction as shown in Table 2. Since 3 experiments for each of the A2 and B2 emissions scenarios were carried out with the CGCM2 model, there were 12 scenarios available, all of which were investigated by us for this study.

Table 2: Availability of SRES climate change experiments for use in scenario construction

GCM	SRES emissions scenario			
	A1	A2	B1	B2
HadCM3[1]		✓		✓
CGCM2[2]		✓✓✓		✓✓✓
CSIROMk2b[3]	✓	✓	✓	✓

[1]U.K. Hadley Centre for Climate Prediction and Research, Climate Model 3 (Gordon et al. 2000; Pope et al. 2000)
[2]Canadian Centre for Climate Modelling and Analysis, Climate Model 2 (Flato and Boer 2001)
[3]Australian Commonwealth Scientific and Industrial Research Organisation, Climate Model Mark 2b (Hirst et al. 2000)

A1, A2, B1 and B2 are emissions scenarios identified in the *Special Report on Emissions Scenarios* (Nakicenovic et al. 2000) representing different world futures with respect to population and economic growth, energy use, technology development, etc. Each of these possible world futures implies different trends in future emissions of greenhouse gases and aerosols. By using this range of emissions scenarios within GCMs we can construct a suite of climate change scenarios reflecting the range of probable future climate. The A1 emissions scenario tends to indicate the largest increases in global mean temperature. A2, followed by B2, represent more mid-range scenarios, and B1 indicates the smallest increase in global mean temperature by 2100. The SRES range in global mean temperature change is 1.4°C to 5.8°C by 2100, relative to 1990 (IPCC 2001).

We constructed climate change scenarios by calculating the change in climate conditions between some future time period and a baseline period using GCM output. The current baseline period recommended by the IPCC-TGCIA is 1961-1990. In theory, any future time period can be targeted for scenario construction. However, by encouraging the standardisation of scenario time periods the IPCC-TGCIA hopes to encourage comparability amongst different climate change impacts studies and in practice recommends constructing scenarios centred on the 2020s, 2050s and 2080s. As these are appropriate and useful time periods for our purposes (the consideration of tree and ecosystem futures), we conformed to this recommendation.

Standard practice is to apply the climate change scenarios to observed climate data. Therefore we constructed 1961-1990 baseline climate normals by calculating the 30-year average climate using data from climate stations within our study area. We then applied climate change scenarios at the seasonal time scale to these observed 1961-1990 climate normals data. The resulting climate scenario values for maximum temperature, minimum

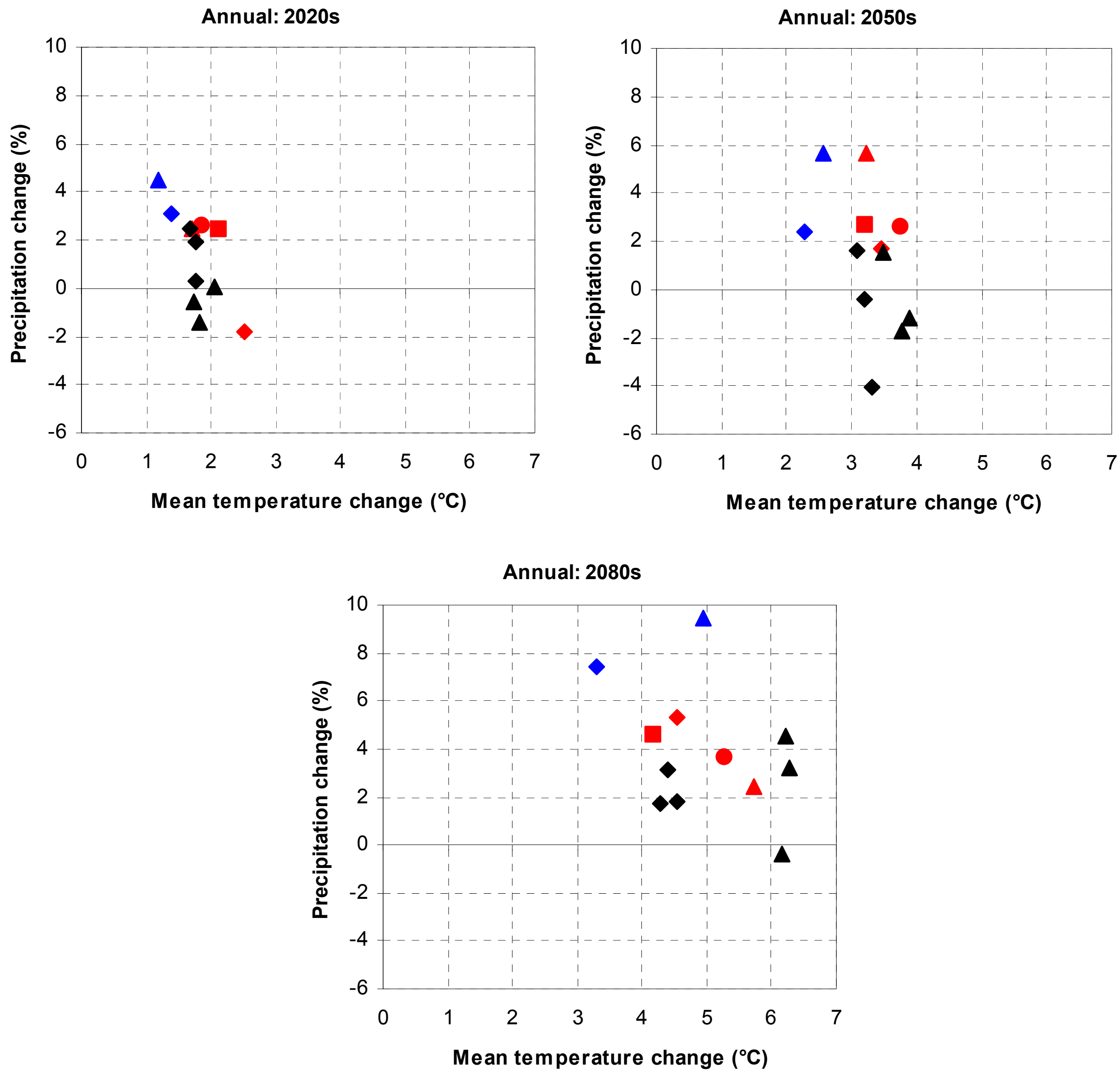

Figure 2: Scatter plots of annual mean temperature change (°C) versus precipitation change (%) over the island forests study region for the 2020s, 2050s and 2080s. All changes are calculated with respect to 1961-1990. Different symbol colours indicate different GCMs; different symbol shapes indicate different emissions scenarios: CGCM2 – black; HadCM3 – blue; CSIROMk2b – red. A1 – circles; A2 – triangles; B1 – squares; B2 – diamonds. Each symbol represents a single experiment.

temperature and precipitation were then used to calculate Climate Moisture Index (CMI) values for the island forests for the 2020s, 2050s and 2080s.

All 36 climate scenarios (the 12 SRES experiments × 3 time periods) were used to calculate CMI values at the island forest sites, but for reporting purposes we have focused on 3 of these scenarios which we judge encompass the range of probable future CMI values at each of the study sites. We made our selection on the basis of annual temperature and precipitation changes averaged over the complete island forest study region shown in Figure 1. We used scatter plots of mean temperature change versus precipitation change (see Figure 2) to identify the climate change scenario which indicated the coolest and wettest conditions in the future relative to the 1961-1990 period. Similarly we identified a scenario representing the warmest and driest future conditions and a scenario representing mid-range conditions.

We focused on the 2050s as the basis for our selection of climate change marker scenarios, as we judged this to be perhaps the most critical timeframe for today's vegetation management decisions. We selected HadCM3 B21[1] as the cool-wet marker scenario, CGCM2 A21 as the warm-dry marker scenario, and CSIROMk2b B11 as a mid-range marker scenario. Seasonal and annual changes in mean temperature and precipitation (calculated with respect to the model-simulated 1961-1990 period) for each of these 3 scenarios are shown in Table 3. This table indicates that most of the scenarios exhibit increases in both mean temperature and precipitation on an annual basis for all future time periods, but the seasonal breakdown of these changes shows that there may be substantial decreases in precipitation in the summer growing season. The spatial patterns of annual maximum temperature, minimum temperature and precipitation change are shown in appendix D.

Table 3: Seasonal and annual changes in mean temperature (ΔT in °C) and precipitation (ΔP in %) over the island forest study region for the HadCM3 B21 (cool-wet), CSIROMk2b B11 (mid-range) and CGCM2 A21 (warm-dry) climate change scenarios

	Winter (DJF)		Spring (MAM)		Summer (JJA)		Autumn (SON)		Annual	
	ΔT	ΔP	ΔT	ΔP	ΔT	ΔP	ΔT	ΔP	ΔT	ΔP
2020s										
HadCM3 B21	1.0	11.6	1.2	5.9	1.7	-4.1	1.6	9.0	1.4	3.1
CSIROMk2b B11	2.6	6.6	2.9	12.9	1.5	-6.5	1.5	0.9	2.1	2.5
CGCM2 A21	1.3	6.8	2.9	8.9	1.7	-9.8	1.0	-2.6	1.7	-0.6
2050s										
HadCM3 B21	0.9	19.3	1.5	13.3	3.5	-9.6	3.2	0.9	2.3	2.4
CSIROMk2b B11	3.7	13.2	3.2	15.5	2.8	-11.2	3.1	0.4	3.2	2.7
CGCM2 A21	3.9	5.2	6.5	10.8	3.2	-9.4	1.9	-6.4	3.9	-1.2
2080s										
HadCM3 B21	2.4	15.0	2.2	14.3	4.5	-1.6	4.0	11.7	3.3	7.5
CSIROMk2b B11	5.1	27.9	3.5	21.1	4.0	-18.0	4.0	0.3	4.2	4.7
CGCM2 A21	7.2	2.4	9.5	33.2	4.9	-16.4	3.4	12.7	6.2	4.5

It is noteworthy that, as in this study, the recent U.S. government submission to the United Nations, *U.S. Climate Action Report – 2002* (EPA 2002), also employs GCM results from the U.K. Hadley Centre and the Canadian Centre for Climate Modelling and Analysis to represent 2 possible climate futures. The emissions scenarios and experiments we use here are more recent and we also include a mid-range (CSIRO) scenario.

[1] The standard naming convention is used here. "HadCM3 B21" indicates that the GCM is Hadley Climate Model number 3 and the emissions scenario is SRES B2. The final '1' indicates that this is the first experiment combining this climate model with this emissions scenario.

5. CLIMATE MOISTURE INDICES: THEIR NATURE AND INTERPRETATION

Moisture is key in determining the regional distribution of vegetation (Looman 1979; Larsen and MacDonald 1995; Hogg 1997) in southern Alberta, Saskatchewan and Manitoba and adjacent regions of Montana and North Dakota. Hogg (1994) showed, by means of a climate moisture index (CMI), that the distribution of vegetation zones in this region was much more closely related to climatic moisture regimes than to thermal indicators such as temperature or growing degree days. Hogg's CMI was calculated as mean annual precipitation minus potential evapotranspiration, measured in centimetres of water per year. Evapotranspiration includes both evaporation (e.g. water vapour loss from wet vegetation or soil) and transpiration (water vapour loss from the leaves of living plants). Potential evapotranspiration (PET) is defined as the expected rate of water loss from a well-vegetated landscape when soils are moist. In his analysis Hogg (1994) estimated PET using the relatively simple Jensen-Haise method (Bonan 1989; Jensen et al. 1990), which can be calculated from long-term averages of monthly temperature, monthly solar radiation and altitude. Similar CMI results were later obtained using 2 other methods of estimating PET (Hogg 1997). We use the Jensen-Haise method in this study. Other methods of estimating PET can be expected to show similar climate trends to those identified here.

CMIs are determined solely from climate measurements, although the slight influence of altitude on the physical process of evapotranspiration (through differences in air pressure) is also taken into account. Unlike the Palmer Drought Severity Index that has been commonly used for agricultural applications (Jones 1984), calculation of CMIs does not require information on the available water-holding capacity of soils. This is advantageous, as available water for trees is affected by both soil characteristics (e.g. texture) and rooting depth, and in forests rooting depth is highly variable and depends on both tree species and site conditions (Strong and La Roi 1983).

CMIs calculated and mapped here (Figures 4 through 13; illustrated and explained later in this chapter) assume level topography, i.e. that all locations on the landscape receive an equal amount of solar radiation. In reality, however, aspect and slope affect radiation impact. North-facing slopes are notably moister than south-facing slopes because the latter are exposed to more solar radiation. White spruce distribution in the parkland region of Saskatchewan and Alberta is therefore often restricted to the north-facing slopes of major river valleys (Hogg 1994, Hogg and Schwarz 1997), while in the parkland region of eastern Saskatchewan and western Manitoba, in the eastern Qu'Appelle Valley, more drought-tolerant bur oak dominates south-facing slopes, while aspen dominates the north-facing slopes (Henderson 2001).

In principle, the effect of local topographic variation on average moisture conditions could be estimated by calculating PET using monthly solar radiation data for various slopes and aspects. For example, the analyses of McKay and Morris (1985) show that during the growing season (May to September), the annual amount of solar radiation on a 30º north-facing slope at Medicine Hat, Alberta, or at Prince Albert, Saskatchewan, is about 28% less than that measured on level terrain. When the reduced value of solar radiation is applied using the Jensen-Haise method, the estimated annual PET is reduced by about the same percentage (28%) at each of these locations. However, the estimated annual PET for level terrain is much greater at Medicine Hat (73 cm per year) than at Prince Albert (44 cm per year), because the latter station has cooler temperatures and slightly less solar radiation during the growing season. This means that the absolute moisture benefits of a 30º north-facing slope are greater at Medicine Hat (28% of 73 cm, or about a 20 cm reduction in PET per year) than at Prince Albert (28% of 44 cm, or about a 12 cm reduction in PET).

Slope also affects precipitation input. More steeply sloping terrain may receive additional moisture from runoff and groundwater discharge, which would increase a local CMI value. In this context it is worth noting that CMIs are not appropriate for estimating moisture conditions at riparian or spring discharge sites – i.e. wherever moisture levels are determined by inflow from headwater sources rather than from local precipitation.

Using 1951-1980 climate normals for the relevant 254 climate stations in western Canada, Hogg (1994) found a strong fit between the zero CMI isoline (i.e. where precipitation and PET are equal) and the boundary of the western Canadian boreal forest ecoregion with the parkland ecoregion. North of this isoline, annual CMI values are positive, and the amount of rain and snow received (in combination with cooler temperatures) is sufficient to produce water runoff from the landscape. In these areas surplus moisture permits the establishment of continuous forests that include aspen and several species of boreal conifers. South of the zero isoline, where CMI values are negative, natural conifers are rare or absent and deciduous trees (predominantly aspen) are restricted to stunted patches in favourable sites. There is also a strong fit between the -15 CMI isoline and the boundary between the parkland ecoregion and the grassland ecoregion, where there are no naturally occurring trees, except at riparian sites (see Hogg 1994 for maps). Table 4 provides a generalised summary of the (non-riparian) natural vegetation associated with various ranges of CMI in our study area.

Table 4: CMI values and their associated natural vegetation

Range of CMI	General Vegetation
greater than +15	Boreal or Cordilleran (coniferous species dominant)
zero to +15	Boreal or Cordilleran (aspen-dominated mixed-woods with conifers)
-15 to zero	Parkland (mixed aspen and grassland; conifers rare)
-30 to -15	Grassland (stunted aspen in favourable sites; no conifers)
less than -30	Grassland (no trees)

As our study region's climate is trending towards milder winters, it is important to note that in regions associated with milder winters, such as the central or southern Great Plains, or interior British Columbia, some tree species form savannah woodlands at considerably lower CMI values than illustrated above. For example, open ponderosa pine woodlands can be found in the British Columbia interior under CMI conditions as low as -40.

While CMIs provide an excellent fit to the boundary lines between several major vegetation ensembles, they do not themselves specify the critical constraint that has, in the past, prevented conifers from growing in areas of water deficit, or prevented aspen from prospering in areas drier than -15. Possible causative explanations are many and may be indirect. It is conceivable, for example, that the climate moisture index is a proxy measure for fire frequency or intensity, or is related to germination success, or is related to insect or other biological control. For example, it may be that under xeric conditions conifer seedlings cannot establish themselves, or that pathogens are more damaging in dry conditions, or that fire is more frequent or more damaging.

Knowing the exact causation is critical to defining the limits of the possible in terms of conservation policy and would aid us in efforts to maintain a vegetation ensemble outside of its expected future geographical distribution as indicated by a future scenario CMI. If fire frequency is the issue, we can suppress wildfires; if pest or disease virulence is the key constraint, we can manage aggressively for pathogens; if poor regeneration under low moisture conditions is the key constraint, we can assist by planting seedlings.

Provisionally it seems very likely that inhibited regeneration is the most important constraint on trees in the study area. Hogg and Schwartz (1997) concluded that low germination and seedling survival rates as a result of inadequate moisture were the controls on white spruce success in the aspen parkland and in the grassland south of the boreal forest. Daniel et al. (1979) identifies the first few weeks of seedling development as the most critical in the lifecycle of conifers and Day (1963) and Noble and Alexander (1977) note that moisture deficiency reduces the rate of survival of conifer seedlings even in naturally forested regions. Solomon (1994) states that most seedlings can survive only within very narrow limits of soil moisture and that seedlings are most vulnerable to shifts in precipitation amounts. There is little doubt that drought tolerance of juvenile and adult trees is much greater than that of seedlings. Understanding the drought tolerances of different species has been identified as one important adaptation research question by the *Canada Country Study* (Herrington et al. 1997). More research across species and sites is required, but regeneration under droughty conditions seems the critical issue.

As explained in chapter 4, the baseline climate period used in this study is 1961-1990. Hogg's (1994) initial CMI paper used a 1951-1980 baseline. A comparison of the 2 normals periods (not illustrated here) shows a shift in the CMI isolines – in general there is a lowering of CMI values over our study region when moving from 1951-1980 normals to 1961-1990 normals. This is a result of the warmer and drier climate experienced in the 1980s as compared with the 1950s and is a finding consistent with climate change. It is also an indication that climate change is not simply something that will occur in the future; it is ongoing today.

Without exception the 12 SRES scenarios we examined for the 2020s, 2050s and 2080s project decreases in CMI values with respect to the 1961-1990 normals baseline. Were we to have used a 1951-1980 normals baseline, the projected CMI decreases would be greater still. Table 5 shows the mean decline in CMI values (compared with the 1961-1990 normals) for our study sites averaged across all 12 scenarios.

Table 5: Mean decline in centimetres of CMI values (mean of all 12 scenarios)

	2020s	**2050s**	**2080s**
Sweet Grass Hills (West Butte)	-10.6	-20.9	-30.3
Sweet Grass Hills (East Butte)	-10.6	-21.0	-30.6
Cypress Hills (West Block)	-9.8	-20.1	-30.1
Cypress Hills (Centre Block)	-8.6	-18.5	-27.8
Moose Mountain	-10.6	-22.3	-34.4
Spruce Woods	-9.0	-20.2	-30.4
Turtle Mountain	-11.0	-22.9	-35.1

If we average the Sweet Grass Hills West Butte and East Butte subsites we can calculate single values for the Sweet Grass Hills site as a whole (declines of 10.6, 21.0 and 30.5 for the 2020s, 2050s and 2080s, respectively). Similarly, we can calculate averages for the Cypress Hills West Block and Centre Block subsites (9.2, 19.3 and 29.0). By averaging these values together with the relevant values for Moose Mountain, Spruce Woods and Turtle Mountain, we can establish means across the 5 study sites. The mean CMI declines of all 12 SRES scenarios, averaged across all 5 study sites, are 10.1 for the 2020s, 21.1 for the 2050s, and 31.9 for the 2080s.

It would be inappropriate to simply consider the scenario averages when planning and implementing a policy response to climate change. For successful adaptation it is far wiser to consider the full range of probable climate outcomes. One useful way to visualise this range is via a box-and-whisker plot. Figure 3 summarises the changes in CMI (averaged over all our 5 study sites) for the 2020s, 2050s and 2080s with respect to the 1961-1990 climate normals. Based on our 12 scenarios, for each time period the plot indicates the complete range of scenario CMIs (the thin vertical line), the range within which 50% of the scenario CMIs fall (the box), the median scenario CMI value (the dash mark within each box), and the CMI value of each of our marker scenarios (coloured icons).

From Figure 3 it is again evident that CMI levels are projected to fall over time. As stated earlier, we focused on the 2050s for the selection of marker scenarios. Nonetheless, our chosen cool-wet scenario generates the highest or near-highest possible CMI value for all time periods. Our chosen mid-range marker scenario is indeed mid-range for the 2050s and 2080s, and slightly drier than the median value for the 2020s. However, most noteworthy is that for the 2020s and 2080s an even drier CMI than indicated by our warm-dry marker scenario is possible. For example, the projection of our warm-dry marker scenario for the 2080s (-35.9) is exceeded by the projection of -43.9 generated by a non-marker scenario (see Figure 3). Equally, the 2020s could possibly be even drier than our warm-dry marker scenario suggests (-12.2), and could be as low as -18.4.

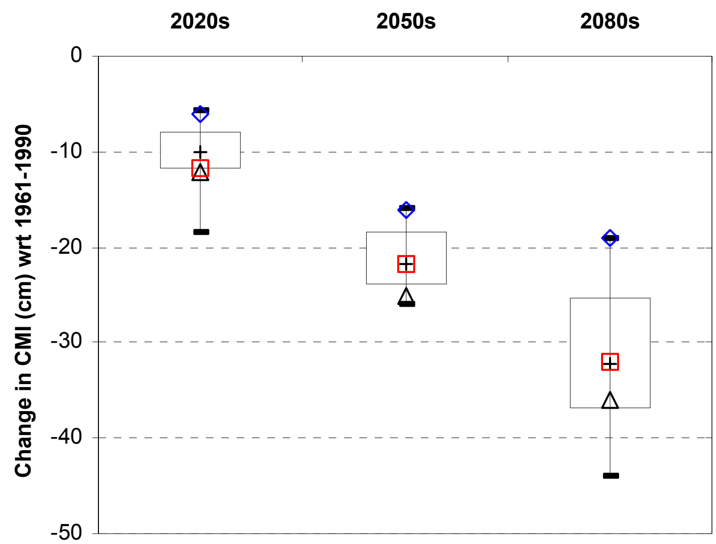

Figure 3: Box-whisker plots indicating the change in CMI (cm) with respect to the 1961-1990 climate normals for the 2020s, 2050s and 2080s. Each plot indicates the extreme values (the "whiskers" top and bottom of the vertical line), the range within which 50% of the data fall (the box), and the median value (–). The three marker scenarios are also indicated: HadCM3 B21 – ◊; CSIROMk2b B11 – □; CGCM2 A21 – Δ.

Table 6 shows the numerical CMI output of our 3 climate change marker scenarios. On the left is the cool-wet scenario for our study region. In the centre is our mid-range scenario, i.e. a scenario which falls close to the median dash mark at all time periods. At right is the warm-dry scenario for our study region.

Table 6: Projected changes in study site CMIs relative to the 1961-1990 normals

	HadCM3 B21			**CSIROMk2b B11**			**CGCM2 A21**		
	2020s	2050s	2080s	2020s	2050s	2080s	2020s	2050s	2080s
Sweet Grass Hills (West Butte)	-9.5	-22.9	-24.2	-9.9	-20.5	-27.0	-10.2	-21.1	-31.9
Sweet Grass Hills (East Butte)	-9.9	-23.3	-25.5	-9.7	-20.2	-26.7	-10.2	-21.5	-32.4
Cypress Hills (West Block)	-8.3	-20.4	-22.0	-8.8	-19.2	-27.3	-10.2	-22.6	-33.1
Cypress Hills (Centre Block)	-6.7	-18.7	-18.7	-7.3	-18.4	-28.7	-9.1	-21.4	-29.2
Moose Mountain	-4.5	-13.1	-17.6	-13.6	-23.3	-35.0	-13.7	-27.6	-39.9
Turtle Mountain	-4.9	-14.0	-18.1	-14.1	-24.0	-36.2	-14.3	-28.2	-40.5
Spruce Woods	-3.3	-10.6	-14.3	-13.1	-22.3	-33.7	-13.3	-26.7	-35.9
Mean (N=5 forests)	-6.0	-16.1	-19.0	-11.7	-21.8	-32.0	-12.2	-25.2	-35.9

By means of a climate station interpolation procedure described in appendix F we calculated estimated values of CMI for each of the study sites across the actual range of elevation present at each site. The results are displayed visually by the study site elevation bar charts which accompany the climate normals map and each scenario map (Figures 4 through 13). The bar charts run from maximum site elevation (or slightly above) to a little below the lowest-elevation forest present at each site. Figure 4 provides a visual summary of the 1961-1990 climate normals

CMI status of both the study region in general (the map) and our island forest study sites (the bar chart). Figures 5 through 13 provide visual summaries of expected CMIs at different time periods as generated from our 3 marker climate change scenarios. Table 7 provides a numerical summary of the expected CMI values around the maximum elevation point of each study site island forest. All lower elevation points would exhibit lower, i.e. drier, CMIs.

Table 7: Projected CMI values at maximum elevation in each island forest

	Maximum elevation	Normals	**HadCM3 B21**			**CSIROMk2b B11**			**CGCM2 A21**		
	metres	1961-90	2020s	2050s	2080s	2020s	2050s	2080s	2020s	2050s	2080s
SGH (West Butte)	2128	18.3	8.8	-4.6	-5.9	8.4	-2.2	-8.7	8.1	-2.8	-13.6
SGH (East Butte)	2114	18.5	8.6	-4.8	-7.0	8.8	-1.7	-8.2	8.3	-3.0	-13.9
CH (West Block)	1465	-5.2	-13.5	-25.6	-27.2	-14.0	-24.4	-32.5	-15.4	-27.8	-38.3
CH (Centre Block)	1290	-3.2	-9.9	-21.9	-21.9	-10.5	-21.6	-31.9	-12.3	-24.6	-32.4
Moose Mountain	831	-5.4	-9.9	-18.5	-23.0	-19.0	-28.7	-40.4	-19.1	-33.0	-45.3
Turtle Mountain	775	-4.1	-9.0	-18.1	-22.2	-18.2	-28.1	-40.3	-18.4	-32.3	-44.6
Spruce Woods	380	-2.3	-5.6	-12.9	-16.6	-15.4	-24.6	-36.0	-15.6	-29.0	-38.2

There are several points worth noting. First, the island forests are relatively small sites such that, although we estimate CMIs for the geographic centres of each forest, these centre-point CMIs are in practice applicable anywhere within the relevant forest. Interpolating a new climate station and associated CMI values at the edge of an island forest would result in a CMI (for a given elevation, scenario and time period) only trivially different to that calculated for the forest centre site. Second, our 1961-1990 interpolated CMI values for the island forest study sites seem "reasonable" in the sense that they accord well with current vegetation found at these sites. Third, the magnitude and direction of future CMI change as indicated by our 3 marker scenarios (numerically summarised in Table 6) are fully independent of our interpolation of 1961-1990 CMIs at each of our island forest study sites. For example, imagine we have erred in some fashion in our calculations in appendix F, such that the 1961-1990 normals CMI at the peak of Moose Mountain is not really -5.4 as indicated in Table 7, but -3.3. In that case the decline in CMI of -4.5 predicted by the Hadley model scenario for the 2020s (Table 6) would not be affected, and the CMI value for the 2020s shown in Table 7 would be -7.8 instead of -9.9.

Finally, it is clear from Figures 5 through 13 that the scenarios project discouragingly dry CMIs over the Plains region that comprises our study area and which surrounds our island forest study sites. It is beyond the focus of this study to examine the many possible impacts these xeric conditions will have in this region. However, it is obvious, for example, that the impacts on agriculture may be severe. One highly probable outcome is that wherever forest or trees can be preserved in this region, they will be cherished.

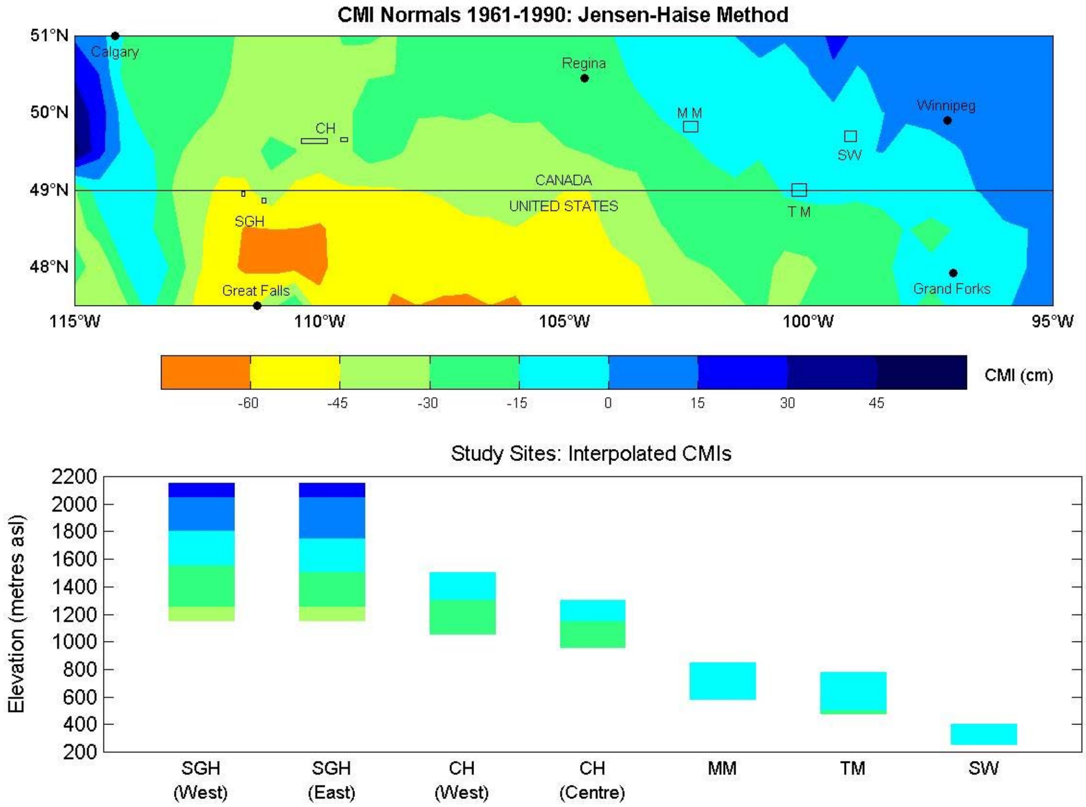

Figure 4: CMI values (cm) for the 1961-1990 climate normal period calculated using the Jensen-Haise method. The map illustrates interpolated CMI values across the island forest study region. The bar chart indicates CMI values with respect to elevation at each study site. SGH – Sweet Grass Hills; CH – Cypress Hills; MM – Moose Mountain; TM – Turtle Mountain; SW – Spruce Woods.

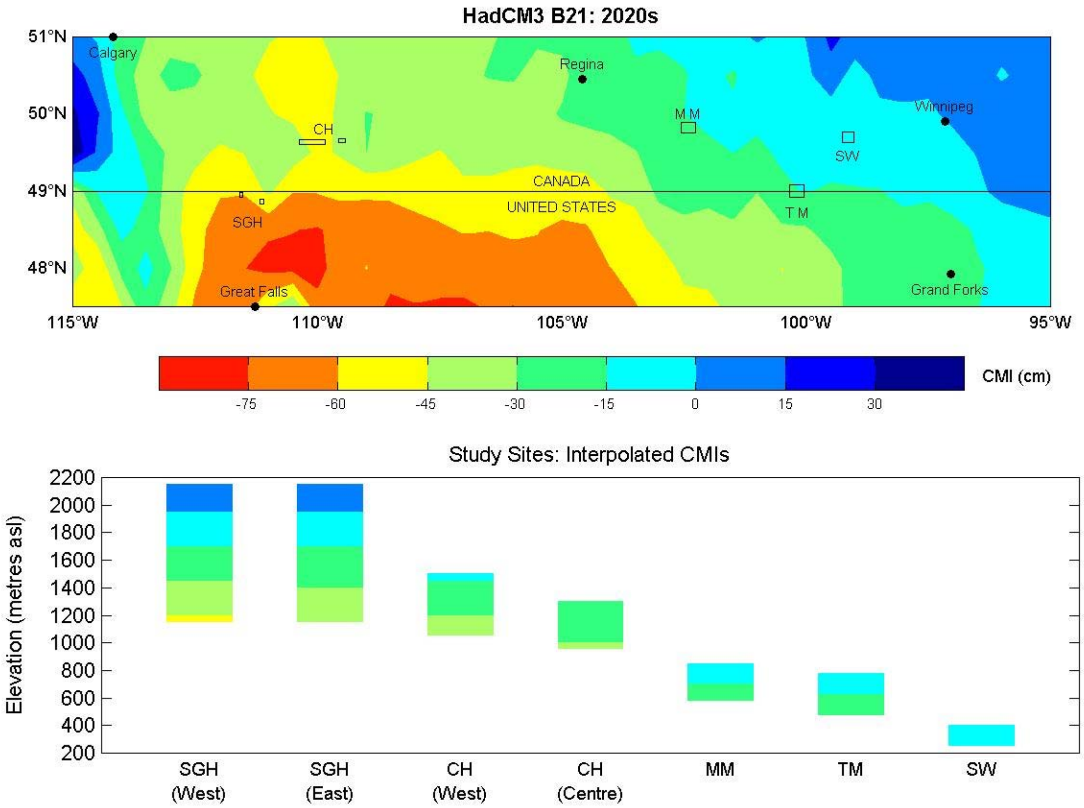

Figure 5: CMI values (cm) for the 2020s according to the HadCM3 B21 (cool-wet) scenario. The map illustrates interpolated CMI values across the island forest study region. The bar chart indicates CMI values with respect to elevation at each study site. SGH – Sweet Grass Hills; CH – Cypress Hills; MM – Moose Mountain; TM – Turtle Mountain; SW – Spruce Woods.

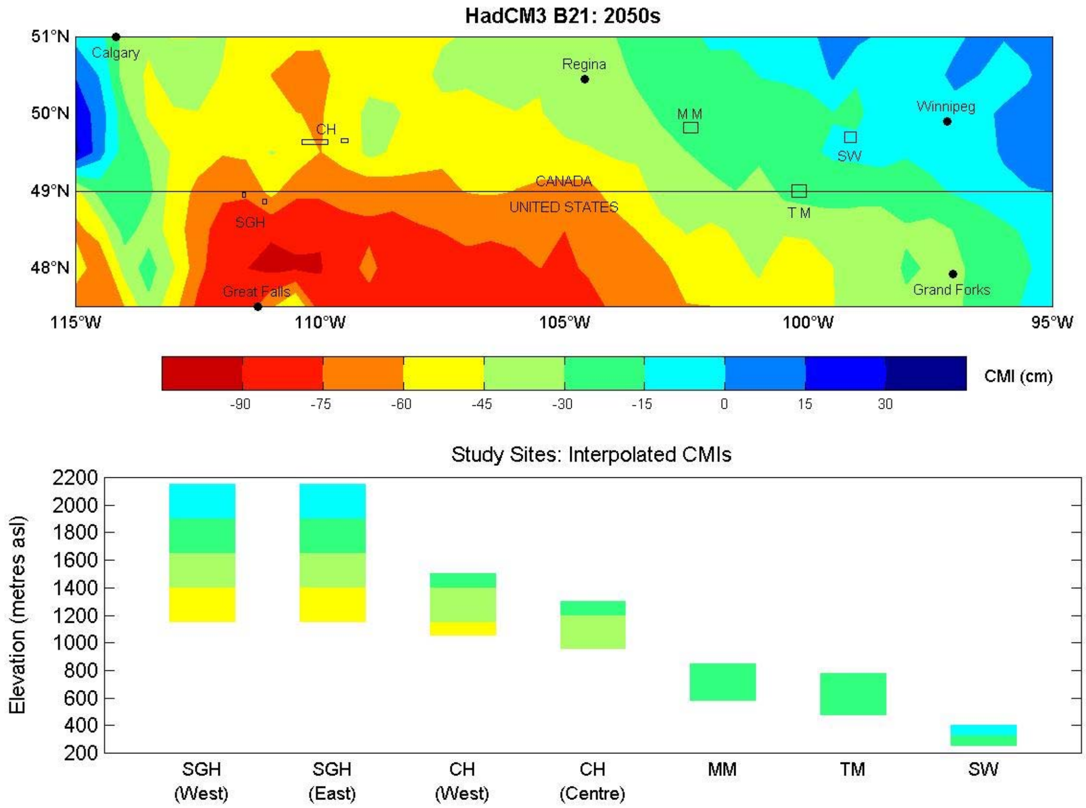

Figure 6: CMI values (cm) for the 2050s according to the HadCM3 B21 (cool-wet) scenario. The map illustrates interpolated CMI values across the island forest study region. The bar chart indicates CMI values with respect to elevation at each study site. SGH – Sweet Grass Hills; CH – Cypress Hills; MM – Moose Mountain; TM – Turtle Mountain; SW – Spruce Woods.

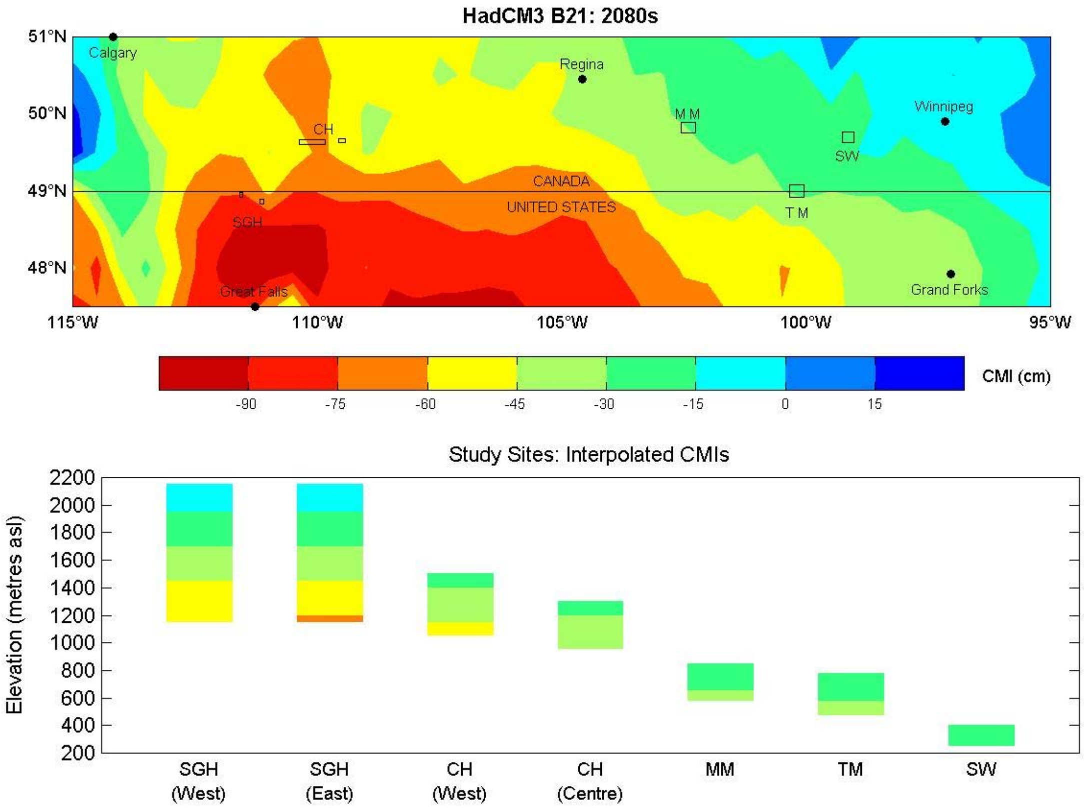

Figure 7: CMI values (cm) for the 2080s according to the HadCM3 B21 (cool-wet) scenario. The map illustrates interpolated CMI values across the island forest study region. The bar chart indicates CMI values with respect to elevation at each study site. SGH – Sweet Grass Hills; CH – Cypress Hills; MM – Moose Mountain; TM – Turtle Mountain; SW – Spruce Woods.

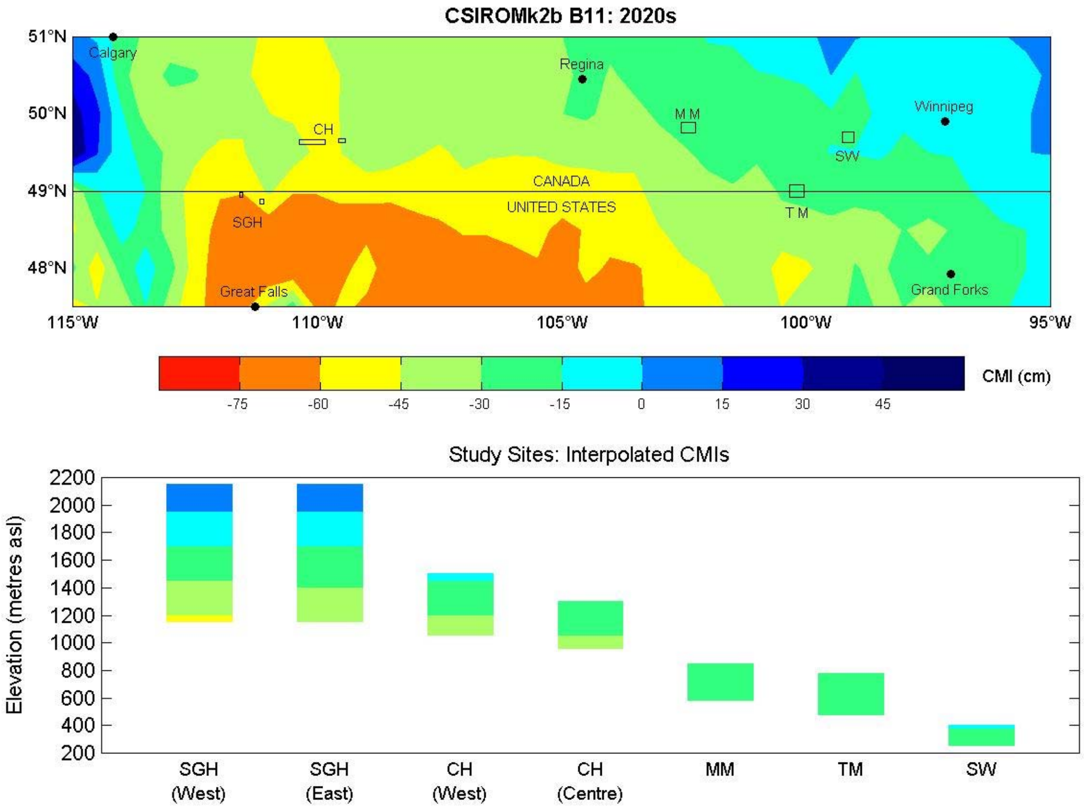

Figure 8: CMI values (cm) for the 2020s according to the CSIROMk2b B11 (mid-range) scenario. The map illustrates interpolated CMI values across the island forest study region. The bar chart indicates CMI values with respect to elevation at each study site. SGH – Sweet Grass Hills; CH – Cypress Hills; MM – Moose Mountain; TM – Turtle Mountain; SW – Spruce Woods.

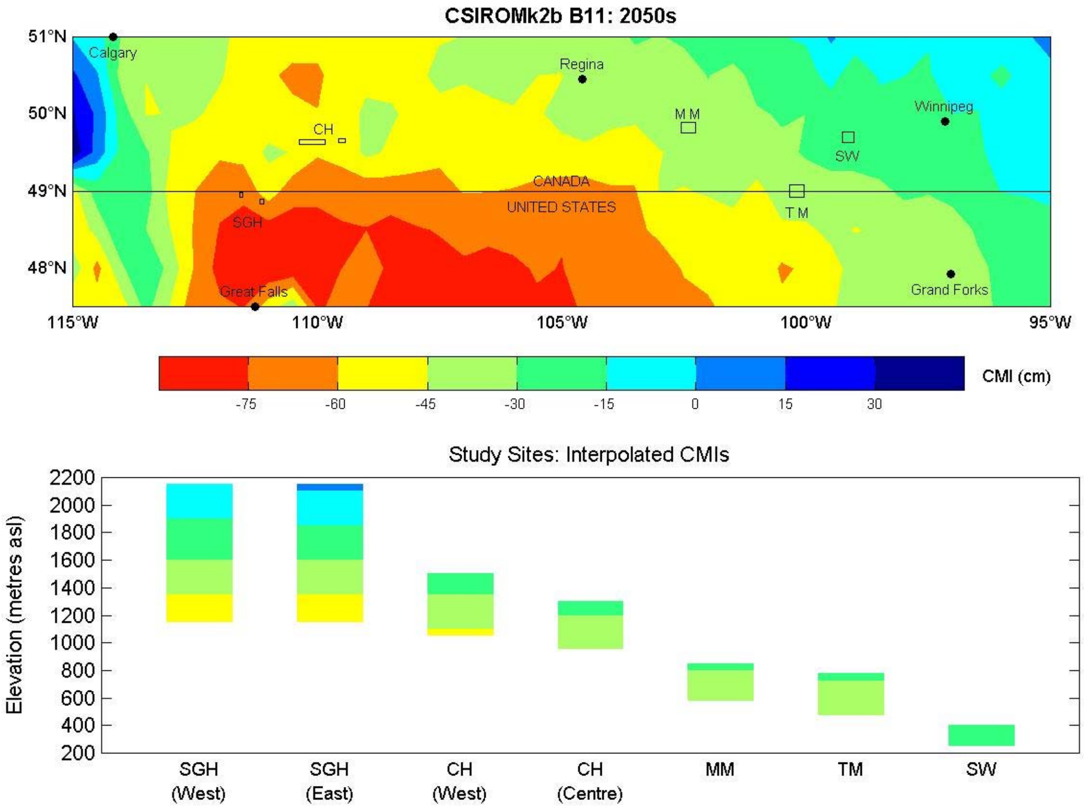

Figure 9: CMI values (cm) for the 2050s according to the CSIROMk2b B11 (mid-range) scenario. The map illustrates interpolated CMI values across the island forest study region. The bar chart indicates CMI values with respect to elevation at each study site. SGH – Sweet Grass Hills; CH – Cypress Hills; MM – Moose Mountain; TM – Turtle Mountain; SW – Spruce Woods.

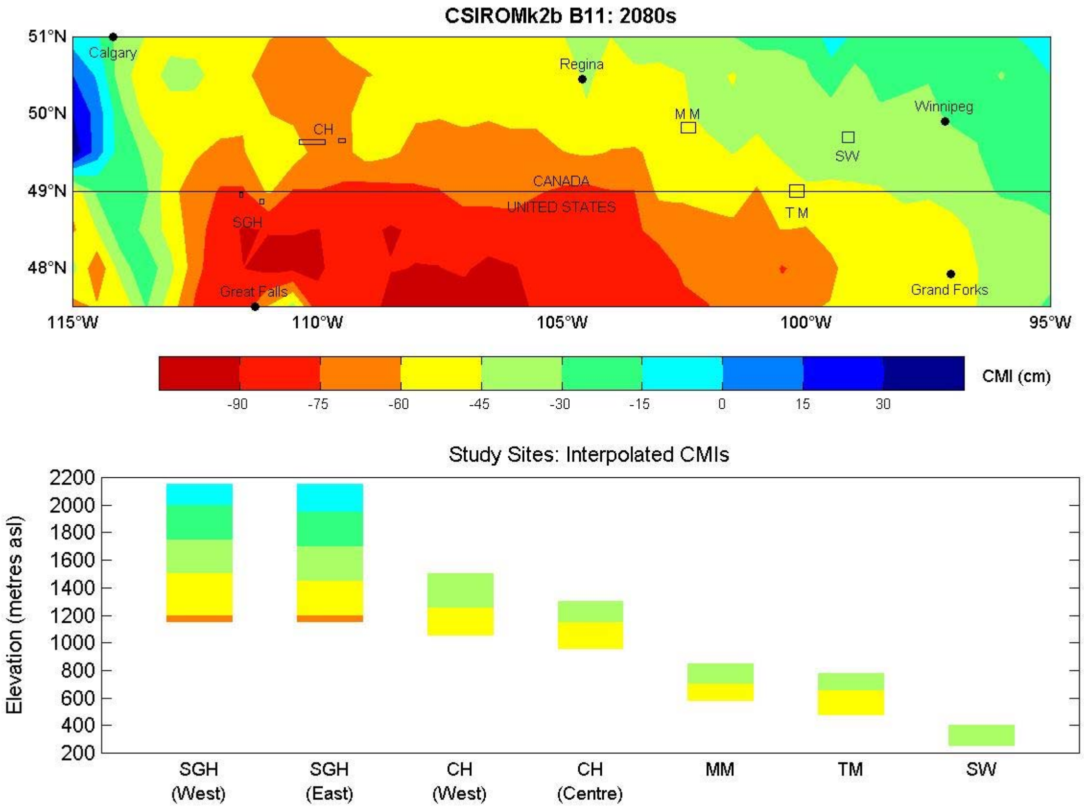

Figure 10: CMI values (cm) for the 2080s according to the CSIROMk2b B11 (mid-range) scenario. The map illustrates interpolated CMI values across the island forest study region. The bar chart indicates CMI values with respect to elevation at each study site. SGH – Sweet Grass Hills; CH – Cypress Hills; MM – Moose Mountain; TM – Turtle Mountain; SW – Spruce Woods.

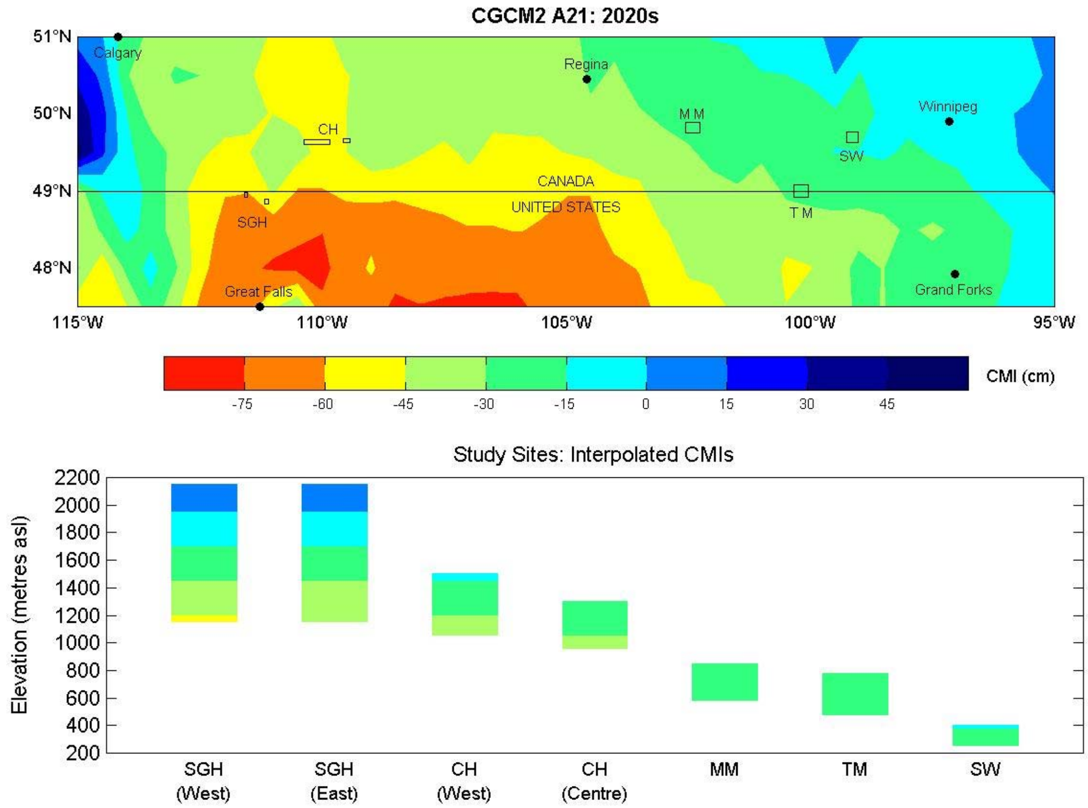

Figure 11: CMI values (cm) for the 2020s according to the CGCM2 A21 (warm-dry) scenario. The map illustrates interpolated CMI values across the island forest study region. The bar chart indicates CMI values with respect to elevation at each study site. SGH – Sweet Grass Hills; CH – Cypress Hills; MM – Moose Mountain; TM – Turtle Mountain; SW – Spruce Woods.

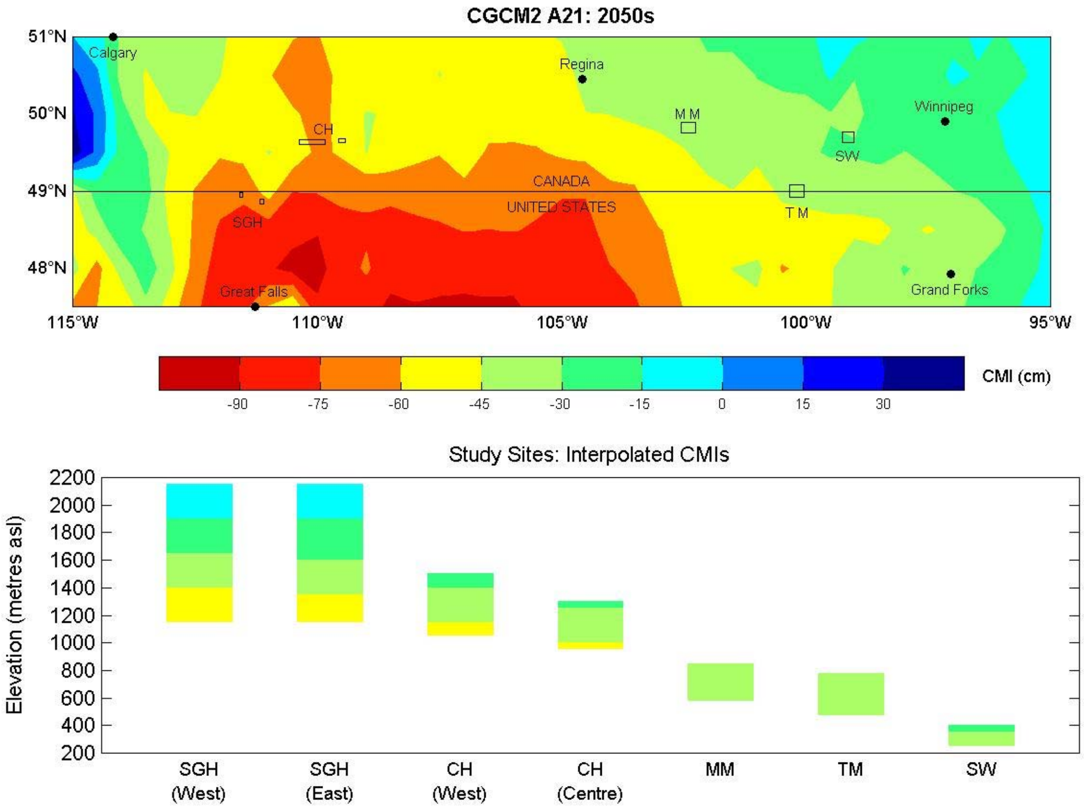

Figure 12: CMI values (cm) for the 2050s according to the CGCM2 A21 (warm-dry) scenario. The map illustrates interpolated CMI values across the island forest study region. The bar chart indicates CMI values with respect to elevation at each study site. SGH – Sweet Grass Hills; CH – Cypress Hills; MM – Moose Mountain; TM – Turtle Mountain; SW – Spruce Woods.

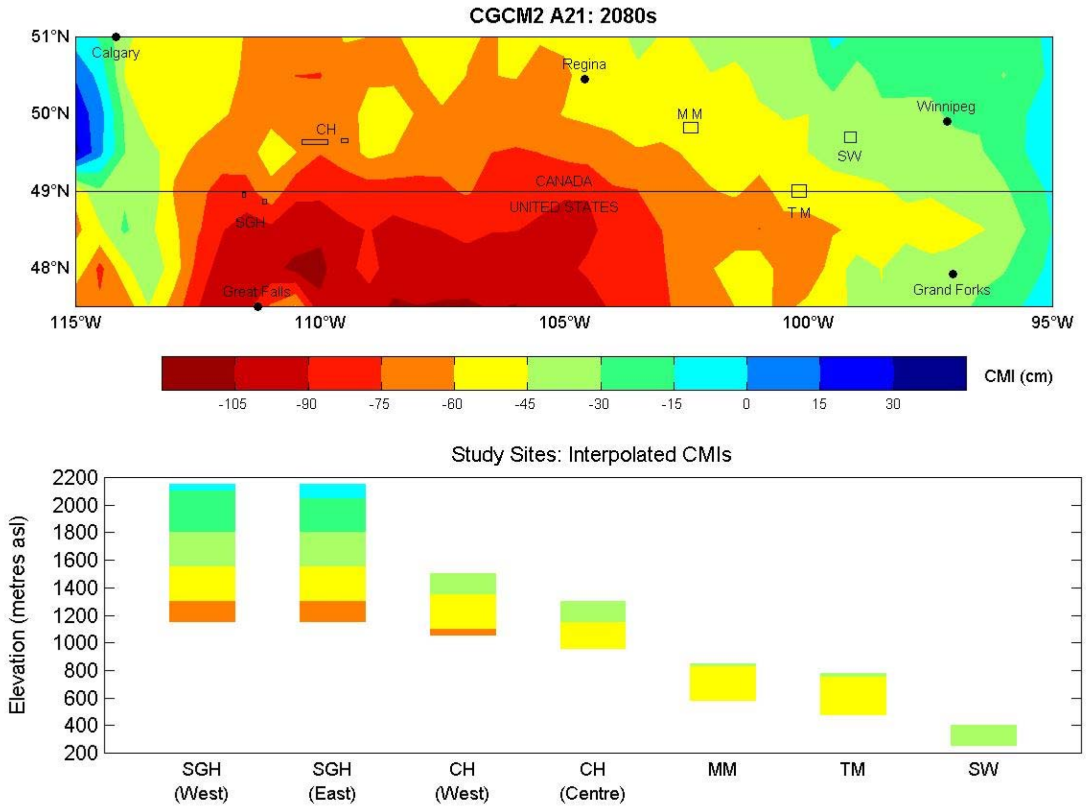

Figure 13: CMI values (cm) for the 2080s according to the CGCM2 A21 (warm-dry) scenario. The map illustrates interpolated CMI values across the island forest study region. The bar chart indicates CMI values with respect to elevation at each study site. SGH – Sweet Grass Hills; CH – Cypress Hills; MM – Moose Mountain; TM – Turtle Mountain; SW – Spruce Woods.

6. MOOSE MOUNTAIN

(a) Site Environment and Landscape

With a maximum elevation of 831 metres the Moose Mountain upland is the highest area in southeastern Saskatchewan. The "mountain," about 400 km^2 in size, rises about 135 to 205 metres above the surrounding plains. Its very irregular rolling moraine terrain creates broken surface drainage, although it is believed that many surface water bodies are linked by subsurface flows. Water levels of the many small shallow water bodies fluctuate greatly. There were notable droughts in the 1930s and 1980s and a high water table period in the 1950s (TAEM 1992; Vance et al. 1997). White Bear Lake, a major water body at Moose Mountain, was very low in the late-19th century (Sauchyn 1995). In general terms the landscape can be described as a mature aspen-dominated hardwood forest containing perennial and seasonal wetlands, ponds and small lakes in frequent surface depressions (TAEM et al. 1992). Dominant soils fall within the grey to dark grey zones (Donauer 1992). According to government survey data, in 1979 about 68% of the provincial park was forested, 17% was water, 6% was muskeg, 5% was "open land" (likely mud flats from retreating water bodies), and about 2.7% grassland (SERM 1979). The landscape ecology can be termed "uniformly heterogeneous," i.e. containing considerable site diversity, largely dependent on very locally determined moisture conditions, while in a broad sense being quite uniform, i.e. the same ecosystem units repeat themselves again and again.

(b) Forest History and Nature

Vance et al. (1997) note an extreme water deficit in the Moose Mountain upland between 4000 and 3000 BP. Palynology evidence from Kenosee Lake, a large lake within Moose Mountain, suggests that over the most recent 1640 years the landscape has evolved from a mixed grassland and aspen forest landscape into a landscape dominated more and more by poplar (likely aspen poplar: balsam and aspen poplar are not readily distinguished in the pollen record). Birch pollen peaked in the period 1220 BP to 700 BP (TAEM 1992). Vance et al. (1997) suggest that the presence of significant amounts of birch in the present landscape is a relatively recent development and dates from the Little Ice Age (circa 1450-1850). Ash pollen is present in small amounts through the pollen record (TAEM 1992).

Palliser (1863) described Moose Mountain as he saw it in 1857 as a series "of lofty irregular mounts, densely covered with woods and enclosing hundreds of beautiful lakes, some of which are of considerable size." Palliser also noted the effect of aspect on the vegetation: "The view of the south side of Moose Mountain was very different to that of the north, being altogether destitute of wood, and it is said that the south side of Turtle Mountain has the same peculiarity" (Spry 1968). John Macoun (1882) noted: "I am safe in stating that there are at least 100 square miles of good timber, nearly all balsam and aspen poplar. Occasionally a few small ash and ash-leaved maple appear, but these are of no value. There is abundance of water in the hills, nearly one-fourth of the surface being covered by it, but the greater part of it is brackish, being in isolated ponds…The whole country to the north of the continuous wood consists of ridges, ponds, lakelets, and hay marshes, with very little level land." Macoun (1882) also claims to have seen oak at Moose Mountain. Dominion land surveyors' reports from 1881-85 give a slightly more detailed picture. A summary of these reports indicates a densely forested landscape dominated by aspen, with the frequent presence of a thick understory, particularly of hazel, and sometimes the presence of significant windfall (Intera 1978). The general impression is of a forest similar to that of the present day.

The build-up of fuel load after European settlement, combined with a period of drought in the 1890s, led to a major fire in 1897 (apparently started by 2 men subsequently fined $100) (TAEM 1992). Most mature trees now date from 1897-1915, i.e. from subsequent to this fire (Donauer 1992).

Five tree species make up the modern forest overstory: aspen, balsam poplar, green ash, white birch and Manitoba maple. Dominant understory shrubs are beaked hazelnut, chokecherry and Saskatoon (Donauer 1992). Other important understory components are red-osier dogwood, rose, willows and thin-leaved snowberry. Browsing pressure from moose, elk and deer has resulted in an even understory of about 1.5 to 2.0 metres in height (Sutherland and Neil 2001). Field sampling in the year 2000 found a distribution of 74.1% aspen, 8.4% balsam poplar, 6.9% birch, and a combined total of 10.5% ash and maple (Sutherland and Neil 2001). Sutherland and Neil (2001) conclude that the current forest is largely dominated by aspen, with balsam poplar prominent in wetter sites. White birch is present sporadically throughout the forest and is generally mature with little regeneration evident. Maple and ash are canopy trees in only a few sites and are found sporadically as subcanopy species throughout the forest. The dominant aspen canopies are becoming more open with time, owing to increasing age, and possibly because of the generally dry conditions of the last decade. There is a high presence of disease and insect impacts. Hazel appears to be increasing in its understory dominance as the aspen canopy thins (Sutherland and Neil 2001).

Relatively recent aspen and balsam poplar colonisation of wetter sites has greatly reduced the wetland meadow component of the park's ecosystem compared to earlier decades. Many local residents report that the forest has lost open areas in living memory. TAEM (1992) noted a decline in permanent standing water of over 62% in the period 1949-88, while Sutherland and Neil (2001) note that many former "wetlands" have been continuously dry in recent years. Climate variability impacts the forest ecosystem as well as the wetlands: the drought of the 1980s led to extensive declines in overstory forest vigour (TAEM 1992). Sutherland and Neil (2001) observe that in the commercial forestry sense aspen production is poor, even compared to low productivity sites in Saskatchewan's Northern Forest.

(c) Site Management and Policy

In 1931 Moose Mountain was converted from a federal forest reserve (dating from 1894) to a provincial park of about 403 km^2 in size. The park is currently classified within the provincial system as a natural landscape park, meaning that the management priority is the protection of the natural landscape, both biotic and abiotic. However, oil extraction, hunting, cattle grazing and haying are important resource extraction activities within the park. With the exception of oil extraction, these activities are allowed on the grounds they are sustainable and do not damage the park's ecosystems. The park encompasses the bulk of the forest, but a significant area of forested land is also found within the boundaries of the adjacent White Bear Indian Reserve.

In 1987 government adopted a wide-ranging management strategy for the park (Henderson 1987). This strategy recommended undertaking an aspen forest management study (from which the TAEM 1992 study resulted), but also made a series of preliminary vegetation management recommendations, subject to the results of such a study. These preliminary recommendations included developing a plan to regenerate 25% to 50% of the park over 20 years in order to maintain a variety of successional stages in the forest. The strategy noted the environmental similarities of Moose Mountain with Turtle Mountain and concluded it was very likely that bur oak and white spruce could do well within the Moose Mountain forest, should vegetation policy target increased diversity. These species are not native to Moose Mountain and current conservation policy would not normally allow their introduction. However, healthy introduced white spruce, circa 75 years old, can be seen growing and regenerating at the site of a long-abandoned federal forestry station within the park (Henderson 1987).

TAEM's (1992) analysis hypothesised the likely successional patterns to be expected in the forest, assuming continued forest fire suppression and no climate change. Green ash and Manitoba maple components of the forest were expected to increase in prominence, while aspen, balsam poplar, and birch would experience a relative decline. There would also be a further loss of grassland ecosystems. TAEM (1992) also suggested that a hazel-dominated understory could become much more prevalent as the overstory thinned through age and decay. In the absence of fire this understory could be dense enough to prevent the regrowth of new trees. In contrast Sutherland and Neil (2001) do not believe that, in the absence of active regeneration, maple and ash will emerge as a dominant overstory. They suspect expansion of the understory in response to continued aspen overstory decay is a more likely scenario.

Key recommendations of TAEM's 1992 vegetation management plan included (Sutherland and Neil 2001):

- manage about half of the park as an old growth hardwood forest by allowing unimpeded ageing and successional processes to occur;
- expand grassland areas to their pre-1949 state;
- harvest and regenerate birch in a portion of the park; and,
- harvest aspen in about half of the park to diversify the age class structure of the forest.

Though approved by government, the TAEM plan was never actually put into effect and has now been superseded by a new approved plan (Sutherland and Neil 2001). This latter plan confirms the 3 overarching goals of the earlier TAEM plan:

- preservation and perpetuation of an environment which gives the park its recreational and educational value;
- preservation and, where possible, perpetuation of representative and/or unique floral and faunal features found in the park; and,
- maintenance of a diversity of cover types within the park.

These goals are so general that many future vegetation landscapes could conceivably fall within their bounds. However, the new plan also includes explicit specific objectives or actions:

- the return of a significant portion of the forest to a seral stage;
- the regeneration of birch; and,
- the expansion of grasslands.

Sutherland and Neil (2001) note that it is possible that aspen harvested in the next few years may still have some commercial value, but that waiting a few years longer may result in much timber having degenerated to the point of no commercial value. Without some commercial value to the harvested wood it may be impossible to finance regeneration. For this reason they envision an aggressive forest regeneration program that will regenerate about 50% of the park over a 5-to-10-year period, rather than a sustained slower rotation that might see the forest regenerated over a much longer time frame. By comparison the operative vegetation management plan for Saskatchewan's Duck Mountain Provincial Park (Wright et al. 1995) only envisions regeneration of up to 20% of that park's aspen forest by 2019. It remains to be seen whether the wood at Moose Mountain has commercial value now and, if so, whether it would lose that value in the near future.

Other regeneration techniques than harvest have not been finalised in the new Moose Mountain plan and may include a variety of methods, such as girdling, prescribed fire, aerial herbicide spraying, livestock grazing, or mowing. Equally, to stimulate regeneration, a variety of after-treatments may be employed as required, such as scarification. Some of the uncertainty as to what techniques will be employed derives from the fact that we have little practical experience in regenerating old growth aspen forest at Moose Mountain. To a degree, the regeneration efforts will be experimental.

Wildfires would be suppressed if possible, given the practical need to protect values-at-risk such as cattle, oil extraction installations, or cottages. Neil (pers. comm. 2002) notes that aspen in North America is now burning more and more often under previously unexpected conditions. Donauer (1992) notes that aspen flammability is highly variable. The wildfire threat at Moose Mountain is not clear and some would argue it is more often difficult to get aspen to burn than to extinguish it. However, it is certainly possible that the Moose Mountain forest could pose a fire threat, given the high fuel loads and dry enough weather. Historically, of course, the forest has burned.

(d) Management Analysis and Vegetation Outlook

The current vegetation plan's objectives are explicit and the possible techniques to achieve these objectives are outlined, but the plan (Sutherland and Neil 2001) is not complete in that it does not contain an explicit vision of the desired vegetation ecosystem across the forest nor an explicit rationale explaining why that targeted end state is preferable to all other options. The plan does not present vegetation ecosystem alternatives and discuss their relative advantages and disadvantages. Other forest managers (or stakeholders) could conceivably arrive at different conclusions. As detailed below, in the similar forest of Manitoba's Turtle Mountain Provincial Park, there is no regeneration plan in effect (although on the American side of Turtle Mountain aspen regeneration is undertaken). Peterson and Peterson (1992a) note that there are legitimate ecosystem roles for aspen stands that have aged beyond the point of economic commercial harvest. These roles include providing wildlife habitat, the retention and recycling of on-site nutrients, or the retention of carbon.

Why undertake regeneration at all? In the popular mind, the idea that the forest is falling down around us and that we must therefore act has some appeal, perhaps because we make an unconscious comparison between a decaying forest and a decaying and unsafe building. But this comparison is misleading; an ecosystem is not a building, and decaying trees are simply members of a different ecosystem to that associated with younger trees. The commercial forester's prejudice against letting mature timber become "decadent," akin to an engineer's dislike of seeing potential hydro-generation sites go undammed, should have no place in the management of an area dedicated to natural values. Nor is the fact that cattle grazers might benefit from increased pasture as a result of regeneration a consideration from a nature conservation point of view. Notwithstanding, from a conservation perspective there are several possible rationales for regenerating part or all of the Moose Mountain forest:

- the current state of the forest is the result of human impacts (for example, the removal of bison and fire as ecological agents) and is therefore not natural (Sutherland and Neil 2001);
- the forest might be considered more "correct" at some time in the past, perhaps prior to European settlement and interventions;
- almost any change in some parts of the forest from dominant over-age aspen would be an improvement, in terms of greater diversity, be the outcome grassland, birch or mixed-age aspen stands; and,
- regeneration may lessen the danger of major wildfires, a danger that may be increasing with the rising fuel load, increasing stand mortality, and drought.

None of these issues are simple. We discuss them in the context of all 5 island forest study sites in the report chapters that follow our discussion of the individual island forests. But whatever the merits of the approved regeneration plan at Moose Mountain, it is likely that the lack of a widely publicised and agreed vision for the Moose Mountain landscape will cause serious difficulties as soon as any controversial regeneration activities commence – and aggressive regeneration on the scale proposed is bound to be controversial with some of the many stakeholder groups with an interest in Moose Mountain.

How would a regeneration plan fare in practice given the expected impacts of climate change? In general, regenerating an aspen-birch forest through the harvest of (over)mature aspen and birch may not ultimately prove successful. It may be the case that the current moisture balance already constrains new tree growth such that we are not likely to see regrowth of aspen at Moose Mountain to the size of the existing mature trees. Under drier future conditions aspen might regenerate only to a relatively small and stunted size. Birch, which is not tolerant of drought conditions, preferring short, cool summers (Safford et al. 1990), will have a harder time of it still. On the other hand, conversion to grassland (if desired) should be possible at many forest sites.

Regeneration outcomes can be expected to be complex and dependent on localised topographic considerations. As conditions become more xeric, the many small lakes and ponds of Moose Mountain will shrink. For a time, aspen, and especially balsam poplar, will be able to advance and prosper over these relatively damp new meadows.

Simultaneously, on thin-soiled, south-facing sites within the forest, dieback may occur. It is conceivable these latter sites will revert to bush and grassland. This reversion process will likely occur in any case, but could occur more quickly via human intervention.

In terms of climatically possible target landscapes for active management, a regeneration plan that wished to "go with the (climate) flow" would target increasing the percentage component of grassland in the park, perhaps by clearing mature stands from drier sites. The location of the old grasslands is a key management question that should be addressed before proceeding with any regeneration plan. Were historic grassland enclaves on exposed and thin-soiled dry sites that aspen subsequently colonised in the absence of fire? Or were they on low and damp sites that aspen colonised as ponds retreated? An examination of old and recent aerial photographs (both of which are available), combined with soil, moisture and aspect analysis of potential regeneration areas of the forest, would allow a reasonably solid evaluation of which sites were once grassland and could most likely be returned to that state.

Wetlands at Moose Mountain will likely by severely impacted by climate change. Grassland and bush can take over droughty sites, while for a time aspen and balsam poplar can migrate onto contracting wetland sites, but ponds, lakes and wetlands look set to shrink in absolute terms. Where water remains, it will be warmer in summer, with a concomitant decrease in dissolved oxygen and greater susceptibility to algal bloom. Maintaining any sport fishery at Moose Mountain (already a difficult proposition) will likely be impossible. Boating, swimming and the extraction of water for the golf course and waterslide complex may be negatively affected.

Over the long term, the scenarios suggest that in the absence of active management the forest could change to a very open parkland state, with trees restricted to low hollows and northern exposures (of which latter there are few significant blocks – Moose Mountain lacks the extensive northern slopes found at Cypress Hills). As a transition to that state, widespread aspen stand break-up is likely as the CMI decreases.

Break-up occurs as holes begin to appear in the canopy with the loss of growth vigour, i.e. when crowns in surviving trees cannot grow fast enough to compensate for voids left by dying trees (Perala 1990). These holes cause sudden environmental changes in sunlight, wind and evaporation, which stress neighbouring, ageing aspen. Insect and disease attacks become more frequent, and mechanical breakage of trees occurs. Increasing impacts from forest tent caterpillars, fungal pathogens and wood-boring insects can be expected (Hogg 2001). Break-up and deterioration can be rapid (Perala 1990). However, the details of aspen stand break-up are site and event dependent and are not yet fully understood in the northern Plains context (Peterson and Peterson 1992a). In theory, many outcomes are possible. Understory shrubs may take advantage of increased sunlight, or previously subcanopy tree species may emerge to form a new canopy, or a site may evolve towards grassland (Sutherland and Neil 2001). However, Perala (1990) notes that dry sites may revert to rangeland and at Moose Mountain the climate scenarios suggest movement in the direction of a more open landscape is very likely.

It is also possible that high fuel loads and low water levels increasingly make Moose Mountain a candidate for another major fire, as occurred in the late-19th century. If not so hot as to destroy aspen root stock, such a fire could encourage suckering during subsequent moist years, but at most sites suckers might survive only a few years until killed by the next droughty period.

Moose Mountain's status as a forest is highly vulnerable. Even before climate change is considered, only a few hundred metres of elevation separate Moose Mountain from the surrounding prairie ecosystem. Regeneration of existing tree species will prove increasingly difficult. The forest lacks tree species diversity that could provide robustness. It also lacks significant refugia such as north-facing slopes, steep coulees or major rivers with water input from more humid regions. It is clear that management faces some difficult – and interesting – choices. The chapters following our discussion of the individual island forests take this discussion further.

7. TURTLE MOUNTAIN

(a) Site Environment and Landscape

The Turtle Mountain landscape is very similar to that of Moose Mountain: undulating hills; marsh and wetlands; numerous small lakes, shallow water bodies and swampy depressions; and thick, mature, predominantly aspen, deciduous forest – the only upland hardwood forest in Manitoba. Macoun (1882) himself stated that the "distributions of wood" found on Turtle and Moose Mountains were "exact counterparts." Turtle Mountain, which is actually an eastern outlier of the Missouri Coteau, rises some 180 to 240 metres above the surrounding plains (Cassel and Stewart 1968) and is roughly bisected by the Manitoba – North Dakota international boundary. As at Moose Mountain, irregular morainic deposits overlie bedrock. The majority of soils on the Canadian side are loam or clay loam, are high in nutrients, and vary from imperfectly to well drained (MNR 1985). On the American side soils are described as typically shallow and well drained (NDPRD 1996). Subsoils tend to be clayey, impeding drainage, such that adjacent ponds can be found at different elevations (Manitoba Conservation 2001). Internal drainage between water bodies is intermittent and there is little runoff to the surrounding plains (MNR 1985). Instead lakes and wetlands rise and fall in response to local moisture conditions. Even the bigger lakes are shallow, such that winterkill makes maintaining fish populations impossible except in one or two lakes on either side of the boundary. The late 1980s and early 1990s were dry periods that lowered water levels on the Mountain. The drought of the 1930s was much more severe, reducing one major water body, Adam Lake, to a small pond (Manitoba Conservation 2001). Recent years have again been dry, but water levels remain higher than in 1992, the low point of the next-to-current dry period.

(b) Forest History and Nature

Although they disagree on issues of timing, both Love (1959) and Ritchie (1976) describe a similar post-glacial vegetation history for the Turtle Mountain area. Spruce forest became established following glacial retreat. The forest was replaced by grasslands during the hypsithermal, which were in turn replaced by the current deciduous forest as a cooler and more humid climate developed.

Photographs taken around 1873 by international boundary survey parties (viewed by one study author at the local museum in Boissevain) show bare hilltops in parts of the Turtle Mountain formation that are now heavily wooded. Aerial photographs beginning in 1938 and extending into the 1980s of Lake Metigoshe State Park (immediately south of the international boundary) show an increasing density and extent of woodland (NDPRD 1996). This seems consistent with the historic evolution of other island forest sites on the Plains.

Dawson (1875) visited Turtle Mountain in 1874 and described it as "a more or less thickly wooded area." The northern, British, half of the formation he described as "more uniformly covered with woods, and probably embraces two-thirds of the forest area." He noted aspen and balsam poplar as the main forest species, with abundant bur oak along the forest margins.

In 1895 much of the Canadian side of Turtle Mountain was designated a federal forest reserve (Smith 1970). In 1959 most of the "East Block" of this forest reserve was converted to community pasture. This area exists today as a quite heavily wooded AAFC-PFRA (federal agriculture department) Community Pasture. In 1961 186 km^2 of the western section of the forested area were designated Turtle Mountain Provincial Park. The entire park, plus a small extension to the east, remains classed as Provincial Forest Reserve. In places the forest extends into privately held land to the west and north of the provincial park. An International Peace Garden is adjacent to the southeastern corner of the provincial park and straddles the international boundary. South of the boundary there are also large areas of contiguous tree cover. One of the most ecologically interesting of these is Twisted Oaks Recreation Area, where bur oak descends into grassland on the dry southwestern slopes of Turtle Mountain. In the driest portions of this pure oak woodland the impression is savannah-like.

Major fires in 1897 and 1903 burned almost all of the forested region now within provincial park boundaries. The site of future Lake Metigoshe State Park was largely burned in 1886 (NDPRD 1996). In 1921 a fire originating in North Dakota again consumed a large part of the Canadian forest (Manitoba Conservation 2001).

A 1953 forest survey of the Turtle Mountain Forest Reserve (DNMR 1960) indicated that 92% of the total land area within the reserve was wooded at that time. Much of this 92% was described as "poorly stocked," particularly low-lying areas with willow cover (849 ha of willow were noted as "non-productive forested land") and also higher land with bur oak and shrub cover. Such areas were not thought to be or to become of much commercial value, at least under the species mix and conditions at the time of survey. Wood volumes of those trees of at least 4 inches dbh (i.e. of a diameter of at least 10.08 cm at 1.37 metres above ground surface) were assessed as follows: aspen 68.1%, white birch 9.2%, balsam poplar 7.2%, green ash 7.1%, bur oak 6.5%, white elm 1.8% and Manitoba maple 0.1%. The survey parties described most stands as uneven-aged and therefore not amenable to age-stand classification. Manitoba Forest Inventory Reports (year 1977 and 1982 data on file in Winnipeg) provide a later assessment based on species area coverage (rather than on wood volumes) in "productive forested lands" within the provincial forest reserve. In descending order of area coverage the reports indicate the following species: aspen, bur oak, green ash, balsam poplar, white spruce, Manitoba maple, and Scots pine (spruce and pine are both introductions).

A resource inventory study undertaken by Smith and de Smit (1979) concurs that aspen clearly dominates the provincial park landscape. Beaked hazelnut, dogwoods and willows are noted as the major understory species. But Smith and de Smit (1979) also map out perhaps 6% to 8% of the park as "mixed oak forest." A few small areas are mapped as white birch forest, while Eagle and Arbor islands in Max Lake, presumably spared from fire, are classed as mature oak and elm forest, with some ash and maple present and aspen entirely absent.

Following on from Smith and de Smit, Guinan and Rewcastle (1982) conducted a systematic vegetation survey of the provincial park in 1979. They confirmed aspen as the most abundant tree species, noting its tendency to form a canopy at about 18.2 metres height throughout the park. Bur oak and green ash were common understory components, the former seemingly suppressed by browsing deer or hares. Balsam poplar was found in reasonable numbers, as was white birch. Manitoba maple and American elm were relatively infrequent. Willow (*Salix* spp.) and cherry (*Prunus* spp.) were noted as common shrubs. Guinan and Rewcastle (1982) disagree with Smith and de Smit (1979) about Eagle and Arbor Islands, noting that the dominant tree in numbers and basal area on Eagle Island is in fact ash, followed by Manitoba maple. Significant amounts of oak, elm, balsam poplar and birch were also noted on Eagle Island. On Arbor Island Guinan and Rewcastle (1982) found evidence of human disturbance throughout, with oak, ash, birch, elm and balsam poplar all present as mature trees. Scrubby aspen was present as well. Oak, they observed at Arbor Island, seemed to favour the establishment of a grass understory and possibly discouraged hazel. Guinan and Rewcastle (1982) also identified a third site of similar mature mixed forest, with the addition of a hazel understory, on the Adam Lake peninsula, where it was presumably also sheltered by topography from past fires. The persistence of hazel under this particular mature forest they hypothesised to be a result of cattle grazing or of a sparser canopy cover.

Guinan and Rewcastle (1982) found beaked hazel to be the dominant shrub species overall. Hazel, they observed, was most successful under an aspen canopy, where it formed dense and almost pure stands about 2 metres in height. Exposure to direct sunlight seemed to stunt growth. Under later successional canopies, for example, under ash or oak, hazel tended to be replaced by chokecherry and Saskatoon. Guinan and Rewcastle (1982) also noted canopy and subcanopy species frequency variations dependent on different soil types within the forest. Aspen, balsam poplar, ash and maple were more prevalent on relatively moister sites, while birch, oak and elm were more prevalent on drier sites. The park management plan (MNR 1985) differs slightly, stating that ash and oak are site-specific to dry soils and southern exposures. At Lake Metigoshe State Park, balsam poplar and maple are noted as prevalent in moister sites, birch in mesic sites, ash on a variety of sites, elm on mesic to dry sites, and oak on drier sites (NDPRD 1996). Guinan and Rewcastle (1982) concluded that the increased diversity of slope and moisture conditions found in areas of rolling topography in the forest seemed to accelerate succession to vegetation canopies beyond pioneering aspen.

The 1985 provincial park management plan (MNR 1985) confirmed that the majority of the park consists of 55+ year-old aspen stands. In the northern and eastern areas of the park the general state of the aspen forest seems comparable to that at Moose Mountain (pers. obs. 2002), although the aspen canopy at Turtle Mountain appears taller at many sites, perhaps indicating better growing conditions. Much of the aspen forest is now "overmature" in commercial terms, and breaking up, although perhaps not as extensively as at Moose Mountain. According to Gary Armstrong (pers. comm. 2002), long-term resident foreman in the provincial park, much overmature aspen is dying back. Armstrong also notes good regrowth of birch from small clearcuts made over the last 10 years. As well, aspen, he notes, seems to generally regenerate in disturbed areas. Bur oak seems not to be under stress, while ash may be stressed in some places. Tent caterpillars may be an increasing problem.

The situation in Lake Metigoshe State Park is similar. Much of the dominant aspen woodland is overmature and contains "decadent" aspen. Historically the area is said to have been more age-diverse. However, oak woodlands are noted as in good ecological condition (NDPRD 1996).

(c) Site Management and Policy

There have been numerous experimental conifer plantings in the forest. DNMR (1960) lists the following summary of species and number of trees planted on crown (government) lands in the Turtle Mountain area between 1912 and 1959 inclusive: white spruce 325,000; Scots pine 8,000; other conifers 1,000. White spruce was planted sporadically in plantation blocks from 1912 until 1943, and then again from 1960 into the 1970s. Many of the mature trees continue to do well at Turtle Mountain and are regenerating naturally (pers. obs. 2002). Keith Knowles (pers. comm. 2002), a Manitoba forest biologist with past responsibility for the park, confirms that white spruce stands from 1912 and 1930 have done well, but not more recent plantations, owing to deer browsing. Scots pine, limited to 0.7 ha of plantation, also continues to be present in the park.

In the still theoretically operative 1985 management plan for the provincial park (MNR 1985) the objectives of park management include the protection and maintenance of vegetation and wildlife communities representative of the Turtle Mountain natural region and the protection and preservation of examples of unaltered hardwood forest. These are typical park system objectives for a park classed, as Turtle Mountain is, as "natural." Limited commercial logging (largely for fence posts and firewood) was expressly permitted in the 1985 plan in some zones of the park in order to maintain "a diversity of vegetation types and therefore wildlife habitats" (MNR 1985). However, in 1997 a new *Provincial Parks Act* came into force in Manitoba. The purposes of the provincial park system are defined in the act (MNR 1998a) as:

- to conserve ecosystems and maintain biodiversity;
- to preserve unique and representative natural, cultural and heritage resources; and,
- to provide outdoor recreational and educational opportunities and experiences in a natural setting.

The act has the effect of disallowing commercial logging in protected areas of provincial parks. Government could itself undertake logging for conservation purposes within a park area zoned "backcountry" but could not sell the logs to outside buyers, i.e. the logging could not be "commercial." In effect the act makes it more costly to regenerate forest in protected areas. This is no accident. The act's restrictions on commercial harvest resulted from concerns that much timber extraction previously allowed in various Manitoba parks and protected areas under the guise of conservation objectives was in reality driven by commercial considerations and was inimical to park values. Commercial timber harvest within the resource management zones of Manitoba provincial parks remains theoretically possible, depending on the definition of the particular resource management zone. At present trapping and oil and gas extraction are the only commercial resource extraction activities now allowed within Turtle Mountain Provincial Park.

In general Manitoba does not have broad management plans (including vegetation management plans) in place for provincial parks (Helios Hernandez, parks planner, pers. comm. 2002). Existing park management studies tend to centre around visitation and resource extraction issues. Historically the vegetation focus in Manitoba has been on commercial forest inventory. Detailed up-to-date vegetation inventories of the province do not exist. There has been little recent attention paid to even basic forest inventory within the parks, and also little in the way of ecosystem trends analysis. The latest forest inventory data of non-commercial forests, dating from 1977, were a response to the need to map elm distributions owing to concern about the advance of Dutch Elm Disease. In this context it is perhaps not surprising that there is no forest regeneration plan in place or under development for Turtle Mountain Provincial Park. Natural regeneration via wildfires is currently suppressed owing to the need to protect infrastructure and property.

In the American portion of the forest disjunct portions of state lands are managed for forest conservation or recreation purposes. Significant state-owned areas in Turtle Mountain include Turtle Mountain Recreational Forest (5,707 acres / 2311 ha), Twisted Oaks Recreation Area (1,120 acres / 454 ha), and small portions of the forest unburned in the fire of 1886 or subsequently. There is also a tree nursery at Turtle Mountain (Liebenstein 1984).

The most notable conservation area is Lake Metigoshe State Park, established in 1937, 1,532 acres (620 ha) in size, and one of only 3 state parks in the North Dakota system. Amongst these Metigoshe is noted as containing the greatest area of land remaining in natural condition (NDPRD 1996). The North Dakota state parks system plan states that natural resource protection takes precedence over recreation provision (Liebenstein 1984). The vegetation management objectives at Metigoshe are to protect and enhance forest diversity. A key element of this is interpreted to mean increasing stand-age diversity of the predominantly aspen forest. The operative vegetation management plan makes no mention of climate change (NDPRD 1996).

According to park manager Larry Hagen (pers. comm. 2002) the vegetation plan has been operationalised as follows: in the last one or two years two sites of mature aspen, each 7 to 10 acres (3 to 4 ha) in size, have been winter-cleared and are being monitored for aspen regeneration. In January 2003 a third site will likely be cleared. In addition, in 2005 the park is planning to begin test burning of some open areas to restore native grassland. A tent caterpillar infestation is foreseen, but is not expected to have a long-term effect on vegetation. Herbicides are used on a spot application basis to control invasive exotics, such as leafy spurge.

The North Dakota Forest Service manages the 7,704 acres (3,120 ha) of the Turtle Mountain state forests, including Turtle Mountain Recreational Forest and the Twisted Oaks Recreation Area. According to Tom Karch (pers. comm. 2002) of the Forestry Service, in descending order of priority the management concerns are: recreation, wildlife and plant communities. There is no formal management plan for the state forests, but the service intends to draft one in 2003. Currently the service is inventorying forest conditions. In the absence of fire, some aspen stands are old and dying and some mechanical regeneration is undertaken in a similar fashion to that implemented at Lake Metigoshe. The focus is on improving ruffed grouse, deer and moose habitat. Karch states that in theory the objective is to maintain the landscape in “pre-settlement” condition, but that the Forest Service does not really know exactly what that was. Climate change is not considered within management planning.

The State Game and Fish Department also manages some forested Wildlife Management Areas within Turtle Mountain. Brian Prince (pers. comm. 2002), resource management biologist for the Turtle Mountain area, notes that the department undertakes mechanical removal of aspen on sites between 2.5 and 50 acres (1 to 20 ha) in size. The goal is the replacement of 80-to-90-year-old aspen stands with a diversity of age classes to improve wildlife habitat, particularly for ruffed grouse. Climate change is not considered within management planning.

The federal Fish and Wildlife Service maintains conservation easements on freehold land within 3 National Wildlife Refuges at Turtle Mountain. The easements originated in the drought of the 1930s and their continuing intent is to preserve lakes for wildlife habitat. Recently there has been discussion about removing some easements and 2 of the Turtle Mountain refuges, School Section Lake and Rabb Lake, may be candidates for removal (Lee Albright, District Manager, FWS, pers. comm. 2002).

The Turtle Mountain Chippewa Indian Reservation, located in the southeastern corner of the Turtle Mountain forest, encompasses about 26,000 acres (10,530 ha) of forested lands managed by the Turtle Mountain Tribal Forestry Office. A forest management plan is now operative (Sawyer 1998), a primary objective of which is the regeneration of old aspen stands (Godfrey 1999). According to the responsible forester, Ron Davis (pers. comm. 2002), harvested or cleared areas seem to be regenerating well. The Tribal Forestry Office is also trying to establish areas of pine and spruce, and is planting evergreens to that effect in 10-acre (4 ha) plots. This is an interesting continuation of past plantings of a wide variety of conifers on the reservation. Between 1962 and 1964, Scots pine, ponderosa pine, Colorado spruce and Black Hills spruce were planted in plantation blocks. In 1987 29,000 seedlings were planted, including Siberian larch, Colorado spruce, Black Hills spruce, Norway spruce, Douglas-fir, ponderosa pine, Scots pine, red pine, jack pine, (eastern?) white pine, Austrian pine, white spruce, balsam fir and "white cedar" (northern white-cedar likely intended). From 1989 to 1991, 40,000 seedlings were planted, including "black pine" (Austrian pine likely intended), ponderosa pine, Scots pine, Siberian larch, Black Hills spruce and Colorado spruce. Of this latter planting, there was poor survival except for Colorado spruce and Black Hills spruce, and eventually even these trees were cleared for crop land. In 1997 plantations from previous years were disked and replanted with 18,600 spruce and pine seedlings (Godfrey 1999).

Finally, a major proportion of the American Turtle Mountain forest is privately owned. Craig Stange (pers. comm. 2002), responsible staff forester for the USDA Natural Resources Conservation Service notes that the 2 main uses of forested private land are for grazing and for subdivision for home sites.

The majority of the trans-boundary International Peace Garden is left unmanaged. However, there are many introduced trees. Their provenance and date of planting are typically not known. According to the site's executive director, John McQueen (pers. comm. 2002), plantations go back 60 years.

(d) Management Analysis and Vegetation Outlook

Knowledge of the current forest is patchy and unsystematic. No large-scale comprehensive forest ecological studies have taken place since Guinan and Rewcastle's (1982) 1979 surveys. We also lack time-series studies of the forest's evolution over the past century or so. In the absence of such data it is difficult to draw firm conclusions. For example, unsystematic observations of current aspen stress must be placed in the context of the dry conditions of the past few years.

Guinan and Rewcastle (1982) propose a hypothesis of a typical forest succession at Turtle Mountain. Their explicit assumption is the absence of frequent fire and an implicit assumption is the absence of hare and white-tailed deer, both of which, if present in large numbers, can act as major inhibitors of plant succession to oak, ash and maple climaxes. First, after some major disturbance, aspen and balsam poplar appear, later joined by birch. Under these trees an understory of ash and oak develops. As aspen and balsam poplar decay and break up, ash and oak are able to form a new discontinuous canopy, under which Manitoba maple and elm are able to establish themselves. Ultimately these last 2 tree species become part of a dense canopy composed of ash, oak, maple and elm.

Assuming the continued suppression of wildfires and no other management, DeByle (1985) suggests that aspen can continue to sucker and regenerate even as the overstory breaks up. However, it is thought at Lake Metigoshe that green ash and bur oak will increase in relative dominance (NDPRD 1996). The many aspen regeneration efforts undertaken by various agencies in the American forest offer good opportunities to monitor aspen regeneration success. The conifer plantations planted by the Turtle Mountain Tribal Forestry Office also offer an opportunity to monitor the success of these exotics, and to consider their suitability for wider introduction.

Knowles (pers. comm. 2002) believes the forest is fairly stable in terms of the health of the different tree species and in terms of their relative competitive success. He also suggests the forest's loamy soils may give it a greater buffer against a possible shift towards drier conditions than exists with the sandy soils of Spruce Woods. He does not see any great immediate vegetation challenges or threats, although he agrees with Armstrong and Hagen that the forest may be on the verge of a tent caterpillar outbreak.

The climate change outlook for Turtle Mountain in fact closely mirrors that for Moose Mountain, and the forest must be considered highly vulnerable. Like Moose Mountain, the site is elevated only a few hundred metres above surrounding prairie. Like Moose Mountain it lacks significant refugia, such as extensive north-facing slopes, steep coulees, or major rivers. As at Moose Mountain, it is probable that there will be lower water levels, making maintaining a sport fishery more difficult or impossible. In this context the removal of easements protecting waterfowl habitat would be particularly unwise.

However, compared with Moose Mountain the forest at Turtle Mountain is slightly more diverse in tree species, which may to a degree make the Turtle Mountain forest more resilient – oak expansion may to a degree compensate for aspen dieback, for example. The presence of pure bur oak forest on the dry exposed southwestern slopes of Turtle Mountain strongly suggests that oak will have the greatest chance of survival amongst the existing hardwoods as CMIs decline over time. The conclusion is that Turtle Mountain will evolve as previously described for Moose Mountain, except for the persistence of oak in savannah landscapes. The final CMI limits of bur oak are not known.

Curiously, it is not clear whether the provincial park management plan's 1985 (and current) objective of maintaining vegetation communities representative of the Turtle Mountain natural region will be violated by this outcome – it depends on the interpretation of "natural," and on whether "representative" is understood in static or dynamic terms. These issues are taken up in the chapters following our discussion of the individual island forests. But it is clear the management plan's objective of the preservation of "unaltered hardwood forest" is impossible under climate change. There will be impacts. Equally, climate change makes impossible the North Dakota Forest Service's objective of preserving "pre-settlement" conditions, as the climate is moving steadily away from its pre-Euro-American state. And we may reasonably assume, subject to consultation, that the great majority of stakeholders and the public would prefer not to trade the diversity of the current aspen-and-mixed-hardwood forest, plus wetlands, for an oak savannah landscape.

The International Peace Garden could provide an excellent institutional site for climate change research. It is directly adjacent to a protected forest of concern, but is not subject to the same legislative management constraints. One could here, for example, engage in test plantings and adaptation trials of native and exotic species. In fact, the health of some existing vegetation within the Peace Garden may be of monitoring interest. An international scientific program might be well received here and could also have a strong public profile. The site, with its high visitation numbers from both sides of the international boundary, would also make an excellent location for both interpretative displays on climate change impacts on forest systems and for social survey research on public attitudes and preferences for adapting to climate change in natural forest systems. At present there is a small museum and educational centre on site largely dedicated to wildlife management issues.

8. SPRUCE WOODS

(a) Site Environment and Landscape

The name "Spruce Woods" refers to a somewhat vaguely defined wooded area centred around part of the Assiniboine River Valley of southwestern Manitoba. Another common name for the same area, focusing on topography, rather than vegetation, is the Carberry Sandhills.

After the most recent glaciation extensive amounts of sand and finer sediment were deposited from rivers flowing from the west into proglacial Lake Agassiz. A vast freshwater delta resulted in the area of the modern-day sandhills landscape. As Agassiz shrank in size, and as the climate became warmer and drier, the delta became an open landscape of light, silty soils and sands, shaped by wind into a hilly dune landscape. Under slightly more humid conditions these dunes were colonised and stabilised by vegetation. Colonisation of more than 6475 km^2 of potentially shifting dunes has reduced the open dunes element of the sandhills landscape to less than 25 km^2, and perhaps to as little as 4 km^2 according to MNR (1998b). Portions of the dunes landscape have been colonised by trees. The first-time visitor is typically surprised by the visual oddity of trees, including white spruce, emerging from sand dunes.

Unlike our other study sites Spruce Woods is not a highland area and the existence of this isolated forest amidst grasslands cannot be explained by orographic precipitation or cooler highland temperatures. Groundwater is a key factor. Shallow aquifers are often within reach of the root systems of mature trees and of some non-arboreal species as well. Sandy, light-textured soils may be droughty in some years, limiting establishment of new trees to wet years, or to shaded sites, but once a tree's roots reach groundwater long-term survival becomes much more likely.

Wrigley (1974) states that two-thirds of the mammals found in the sandhills region are at or near the limit of their geographic distribution. The plains spadefoot toad, the northern prairie skink (Manitoba's only lizard) and the western hognose snake are examples of locally rare or disjunct herptiles dependent on the idiosyncratic sandhills and forest environment.

(b) Forest History and Nature

The balance between vegetated and non-vegetated dunes is a fine one. Many dunes contain buried dark soil horizons representing former vegetated dune surfaces (Wolfe et al. 2000). These paleosols suggest the area has alternated between open and vegetated states 4 or 5 times within the past 5000 years, and that much of the dune field has been active within the past 3000 years (Wolfe et al. 2000). Most interesting is Wolfe et al.'s (2000) finding that all the paleosols of the sandhills region have profiles typical of prairie – none of the buried soils display the Ae horizons typically found under boreal forest. This leads Wolfe et al. (2000) to conclude that the present boreal forest cover found on some parts of the sandhills is a recent phenomenon. They also note that the minimally developed surface soil profiles found in much of the sandhills area, historic observations of early travellers, and time-series photographic evidence, all suggest that much of the landscape is only recently stabilised. Consistent with this hypothesis Shay et al. (2000) note extensive vegetation colonisation of dunes in the Bald Head Hills portion of the sandhills based on aerial photographic evidence between 1948 and 1994.

An explanation for the persistence of boreal forest elements may be the presence of the Assiniboine River and Valley. Presumably, when the hypsithermal promoted the expansion of grassland northwards, and also during more recent Holocene dry periods, white and black spruce were able to persist as remnant boreal forest elements in the river valley proper or in some of the relatively steep-sloped coulees tributary to the Assiniboine. Epinette Creek Coulee is an example of one such well-sheltered and relatively humid environment. Alternatively, conifer seed may have been carried downstream from the upper Assiniboine watershed and recolonised the Spruce Woods region when conditions allowed. The Assiniboine Valley likely also served as a corridor for the westward spread of eastern deciduous species into the sandhills. In this case seed could not have been carried upstream, but the valley may have provided a viable habitat corridor for tree growth and slower overland migration westwards.

The modern sandhills ecosystem contains elements of grassland, eastern deciduous and northern coniferous biomes. Green ash, black ash, American elm, basswood, Manitoba maple, eastern cottonwood, balsam poplar and willows occupy the valley lowlands. Scattered white spruce, aspen and bur oak are found amidst mixed-grass prairie on the sandy, rolling, upland hills, with oak more prevalent on the drier sites. White birch is found in both lowland and upland sites. Tamarack and black spruce are found in boggy areas. While the forested parts of the sandhills are often loosely referred to as "spruce woods," after their signature tree species, aspen is actually the dominant tree in numbers and wood volume. Spruce is only locally dominant, which might explain why Wolfe et al. (2000) did not find evidence of boreal paleosols – even today a random sandhills-area soil profile would be relatively unlikely to happen to coincide with a dense spruce stand.

Early historical accounts support the theory that the sandhills have become, in general, more densely forested since European settlement, presumably because of the suppression of fires. For example, Alexander Henry (1897), a North West Company fur trader, commented upon passing by the former site of Pine Fort on the Assiniboine River in 1806: "Here we had an establishment for several years, but from the scarcity of wood, provisions, and other circumstances it was abandoned, and built higher up the river…The country hereabouts is very hilly and rough, with deep valleys, in which grow some épinettes [spruce] and stunted birches and poplars." However, it must be noted that prairie fur traders regularly consumed surprising amounts of local wood supplies around their forts for construction and for firewood and often had to move to a new fort site for fresh wood every few years (Morton 1941; Henderson 1996).

During his 1858 exploratory expedition Henry Hind (1860) crossed the sandhills, commenting: "We arrived at the Sandy Hills; they consist of sandy knolls covered with scrub oak and aspens…These rounded eminences have all the appearance of sand dunes covered with short grass and very stunted vegetation…The aspect of the country for many miles is that of a plain sloping gently to the east, and studded with innumerable mounds or hillocks of sand, thinly covered with a poor and scanty growth of grass; here and there small lakes or ponds occur fringed with rich verdure, but its general character is that of sterility. From the summit of an imposing sand-hill…the country lay mapped at our feet; as far as the eye could reach, sand-hills, north, east, and west, sometimes bare and ripple-marked, but generally covered with short grass, were exposed to view…The following morning we reached the "pines," for which we had been anxiously looking, but to our disappointment they proved to be nothing more than balsam spruce in scattered clumps."

In the early years of farm settlement Ernest Seton wrote extensively on the fauna of the sandhills, giving the impression of a relatively open savannah landscape. One of his books focused on tracking a magnificent mule deer stag. Mule deer prefer relatively open habitat and are no longer present in the area. Seton noted the changes brought about by settlement as early as 1892: "Aspen were springing up everywhere as prairie fires were controlled, and the white spruce in the sandhills south of Carberry had filled in the gaps caused by previous fires" (Houston 1980).

(c) Site Management and Policy (plantation era)

It is convenient to break the complex management history of the sandhills into plantation and post-plantation eras. In 1895 (MNR 1998b), in part as a response to failed agricultural settlement, a federal forest reserve was established. A planting program of native and non-native species was initiated around 1904. According to Jameson (1956), planting of about 712,000 seedlings was undertaken by the federal Department of the Interior from 1904-1929 in the following proportions: Scots pine 38%, white spruce 31%, jack pine 27%, lodgepole pine 4%, Norway spruce <1% and caragana <1%. Survival rates of the 1904-1926 conifer plantations in the year 1953 were estimated by Jameson (1956) as: jack pine 40%, lodgepole pine 35%, Scots pine 20%, white spruce 0% and Norway spruce 0%. Jameson (1956) believed drought and heat were the greatest cause of mortality, affecting white spruce, Norway spruce and Scots pine most of all. Winterkill also caused mortality in Scots pine and white spruce. Rabbits, pocket gophers and competition from grass damaged all species. Jack pine, he felt, exhibited the best growth rate and form.

In 1930 control of the greater part of the federal forest reserve passed to the provincial government and the reserve was redesignated as provincial forest. The pace of forestation increased. Haig (1957) reported on the 4.5 million seedlings planted by the Manitoba Department of Natural Resources from 1930-1946, noting the following species distributions: jack pine 74%, Scots pine 16%, lodgepole pine 8%, white spruce 1% and red pine 1%. A 1956 survey (Haig 1957) established the following survival rates: jack pine 62%, Scots pine 35%, lodgepole pine nearly 0%, white spruce 53% and red pine nearly 0%. Jack pine generally showed the greatest growth as measured by height or diameter. Haig (1957) concluded that jack pine should be favoured for reforestation, but that experimentation with other conifers should continue.

Walker (no date, but 1989 or later) investigated plantations originating between 1947 to 1979. Over this time about 5.7 million seedlings were planted in the following proportions: jack pine 58%, Scots pine 26%, white spruce 12% and red pine 1.5%. Small numbers of lodgepole pine, tamarack, (Colorado) blue spruce and Blackhill spruce were also planted and, from 1976, small numbers of native hardwoods.

Quantitative survival rates of these most recent conifer or hardwood stands are not known, but Walker (no date) makes a number of qualitative observations. Blue spruce and Blackhill spruce show poor survival. Jack pine and Scots pine plantations have both suffered from windthrow and ice damage from freezing rain, but jack pine seems significantly more susceptible to these hazards. Jack pine budworm has attacked jack pine, Scots pine and lodgepole pine. In general, Walker favours Scots pine over jack pine owing to the former's resilience in wind and to its greater wood volume production. However, he notes both species have proven difficult to regenerate in cut areas, which he attributes to heavy browsing by high white-tailed deer populations. Lodgepole pine mortality has been high and many plantations have been failures, but some individual specimens have been very successful. Lodgepole, Walker believes, should not be dismissed as being unsuited to the sandhills and he suggests controlled pollination of successful specimens could lead to a strain well adapted to the area. However, left alone, he expects existing lodgepole to cross-pollinate with jack pine. According to DNMR (1960), experiments with ponderosa pine are said to indicate that this species is not suited to the region, but no details are given.

A 1953 survey of the provincial forest (DNMR 1960) assessed wood volume percentages of those trees of at least 4 inches dbh as follows: white spruce 55.8%, black spruce 2.9%, tamarack 1.4%, aspen 28.9%, bur oak 6.2%, balsam poplar 3.3%, white birch 1.3%, white elm 0.1% and green ash 0.1%. Manitoba maple was present in insignificant amounts. As at Turtle Mountain, the survey parties found that most stands were uneven-aged and therefore could not be age-classed.

Manitoba Forest Inventory Reports (year 1977 data on file in Winnipeg) indicate species present on "productive forested lands" in the provincial forest as follows (in descending order of areal extent): aspen, tamarack, jack pine, white spruce, bur oak, ash, Scots pine, balsam poplar, Manitoba maple, black spruce, white birch, red pine (found on 2 plantations covering a total of about 12.5 ha within the provincial forest, but outside the provincial park) and basswood. In addition, as "non-productive forested lands," black spruce muskeg, tamarack muskeg, willow, and dwarf birch areas are noted. Knowles (pers. comm. 2002), formerly the responsible forester for the region, notes as a sandhills idiosyncrasy that while black spruce is present in theory, he has yet to see it. Often in wet lowland sites where one would expect to find black spruce one instead finds white spruce in association with tamarack. Zoltai (1975) also failed to find black spruce at Spruce Woods.

According to Knowles (pers. comm. 2002), a lodgepole plantation dating from the 1930s survived until about 1979, a year in which a combination of winter browning and an attack by jack pine budworm so damaged the stand it was harvested for wood. Lodgepole and Scots pine are susceptible to winter browning in the sandhills as the long freeze-up stresses their ability to maintain adequate moisture in their needles. Knowles notes that Scots and jack pine plantations from the years 1917-18 and the 1930s are doing well – some of these stands are reaching maturity, at which point they may be harvested. However, he notes there is little regeneration of Scots pine as porcupines attack the young trees. It is conceivable that porcupines have become more numerous and their impacts correspondingly more severe as the modern forest has expanded. Jack pine shows little natural regeneration, but would not anyway be expected to do so within a dense plantation stand. There is currently a spruce budworm outbreak and jack pine budworm may become a problem. From a commercial forestry perspective, dwarf mistletoe is another potential problem. Dutch Elm Disease is now present in the forest.

(d) Site Management and Policy (post-plantation era)

Spruce Woods Provincial Park was established in 1964. As at Turtle Mountain, the provincial park is entirely encompassed within the provincial forest (which also incorporates most of Canadian Forces Base Shilo immediately adjacent and to the west of the park, as well as some further disjunct forested areas to the north). In theory, Knowles (pers. comm. 2002) notes, commercial forestry remains an objective for the provincial forest. However, in practice, current policy does not allow commercial harvest or afforestation in either the provincial park or CFB Shilo, and there is little commercial wood outside these 2 blocks of the forest. The 269 km^2 provincial park is classified as a "natural park" within the Manitoba parks system (MNR 1998b). 75% of the park is zoned as "backcountry" and is by definition off-limits to commercial resource extraction. Part of the park's purpose is to preserve areas representative of the Assiniboine Delta Natural Region.

David (1977) noted a general trend to stabilisation and advancing vegetation colonisation of the relatively small – and highly valued – active dunes areas of the sandhills. Ward (1980) predicted the complete colonisation of the open dunes between 1987 and 1994. As a stopgap measure to maintain open dunes until the approval of a (never completed) master plan for the entire park, "Interim Management Guidelines" for the open dunes area of the park were approved (MNR 1982). These guidelines included the suggestion that selective and experimental de-vegetation programs should be investigated, including herbicides, prescribed fire and mechanical disturbance. In reality, the dunes did not become colonised as rapidly as was feared and no program of dunes de-vegetation has been instituted.

In their 1993 study of park habitats Higgs and Holland (1999) warned that several remnant reptile and plant species might soon be lost from disappearing open sandhill communities. They also concluded that willow, white spruce (via seedlings sheltered by parent trees) and especially aspen were invading grassland communities as a result of fire suppression. Expansion of leafy spurge, an aggressively invading exotic in grasslands, was also a problem. This is a local example of a widespread problem – as part of the U.S. climate change national assessment exercise, Joyce et al. (2001) note that the disappearance of the natural disturbance regimes of fire and grazing has expedited the progress of invasive exotics throughout the Great Plains. Higgs and Holland (1999) further identified some white spruce as dead or dying from dwarf mistletoe and concluded that habitat management, including prescribed burns, was necessary "to sustain and enhance the area's habitat diversity and integrity." They specifically recommended the removal of jack pine and Scots pine plantations on the grounds these trees did not represent natural ecosystems. Manitoba parks planner Helios Hernandez (pers. comm. 2002) would also be inclined to remove them for the same reason – this contrasts with Smith and de Smit's (1979) recommendation to retain the exotic white spruce plantations at Turtle Mountain for aesthetic and diversity reasons. Marr Consulting (1995) are in accordance with Higgs and Holland (1999). Within the context of a study of Manitoba's sandhill communities, Marr Consulting (1995) stated that a vegetation management plan for Spruce Woods Provincial Park should be prepared with the objective of maintaining a balance of successional stages within the park's vegetation communities.

Schykulski and Moore (2000) confirm that the sandhills have become increasingly encroached upon by forest expansion since European settlement and also suggest that the rate of colonisation may have increased within the current generation. Lack of fire is thought to be the cause. Ransom (1969a) suggests browsing and rubbing by bison may have restrained aspen encroachment in pre-European times in the sandhills area. Roe (1970) discusses the destructive impact of bison on trees throughout the Plains, while Nelson (1973) notes the earlier impact and importance of many other grazers besides bison on the shifting forest-grassland boundary.

Exceptionally, as a result of continued concern about habitat loss, Spruce Woods Provincial Park does have one vegetation management plan (Schykulski and Moore 2000) in place, albeit focused on a narrowly defined issue – the need to try and retain or restore prairie elements within a largely forested landscape. The plan's objective is to maintain open grassland and to maintain corridors between grasslands, where possible, to allow for the movement of grassland species. Typically this is done by controlled burns in early spring, normally while there is still some snow on the ground (Ken Schykulski, parks planner, pers. comm. 2002). Both herbicides and biological agents (i.e. non-native herbivorous insects) are also employed. A key sub-objective is the control of leafy spurge. Most recently a "hydro axe" mower has been used to control aspen encroachment. A management plan (MNR 1988) for the elk population in the wider sandhills area also exists. The objectives are both to manage elk numbers to fit the available habitat and to maintain elk habitat.

CFB Shilo provides an important management contrast to the provincial park. Shilo consists of about 404 km^2 of land, 15% owned by the Department of National Defence and 85% leased from Manitoba. The base was founded in 1910. The western portion of the then federal forest reserve was leased in 1915 and is still used today for intensive artillery and tank training. Whereas the adjacent park has become densely wooded since agricultural settlement and fire suppression, Shilo presents the strikingly different image of a largely open grassland landscape with occasional small groups or lone specimens of spruce. The difference can be entirely attributed to fire. The military site has been, and still is, the scene of frequent fires resulting from shelling and other training activities, and consequently unintentionally bears a greater resemblance to the pre-European-settlement landscape than does the provincial park (Dillon 1996). In other countries, such as Britain (Henderson 1992), military training ranges have also on occasion maintained or created valued habitat for particular species.

In contrast to the park, Shilo boasts a comprehensive vegetation management plan (Dillon 1996). The overall landscape-ecosystem objective is to "maintain the current percentages of the various [vegetation] cover types that currently exist." For practical purposes, 1988, a year for which an aerial photographic record of the site is available, is considered the base year from which deviations from vegetation patterns are measured. Specific management sub-objectives include the control of leafy spurge, brome grass and Kentucky bluegrass. Prescribed burns are used to control aspen encroachment. The oldest stands of white spruce are protected against fire where possible, with the intention of allowing these stands to regenerate by natural propagation. The planting of nursery seedlings grown from spruce cones collected on the base is envisioned as possible if the stands do not naturally regenerate successfully. Another sub-objective is to maximise habitat for ungulates. Shilo's management plan acknowledges the critical impact of climate on the landscape, but does not address climate change as an issue.

(e) Management Analysis and Vegetation Outlook

It is possible that the current ongoing loss of open dunes is a natural evolution of the landscape. There are, after all, buried paleosols present, i.e. the sandhills have evolved from active, open dunes to stabilised, vegetated systems several times in the past few millennia. However, it is more likely that anthropogenic suppression of fire has encouraged dune stabilisation. The loss of a major source area of sand to grass in the Shilo base immediately to the west of the provincial park has removed a source of fresh dune material and possibly encouraged vegetation colonisation of the parks' open dunes.

Whether or not current dune stabilisation is a largely natural process or not, it is clear that public and scientific interest would favour the preservation of some active dunescapes. Therefore, if increasingly xeric conditions acted to stop or reverse colonisation of the few remaining active dune areas in the sandhills, the impact might actually be welcomed, as it would preserve landscape diversity and interest. However, a significant time lag between a period of drought and any re-activation of stabilised sand dunes is to be expected (Thorpe et al. 2001). In the interim management could relatively easily keep the dunes open without the aid of climate change, should we wish to act. While prescribed fire might not be effective if the vegetation is too thin (as it is even in some areas of grassland), herbicides could certainly do the job, although this might not appear an aesthetic option.

Similarly, more xeric conditions may prove advantageous in aiding retention or expansion of grasslands within the provincial park. But from the point of view of tree species, climate change poses major challenges. While Knowles (pers. comm. 2002) has not noted recent changes in tree vigour, he notes that aspen is already normally under stress and short-lived on most sites within the sandhills. Under the significantly drier conditions foreseen in our climate scenarios, aspen could be expected to die back. This could impact in turn on spruce regeneration, since in the drier, more open, areas spruce may get its start under aspen canopy. As conditions become steadily more xeric, trees will very likely persist in the sheltered coulees and Assiniboine Valley slopes, as probably occurred in previous Holocene dry periods. However, the long-term ability of trees to persist on the much greater sandhills landscape beyond the valley and its tributaries is doubtful. Aside from favoured slope or riparian sites, trees can be expected to decline as the CMI for Spruce Woods (at -2.3 for the highest elevation in the 1961-1990 climate normals) declines over time. However, the rate of predicted CMI decline varies substantially between the models: the CGC (warm-dry) GCM and the CSIRO (mid-range) GCM scenarios both suggest the area will be approaching the limit of aspen viability by the 2020s, while the Hadley (cool-wet) GCM scenario suggests this will not occur until the 2080s. Mature white spruce would likely survive at least until the CMI declines to around -30, but regeneration will have ceased much earlier, and is likely already becoming difficult. As at Turtle Mountain, bur oak may prosper in some areas as aspen declines, forming an increasingly open savannah landscape.

The presence of shallow aquifers is a key complicating factor at Spruce Woods. Thorpe et al. (2001) correctly note that climate change impacts on groundwater are difficult to predict and subject to a number of variables. For example, in theory an increase in winter precipitation and snowmelt might allow for increased groundwater recharge before summer heating. The CMI model employed in this study provides sound generalised landscape scenarios, but does not provide analysis of site-specific soil and subsurface water complexities. Nonetheless, it seems highly probable that a long-term shift to significantly drier conditions will be reflected in local groundwater levels, although the impacts may be very site specific.

9. CYPRESS HILLS

(a) Site Environment and Landscape

The Cypress Hills, located in southeastern Alberta and southwestern Saskatchewan, are a dissected sedimentary-origin plateau that rises up to 600 metres above the surrounding grasslands and extends about 130 km west to east and 24 to 40 km north to south (SPRC, no date). The plateau is broken, in descending order of altitude, into West, Centre and East Blocks, each of which is surrounded by lower-elevation grasslands. The rise on the northern and western sides of the plateau is steep, while to the south and east the hills slope away more gently to the surrounding plains. The West and Centre Blocks, upon which this study focuses, are each sites of a disjunct mixed-wood forest and disjunct fescue grasslands. The East Block has extensive areas of aspen-dominated hardwood forest along its north slope and within sheltered coulees. Maximum plateau elevation, at the extreme western edge of the West Block, is 1465 metres.

Brown chernozem soils typical of southwestern Saskatchewan grasslands are found in grassland areas below the tree line and dark brown chernozems are found under the fescue grasslands atop the plateau. The forested slope soils are typically dark grey luvisols. On a few high-elevation sites there are black soils, typically under aspen or under aspen and grass (Harris 1984). The uplands are generally well drained, with some imperfectly drained areas. Groundwater flows from the highlands to springs on the adjacent lowlands are common (SERM 1996). May and June precipitation amounts to about 65% of the annual total. Thunderstorms occur and pose a significant fire risk (SERM 1996). In the last several years the region has been very dry, although heavy late-spring rain and snow and summer rain in 2002 have brought a respite from drought.

The hills are subject to chinook winds, dry westerlies or south-westerlies which blow in during the winter and raise temperatures well above the freezing point. Chinooks may put moisture stress on conifers by greatly increasing evapotranspiration through their needles at a time when water uptake through their root system is impossible owing to frozen soil.

(b) Forest History and Nature

An unusually good paleoclimate record exists for the Cypress Hills (Sauchyn 1997). An aspen forest – grassland complex with very little coniferous forest elements was established on the West Block by 9000 BP. A period of maximum aridity likely occurred around 7700 to 6800 BP, resulting in a vegetation cover dominated by grassland taxa, intermediate amounts of *Populus* and low levels of *Betula*. There were very few conifers at this time; the currently extant white spruce and lodgepole pine were presumably only able to survive in a few isolated refugia (Sauchyn and Sauchyn 1991). As the climate cooled and moisture levels increased, spruce and pine became more abundant, becoming prevalent after about 4600 BP. The modern vegetation pattern was essentially established by about 3230 BP.

It is thought the hills were not as densely wooded prior to European settlement as they have become subsequent to systematic wildfire suppression. Macoun visited the Cypress Hills in 1880 and noted wood as abundant in all the hills' coulees. He was also impressed by an unusual vegetation landscape: "As we proceeded westward over the plateau, it became more elevated and other species began to take prominence, notably *Lupinus argentea* [silvery lupine] and *Potentilla fruticosa* [shrubby cinquefoil] covered miles of country to the exclusion of other species, and as both grew about eighteen inches in height, and had a bushy habit, the whole country, for a day's travel, was either blue or yellow or both, as either species prevailed or were intermixed. In all my wanderings I never saw any spot equal in beauty to the central plateau of the Cypress Hills" (Macoun 1882). This exact landscape no longer exists in the hills, although there are areas of the Saskatchewan West Block dominated by cinquefoil. The reason for cinquefoil's modern presence in high densities in some areas is not known with certainty, although these areas are also thought to be heavily grazed (Kelvin Kelly, Park Management Specialist, pers. comm. 2002).

In 1885 – Strauss (2001) notes that some sources erroneously state "1886" – a major fire swept through the Cypress Hills. Almost the entire forested area of the hills and 95% of the lodgepole pine forest in the Centre Block were burnt (SPRC, no date; SERM 1996). Another fire ran through the hills in 1889 (Strauss 2001). As a result of fires and intensive logging, by 1893 only a few isolated trees or groves persisted in sheltered locations (Scace 1972). There have been 4 smaller 20th-century fires in the Centre Block (SERM 1996) and several in the West Block (Strauss 2001).

The Cypress Hills' current vegetation can be broken down into 6 major associations: lodgepole pine forest, white spruce forest, aspen forest, fescue grassland, mixed-grass prairie and wetlands. Aspen is the commonest tree, while lodgepole is dominant in terms of total wood volume (Harris 1994). Cooler, north-facing slopes are the most heavily wooded. The lodgepole pine association is found in drier areas above about 1280 metres, typically on steep, well-drained slopes. The pine grow close together, creating heavy shade and allowing little undergrowth. White spruce is predominantly found in moister areas such as north-facing slopes, on bottomlands, or near springs. It is often found together with aspen and frequently supports rich understory vegetation. Aspen woodland is found just below lodgepole in pure stands along the north slopes, in bluffs amidst grassland atop the plateau, and along springs and creeks amidst the lower elevation mixed-grass prairie. Balsam poplar, Manitoba maple or eastern cottonwood may also be found in association with aspen, which typically supports a thick understory. A single small stand of white birch is present on the north slope of the Alberta West Block according to the responsible forester, Les Weekes (pers. comm. 2002). Greene mountain-ash is present in at least the Saskatchewan West Block (Kelly, pers. comm. 2002; Breitung 1954). Spring-fed marshes and wetlands support fringing willows, balsam poplar, birch, or white spruce (SPRC, no date). Weekes (pers. comm. 2002) notes that there is possibly a population of Engelmann spruce in the forest and that the forest's "lodgepole" pine may actually contain a genetic admixture of 10% jack pine.

The disjunct fescue prairie association which extends over much of the top of the plateau is an important ecosystem in the Hills. Like the forest associations, the fescue grassland is dependent on moister conditions than prevail on the surrounding plains. Rough fescue is a major species component. Other important grasses include bluebunch fescue, wild oat grass, awned wheatgrass, June grass, western wheatgrass, northern wheatgrass and mat muhly. Shrubby cinquefoil is the most characteristic shrub. Western snowberry is invasive in the lowest elevations of the fescue grasslands, cinquefoil in the higher elevations (SERM 1996). In the Saskatchewan portion of the hills Kelly (pers. comm. 2002) notes the loss of up to 800 ha of fescue grassland to forest encroachment by pine and especially by aspen since 1945, as shown by a 1999 GIS study of forest boundary changes. This loss is thought to be ongoing.

Some Cypress Hills vegetation (such as heart-leaved arnica, thimbleberry and lodgepole pine) parallels that found in the western cordillera and in the Sweet Grass Hills (Marquis and Voss 1981). Species representative of cordilleran, arctic subalpine, shortgrass prairie, mixed-grass prairie and boreal flora are also present (SERM 1996). Some rare plants are dependent on the hills elevation and lodgepole canopy, and the hills are known for their various orchids (SERM 1996). Harris and Lamont (1978) and Harris (1984) detail the common understory vegetation associations within the Saskatchewan portion of the forest.

The forests support isolated populations of elk and moose. Pine marten has been reintroduced in the West Block. Disjunct reptile (red-sided garter snake) and amphibian (tiger salamander) populations exist (Stewart and Lindsey 1983), as do isolated insect populations (Lehmkuhl 1980).

On the Alberta side (sometimes known as the Elkwater Block) of the West Block, Peterson and Peterson (1992b) noted the following major shifts in vegetation in recent decades: an expansion of pine and spruce onto formerly untreed plateau fescue grassland; an expansion of spruce into valley grasslands at the lower altitudinal limits of tree cover; an overall increase of spruce underneath pine and aspen overstories; and some thinning of dense lodgepole stands which date from fires of 1919 and 1934. They also noted abundant evidence of lodgepole regeneration (on exposed mineral soil) in the absence of fire. They did not find logging, dwarf mistletoe, insects or disease to have been major factors in recent vegetation trends. According to Weekes (pers. comm. 2002) pine and spruce continue to expand on the Alberta plateau and, as in Saskatchewan, spruce is developing under pine stands. A slow conversion of some lodgepole stands to spruce has also been noted in the Saskatchewan forest (SERM 1996). Most recently Weekes (pers. comm. 2002) notes that in the Alberta West Block forest aspen is being lost. Both clones on the plateau and individual trees on the north slope are breaking up, with poor regeneration. The reasons for this contrast with Saskatchewan experience are not clear.

It is widely accepted that the suppression of fire over many decades has resulted in stands of dense mature forest that present a real danger of catastrophic fire (SERM 1996). Gary Neil (pers. comm. 2002), formerly park ranger in the Saskatchewan West Block, notes that the fire season has expanded greatly in the Cypress Hills. In dry periods the fire threat can be severe.

(c) Site Management and Policy

In 1902 the federal government created a 47 km^2 forest reserve in the Cypress Hills, which was expanded to 3 discontiguous blocks (West, Centre and East) totalling 492 km^2 in 1911 (SERM 1996). In 1931 these lands were transferred to provincial control. The Centre Block was designated a Saskatchewan provincial park in 1931, as was the Alberta portion of West Block in 1951 and the Saskatchewan portion of West Block in 1976. Two small artificial recreational lakes (Lochs Lomond and Leven) were created at Centre Block and are dependent on groundwater pumping (SERM 1996). The East Block has been primarily leased as agricultural grazing land. The total Saskatchewan provincial park area is about 185 km^2, with both West and Centre Blocks classified as “natural environment park.” Notwithstanding, management recently introduced an exotic – wild turkey – in the hills. The Alberta and Saskatchewan provincial parks were jointly designated an interprovincial park in 1989. Part of the intent of this designation is the coordination of resource management. Fort Walsh, a small national historic park, adjoins the southern boundary of the Saskatchewan West Block. Haying and cattle grazing are longstanding activities almost throughout the interprovincial park. It is argued these ranching activities are helpful in controlling fire risk and in reducing shrubby cinquefoil and aspen encroachment onto the grasslands.

A 1985 Vegetation Management Plan for the Saskatchewan forest initiated in response to an outbreak of mountain pine beetle adopted as a goal “the diversification of the age structure of the existing forest, in an attempt to alleviate the current problem of an entire forest reaching maturity and dying at the same time” (SERM 1996). In the Saskatchewan West Block “sanitation cuts” have been undertaken to control a mountain pine beetle infestation. Small spruce and lodgepole (no higher than 14 feet in height) may be taken from designated areas for Christmas trees. Limited removals of trees for transplanting are allowed from road rights-of-way and from under power lines (SERM 1996). Harris (1984) noted the existence of jack pine plantations in both Centre and Saskatchewan West Blocks and recommended they be cut down to avoid contamination of the native lodgepole genetic stock. All jack pine has now been removed (Kelly, pers. comm. 2002). Shrubby cinquefoil has been mowed to maintain fescue habitat (SERM 1996).

In their study of range management within the Saskatchewan Cypress Hills Godwin and Thorpe (1994) argued, with respect to grassland, that a good goal is to manage for a diversity of natural vegetation types with a variety of successional stages, including heavily grazed grassland and completely ungrazed grassland. Uniformly excellent range condition, which in the hills equates to a high proportion of rough fescue, would not, they noted, necessarily be the natural state of the vegetation prior to European settlement. They also argued strongly for the suppression of exotics and were concerned about the possible expansion of Kentucky bluegrass.

Kelly (pers. comm. 2002), the current responsible forester, highlights the main management concerns in the Saskatchewan forest as follows:

- there is a lack of forest regeneration (and therefore poor age-class diversity);
- the forest is vulnerable to insect and disease attack;

- there is increasing wood rot and forest decadence;
- the fuel load is dangerously high; and,
- fescue grassland is being lost to encroachment by lodgepole and aspen.

A forest management plan for Centre Block was approved in 1997 to deal with the first 4 issues. (The second, third and fourth issues derive directly from the first.) The objective is to regenerate up to 50% of the lodgepole pine forest outside of sensitive areas (i.e. outside of cottage areas, roadsides, riparian areas, and steep slopes) to create a diversified age-class forest (year 1997 update to SERM 1996). Currently, irregular-shaped clearcuts up to 0.75 ha in size are being harvested, scarified and regenerated through either natural or planted seedlings from local seed stock. These patch cuts are designed to regenerate the mature forest over a 50-year rotation (Kelly, pers. comm. 2002).

To reduce fuel load build-up Saskatchewan management allows (under permit) the salvage of dead wood. This policy has been successful, judging by a year 2000 report that: "over the past two years, the supply of standing dead, pine fuelwood has been greatly reduced, necessitating a change to fuelwood clear-cut harvest of standing green aspen. This also serves the dual purpose of reducing encroachment of Aspen on to the endangered prairie" (year 2000 update to SERM 1996). Lodgepole is also thinned by allowing some own-use cutting for rails. This policy is described as successful. Own-use cutting of poplar is allowed in the West Block for fence posts (SERM 1996).

Forest activities during the 1980s to control a mountain pine beetle infestation resulted in the removal of approximately 400 ha of West Block forest. A harvest moratorium was then instituted. Work on a forest management plan for the Saskatchewan West Block continues and will be a component of the overall ecosystem management plan for the Cypress Hills (Kelly, pers. comm. 2002). At both Centre and West Blocks the management objectives for aspen are unclear (Kelly, pers. comm. 2002).

On the Alberta (Elkwater) side of West Block, management efforts were expanded beyond fire control to include control of mountain pine beetle in 1980 and control of dwarf mistletoe in 1985. In 1987 commercial logging returned to Elkwater for a 5-year trial of limited clearcuts to reduce fuel loads and remove pine under threat from disease or insects (Dickinson 1992). As in Saskatchewan, the experiment was not an entirely happy one. Insensitive harvest practices on high-slope terrain and subsequent erosion and poor re-vegetation led to the current forest harvest moratorium in Alberta (Weekes, pers. comm. 2002).

As in Saskatchewan, tree encroachment onto grassland is a major concern. There have been controlled grass burns to restore grassland, but no burns of standing trees (Weekes, pers. comm. 2002). Grassland burns can successfully control cinquefoil (SERM 1996). According to Kelly (pers. comm. 2002), Saskatchewan managers are currently discussing the implementation of controlled burns to control exotic species and aid forest renewal.

Both provinces are now trying to develop a joint ecosystem management strategy for the hills that would encompass both the fescue grasslands and the montane forest, as well as wetlands. It is recognised that any agreed strategy is likely to contain controversial elements.

(d) Management Analysis and Vegetation Outlook

Harris (1984) states that no lodgepole exists in the East Block, but that charred pine stumps can still be found there. He hypothesises that the fire that destroyed these trees was so intense it destroyed their seed as well. It seems unlikely that all seed would have been destroyed, but more probable that subsequent dry years discouraged regeneration of surviving seeds, as the lower-elevation East Block is warmer and drier than Centre and West Blocks. Breitung (1954) in fact noted "one or two isolated individuals of lodgepole pine" on the East Block. If lodgepole can disappear from the East Block, owing to fire or drought or both, it is natural to speculate that it might equally disappear from West or Centre Block as climate change progresses. However, it is unclear whether

East Block was ever heavily wooded with lodgepole in the manner of West and Centre Blocks now. It is conceivable lodgepole never really declined much on the East Block, as it was possibly never widely established. Spruce, “fine groves” of which Macoun (1882) observed in 1880 a few kilometres north of the modern town of Eastend, still exist in sheltered coulees in the East Block today (pers. obs. 2002).

On West and Centre Blocks there is abundant evidence of lodgepole expansion onto grassland in the absence of fire (Peterson and Peterson 1992b). Harris (1994) notes that lodgepole cones will open simultaneously and completely under adequate fire conditions, but that alternatively seeds will also be slowly released with the ageing of cones. However, lodgepole will not regenerate in shaded conditions so, over time, in the absence of fire or other major disturbance, white spruce could well tend to replace lodgepole, as observed at many sites by Peterson and Peterson (1992b) and SERM (1996). However, this succession is slow and can only occur under conditions moist enough to support white spruce.

While the current lodgepole stands certainly appear mature to the commercial forester’s eye, Peterson and Peterson (1992b) believe it is incorrect to assume that the stands are overmature simply because they are over 100 years old. In some areas of North America, they note, lodgepole survives well for 300 to 400 years, and only at this advanced age has achieved true old-growth status. However, such long-term survival can only be achieved if environmental conditions are especially favourable to lodgepole. The Cypress Hills are likely not ideal lodgepole habitat even under the current climate and are becoming steadily less hospitable as climate change progresses. Strauss (2001) found 10% to 30% of lodgepole pine and white spruce in the hills suffered bole rot or red stain. Kelly (pers. comm. 2002) notes that recent cuts of lodgepole in Centre Block indicate that wood rot percentages as high as 30% are not unusual. Together with evidence of increasingly frequent windfall, this suggests the Cypress Hills lodgepole stands are at or past maturity.

By the 2050s all 3 GCM scenarios indicate CMIs in the negative 20s for the Cypress Hills at maximum elevation and drier still at lower elevations. In addition, high wind speeds, and in particular chinooks, likely make growing conditions for trees even more stressful than would appear from CMIs alone. On the other hand, there are extensive north slopes and areas of dissected terrain in the hills which may generate site-specific CMIs up to 20 cm higher than are indicated for level terrain. These refugia may ensure that declining CMIs do not cause complete extirpation of extant tree species. But by the 2050s natural regeneration of lodgepole or spruce certainly, and probably of aspen as well, is unlikely to be possible outside of very localised sites. The future landscape is likely to be one of small patches of stressed woodland persisting only in the most favourable sheltered sites. By the 2080s, the scenarios for the mid-range and warm-dry GCMs are so xeric that it is conceivable that there will be no regeneration of spruce or lodgepole anywhere in the hills, particularly as wetlands in the hills depend on local precipitation.

It is possible that under even current climate conditions forest encroachment onto grassland may be ending. Aspen, at least, is already noted as being under stress in the Alberta West Block. This may be because the Alberta forest has begun to suffer from drier and warmer conditions a little more severely, or ahead of, the Saskatchewan forest. Our CMI data show that, at their respective maximum elevations, the West Block has a slightly lower CMI in the 1961-1990 climate normals than the Centre Block, a difference that persists and increases slightly in the 2020 scenarios of all 3 GCMs.

Another climate change factor will be the increasing vulnerability of lodgepole stands to mountain pine beetle attack. It is widely believed that periods of very cold winter weather act as an effective control on mountain pine beetle outbreaks. Some GCMs suggest that while average annual temperatures will rise on the Plains overall, the effect will be disproportionately greater in winter. The GCM scenarios we constructed are ambiguous on this question (see Table 3). In any case, periods of sanitizing cold will become much less frequent. It is probable we are already experiencing warmer winters than in earlier decades. Spruce budworm attack is also possible.

Fire is the risk uppermost in most people's minds. Strauss (2001) predicts the next forest fire that escapes control will be catastrophic in impact as a result of previous fire suppression. Such a fire would reduce the recreational value of the hills dramatically and would be a disaster in terms of public perception. Sudden removal of the forest could also be expected to have an effect on wetlands and water flows in the hills. But would such a fire be an ecological disaster? This depends partly on one's view of nature conservation and the correct landscape to target, issues we discuss subsequent to consideration of the 5 island forest study sites. But certainly any resulting post-fire landscape could not legitimately be called "natural" for at least 2 reasons. First, the current heavy fuel load in the forest is the result of long-term fire suppression and is unlikely to have ever existed naturally in the recent past. Therefore, the hot, all-consuming fire it may support would be an anthropogenic (i.e. cultural) event in origin. Second, regeneration of the post-burn forest, grasslands and wetlands in the hills would have to occur under a warmer, effectively drier, emissions-driven climate.

It is possible that unassisted post-burn forest regeneration would be slow and patchy, even under current climate conditions, as these are already more xeric than those under which the existing forest developed. The future climate will be drier still, with regeneration becoming steadily more challenging.

10. SWEET GRASS HILLS

(a) Site Environment and Landscape

The Sweet Grass Hills lie about 90 km southwest of the Cypress Hills and about 140 km east of the Rocky Mountains cordillera. They consist of 3 separate butte complexes (from west to east: West, Middle and East Buttes) which rise about 900 metres above the surrounding plain. The buttes are laccoliths (uplifts resulting from igneous intrusions). Reaching above 2100 metres, West and East Buttes are considerably higher than the Cypress Hills. Middle Butte (also known as Gold Butte) is 1985 metres in height, is about midway between West and East Buttes, and lies about 15 km from either. Owing to parent material of igneous, metamorphic and sedimentary origin, and to the variety of slopes, aspects, altitudes and vegetative histories, the soils on the buttes are complex. Typically they are well drained with a high proportion of cobbles and stones in some soils. In general the buttes are steeply sloped (averaging between 40% and 70%). There are areas of scree and talus virtually free of vegetation and areas vulnerable to erosion owing to steep slopes (BLM 1996). All 3 buttes are dissected by steep ravines.

Domestic and stock water supplies are generally drawn from shallow aquifers in the areas surrounding the hills. These aquifers are dependent upon recharge flows from the hills (BLM 1996). Around the buttes there are numerous small oil and gas fields and the entire area is classified as being of high potential for oil and gas occurrence (BLM 1996).

(b) Forest History and Nature

The Sweet Grass forests are broken into 3 discontiguous blocks centred on each of the 3 buttes and separated by grasslands. The total coniferous forested area is less than 20 km^2, about half of which is found on West Butte and half on East Butte. Middle Butte has only a very small forested area, perhaps less than 1 km^2 in extent. Compared to the Cypress Hills, the Sweet Grass forests contain a markedly greater variety of conifers: Douglas-fir, subalpine fir, limber pine, whitebark pine, lodgepole pine, hybrid spruce (*Picea glauca* X *engelmannii*) and Rocky Mountain juniper (Thompson and Kuijt 1976). Strong (1978) also reported Colorado spruce as present, but was not certain as to its genetic purity. Elk, mule deer and white-tailed deer are found in the hills (BLM 1996). The dwarf shrew, the northern water shrew, and the heather vole survive, among others, as disjunct species (Thompson and Kuijt 1976; Pielou 1991).

As the hills have never been considered as potential timber sources of any significance, the nature and health of the forest has not historically been regarded as of great importance. Information on fire history, on possible vegetation change over the period of Euro-American settlement, and on management history and objectives, is consequently not readily available. The British (Dawson 1875) and American (Chickering 1878) international boundary surveys provide only a few basic comments on the hills' vegetation at that time. The main source of available data is Thompson and Kuijt's (1976) detailed (though not comprehensive) vegetation survey. It is, fortunately, a credible piece of work.

The following is a summary of Thompson and Kuijt's (1976) findings as they apply to tree species on West and East Buttes. Middle Butte does not seem to have formed part of their investigations. Generally speaking, forested areas on West and East Buttes are on north-facing slopes or in sheltered canyons. Fescue grassland associations, similar to those of Cypress Hills, typically dominate the drier, south-facing slopes. A few summit areas of the buttes are also under tundra-like sub-alpine grassland, rather than trees. This is thought to be a result of severe winds, cold and desiccation.

There are scattered stands of Manitoba maple in riparian areas below the coniferous forests. Eastern cottonwood and balsam poplar are also found near streams below coniferous forests and, when sheltered by steep canyons, at elevations as high as 1500 metres. Aspen woodland covers a larger area of West Butte than of East Butte.

Surrounded by grassland, aspen is found in isolated bluffs as high as 1800 metres on south-facing slopes and, together with Rocky Mountain maple, in the steep rocky ravines found between about 1500 and 2000 metres. At elevations between 1400 and 1600 metres aspen can be found within Douglas-fir forest.

Rocky Mountain juniper is found on the lower rubble slopes of West Butte, sometimes hybridised with creeping juniper. Douglas-fir, the lowest elevation conifer, is found as far down as about 1400 metres on north-facing slopes and is common up to 1900 metres. Douglas-fir forest dominates the lowest coniferous woodlands. It sometimes includes scattered hybrid spruce along streams. There is a rich understory. Douglas-fir also forms a savannah landscape on some drier slopes and is associated with limber pine on the lower south slopes of West Butte. Limber pine is found in forest edges throughout the hills, on exposed ridges, and in pure stands on some dry ridges of East Butte. Hybrid spruce, entirely absent from West Butte, is common and widespread between 1500 metres and the summit of East Butte. White spruce characteristics predominate at lower elevations and Engelmann spruce characteristics predominate at higher elevations. Strong's (1978) reports of Colorado spruce place it on north-facing slopes on East Butte between 1720 and 1810 metres, at the lower limits of the hybrid spruce association, on the north-north-east side of Mt. Brown and within the valley of Iron Creek.

Dense lodgepole forest is found on north-facing slopes from 1500 to 2100 metres and may date from fires of 1889. Lodgepole is typically bordered by limber pine at high elevation ridge tops and by spruce (on East Butte) in drainage bottoms. There is little understory vegetation. A mixed spruce-lodgepole forest is found on East Butte on north-facing slopes from 1600 to 2100 metres.

On steep north-facing slopes from 1650 to 1800 metres subalpine fir and spruce (on East Butte) dominate, with interspersed lodgepole pine and Douglas-fir. It is hypothesised that this unusually low-altitude occurrence of subalpine fir is due to cold air drainage. The highest elevations of East Butte have whitebark pine as a major component. Lodgepole pine, limber pine, spruce, and subalpine fir can also be found here. Whitebark and lodgepole pines dominate the upper timberline on West Butte, with subalpine fir on this butte found only on a dry rocky ridge south of the summit. Tree growth near the summits is short and stunted by high winds, and resembles krummholz.

Although, as noted, there is no systematic study of the forest's evolution over time, the general consensus is that at least some parts of the forest are now denser than prior to European settlement, presumably as a result of fire suppression.

(c) Site Management and Policy

Middle and East Buttes experienced placer mining for gold, silver, iron and fluorite in the early years after Euro-American settlement, while coal was mined at West Butte until World War II (BLM 1996). However, the primary historic economic use of the hills area has been for cattle grazing. The buttes are partly under BLM administration and partly under state of Montana ownership, while some areas are freehold. The BLM administers a total of 7,717 acres (3,125 ha) in the hills (BLM 1996), including the greater part of forested areas. Some significant areas of forest are on state lands (administered by the Department of Natural Resources and Conservation) on East Butte and on freehold lands on West Butte. All summits are either BLM or state lands.

The hills are high-value recreational lands, but as they are relatively small and remote they have never been major tourist destinations as they lack lakes, good road access and nearby major population centres. Until the last 20 years or so there have been relatively few resource conflicts of the kind that sometimes lead to detailed scientific study and the development of management plans. In the 1980s and 1990s controversy emerged over minerals exploration and possible mining operations in the hills. There was also growing concern over questions of access, particularly around the use of off-road vehicles. A long study and consultation exercise ensued (BLM 1988; BLM 1992), culminating in a "final" management document in 1996 (BLM 1996). Key management elements of the final decision include the following:

- most (i.e. 7,580 acres, or 3,070 ha) BLM-administered lands in the Sweet Grass Hills are designated an Area of Critical Environmental Concern (ACEC);
- the ACEC is closed to surface minerals exploration and development beyond that undertaken by pre-existing mineral rights holders;
- the ACEC is closed to off-road motorised vehicle use except under special circumstances;
- the protection of high-value potential habitat for the reintroduction of peregrine falcons, the protection of areas of religious importance to Native Americans, and the protection of elk and deer habitat are key objectives;
- activities, such as hunting, grazing and recreation are supported so long as they do not conflict with the key objectives;
- Native Americans will be consulted on management decisions affecting their traditional interests; and,
- aquifers and watershed flows are to be protected.

In recent years mineral interest in the Sweet Grass area has declined somewhat and the management situation is reasonably stable. Lou Hagener (pers. comm. 2002), BLM ecosystem specialist, notes that subsequent to the 1996 decision the BLM extinguished any potentially active surface mineral leases and there is presently no surface minerals activity in the hills. Grazing on forested BLM land is allowed at densities calculated according to available forage. Proposals for some timber use have been refused by the BLM (Stanley Jaynes, BLM field manager, pers. comm. 2002). The timber is not considered marketable, although there was some historic use for fence poles and some people take trees for Christmas (Dawn Wickum, USDA district conservationist, pers. comm. 2002).

The relevant local fire department is responsible for fighting fires on state lands and typically also responds to fires on BLM lands – the BLM is required to ensure appropriate fire suppression on lands under its administration (Casey Kellogg, Montana Department of Natural Resources and Conservation, district manager, pers. comm. 2002). There has not been a modern history of serious forest fires in the hills (BLM 1988), however fuel load build-up, particularly in thick lodgepole stands and also in some spruce stands, means wildfire is a concern (Richard Hopkins, BLM special assistant, pers. comm. 2002). Until heavy rains in June 2002 the hills had been in a 4-year drought.

On the state-owned forested lands, as the forest is not viewed as commercial timber, lands are leased for grazing, with allowable stockage rates reduced dependent upon the extent of forest cover. Elsewhere in Montana, where lands managed by the state Department of Natural Resources and Conservation contain commercial forests, these forests are managed both to maximise long-term state revenues and to promote biodiversity via the maintenance of diverse-age stands (MDNRC 1996). Although not a forest owner in the hills, the Montana Fish, Wildlife and Parks Department has a strong interest in the health of the forest as it provides elk and deer habitat (Gary Olson, MFWP, field biologist, pers. comm. 2002).

There does not seem to be any coordination of BLM and state management of the forested portions of the buttes. For example, Mount Brown, a significantly forested part of East Butte, is state-owned and almost entirely enclosed within BLM lands that are designated as part of the Area of Critical Environmental Concern. However, Mount Brown is managed independently of BLM objectives.

(d) Management Analysis and Vegetation Outlook

Climate conditions during the hypsithermal were likely xeric enough to drive off tree species from the comparatively lower elevation Cypress Hills that survived either at the comparatively higher Sweet Grass summits or, more likely, in the high-elevation canyon refugia of East and West Buttes (Pielou 1991). Intermittent streams have cut steep canyons and gulches in the buttes which increase habitat variety and provide some wetter areas which shelter many trees. These canyons may also explain why spruce is now found only on East Butte, which may have offered slightly more humid refugia than West Butte during the hypsithermal. Specifically, the cool, moist canyon of Ribbon Gulch, which currently provides habitat for subalpine fir, may be the most humid habitat site in all the Sweet Grass Hills, and have thereby favoured the survival of spruce.

As regards non-extant, but possibly viable, species in the Sweet Grass forests, Thompson and Kuijt (1976) suggest that after glacial retreat jack pine expanded from eastern North America through established white spruce forest, but did not reach either the Cypress or Sweet Grass Hills before these were isolated by grasslands. They also hypothesise that ponderosa pine remains excluded from the buttes because of cold intolerance (and presumably because of the colder conditions prevailing at the Sweet Grass Hills prior to their isolation by grasslands).

Interestingly, Thompson and Kuijt (1976) suggest that subalpine fir and whitebark pine may now be on the point of disappearing from the Sweet Grass Hills. They note that (with the exception of subalpine fir, which can be found as far down as 1650 metres in Ribbon Gulch) these trees generally persist only in small numbers around the summits and do not appear to be reproducing successfully. Sub-alpine understory plants normally associated with subalpine fir are absent, which may suggest that in the long run the summits are no longer cold and humid enough to allow subalpine fir to persist. The EPA (2002) notes that habitats of alpine and subalpine spruce-fir are likely to be reduced and possibly eliminated entirely in the long term in the contiguous United States as mountain habitats warm. With respect to Yellowstone National Park, Romme and Turner (1990) project an upward shift in the lower tree line of 460 metres owing to climate change, with severe negative impacts on whitebark pine. The fate of subalpine fir and whitebark pine, within the context of overall vegetation change on the buttes, and within the wider context of climate change, is an intriguing and important topic for further investigation.

The projected rate of climatic drying for the Sweet Grass Hills is similar to that of Cypress Hills. The GCM scenarios indicate future CMI decreases in the range of 10, 20 and possibly 30 cm in the 2020s, 2050s and 2080s, respectively. As at Cypress Hills, high wind speeds and chinooks may make large parts of the Sweet Grass Hills still more challenging for tree survival and regeneration than CMI values alone suggest. Some summit sites, despite favourable current CMI levels, are currently not treed, partly owing to high winds and exposure. The smallness of the Sweet Grass forests also increases their vulnerability. However, the Sweet Grass forests are relatively diverse in conifer species and are therefore potentially more resilient than the forests at Cypress Hills.

Some forest species may be able to migrate up-slope to higher elevations to compensate for climate change. However, the speed of climate change may make natural migration to higher elevations impracticable for some species. From a moisture perspective, the projected rate of drying is roughly equivalent to 60 vertical metres a decade. It would not be surprising if trees were unable to migrate up-slope, establish, and reproduce at that rate. Those species present both at the summits and far down the slopes do not need to migrate, but will see their natural range contract up-slope. Tree species found only towards the top of the buttes, i.e. subalpine fir and whitebark pine, are at clear risk of extirpation. Most other tree species will likely persist, either in small favoured sites near the summits or in steep ravine refugia. Physically, however, the bulk of the current forest can be expected to disappear. But because some mature trees can be resistant to dry conditions, in the absence of a major disturbance the current forest distribution may persist for some time, albeit with ever-decreasing regeneration. As at Cypress Hills and elsewhere, the fire risk is increasing and high-impact pathogen disturbances are becoming more likely over time. This means that sudden change is becoming increasingly likely.

The BLM's management objectives do not directly focus on forest ecosystem health. However, concern for wildlife habitat and for watershed protection indirectly suggests strong concern for forest preservation. Elk, mule and white-tailed deer, for example, shelter and feed in the dense forests of the north and east slopes of the buttes (BLM 1996). The state lands forestry policy management strategy (MDNRC 1996) is an uneasy marriage of sometimes incompatible objectives: maximum biodiversity and maximum timber cut revenues. Unfortunately, from a strictly harvest economics point of view, monoculture stands are often more efficient than natural mixed-forest stands. The state strategy targets reducing the risks of such natural occurrences as catastrophic fires and insect or disease attacks. It also envisions the replication of forests that were historically present on the Montana landscape. Given the magnitude of probable climate change, this will be impossible in many and possibly all cases. The strategy does not incorporate climate change impacts.

11. THE NATURAL STATE OF THE PLAINS AND ITS ISLAND FORESTS

Comparing northern Plains Holocene environmental conditions across millennia, or even across only centuries, indicates a highly variable landscape, with a given site sometimes densely wooded and sometimes largely grassland. It required only relatively small shifts in temperature or precipitation (i.e. in CMIs) to effect significant landscape and ecosystem change.

The historical evidence indicates Plains island forests of the previous 120 years were characterised by younger stands of trees, a much greater percentage of grass cover, and less closed canopy. Moose Mountain may have been an exception; early reports of that area speak of a fairly densely wooded landscape. However, even at Moose Mountain modern residents unanimously report much loss of grassland in living memory. Frequent fires seem almost certain to have been the key causative agent restricting island forests' growth. How many pre-European fires were natural and how many were set accidentally or deliberately by Aboriginal peoples is not known. There is some evidence that fire frequency increased temporarily to above pre-European frequency in the early years of European settlement and with the introduction of the railway, before declining to below pre-European frequency as agricultural settlement became more extensive.

For conservation purposes it is critical to bear in mind that Plains landscapes vary enormously not only across centuries and millennia but even on a year-to-year basis. By nature the Plains are a region of cyclical ecological catastrophes (Henderson 2001). In the early years of European contact a given site was sometimes flooded and sometimes under severe drought. Fire raced over the Plains at irregular intervals. Many animals (not just bison), and some insects, often had a devastating and unpredictable grazing impact over large areas – indeed, these natural grazing impacts were probably more akin to fire in the irregularity and variability of their impacts than they are comparable to the controlled and regular grazing patterns of modern cattle. For example, the vast expanse of cinquefoil and lupine encountered by Macoun (1882) in the Cypress Hills may well have resulted from a natural event, extreme grazing by bison, as shrubby cinquefoil generally increases with grazing (BLM 1996). The reality of a landscape that was so unstable in appearance often confused early assessments of the Plains, which were variously described as a desert or as rich and wooded meadowland, depending on the chance appearance of the landscape to the various European-origin travellers passing over it.

Typically our nature conservation management policies do not in practice acknowledge the catastrophic nature of the Plains. Within a protected landscape our conservation objectives focus on maintaining landscapes in stable conditions, or on maintaining balanced mixtures of ecosystems, or on promoting age-class diversity of keystone species. For example, current management policy targets the maintenance of Cypress Hills Provincial Park grassland in good to excellent range condition for grazing. Yet Godwin and Thorpe (1994) are correct to argue that uniformly excellent range condition is an unnatural state of affairs in the Cypress Hills grassland, at least on any other than a transitory basis. The Cypress Hills forest and all our study site forests are as naturally highly variable over time as the grasslands – most of these forests have, at some point in history, largely disappeared in a single fire event. The modern Plains public has the industrial world's distaste for catastrophic (albeit natural) environmental change and would not willingly accept this dramatic natural cyclicality. Instead we pursue public policy objectives of stability and careful ecosystem balances which may be sensible, but which do not reflect or emulate the pre-European Plains.

12. ISLAND FOREST FUTURES

One of the challenges in examining potential climate change impacts is that climate change will not be alone in impacting on the vegetation on a given site, but will be only one factor amongst many. As noted at Spruce Woods and Turtle Mountain, for example, wildlife grazing has influenced tree regeneration in the past, while elk impacts on aspen regeneration are an ongoing issue in parts of the United States. Most important, though, will be the choices we make in our management practices with regard to issues such as logging, wildfire and prescribed fire management, grazing, wildlife management, watershed management, pesticide use, reforestation, and species introductions. These management decisions will be critical to the existence and nature of the future island forests and their associated biodiversity[2]. Simply put, climate change and human management will together co-determine the landscape futures of the island forests. These 2 influences can be understood in terms of 2 key questions:

- what are the bounds of climatically possible landscapes; and,
- within those bounds, what type of future landscape, if any, do we wish to target?

We discuss the bounds of the climatically possible in this chapter and discuss alternative management responses in the following chapter, "Conservation Management Model Options."

CO_2 fertilisation has long been recognised to increase the water use efficiency of some plant species (Lemon 1983). This could possibly make such species more resistant to dry conditions and thereby have impacts within the island forests. But Wheaton (1997) notes there are many uncertainties about the sum effect of CO_2

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[2] "Biodiversity," a superficially straightforward conservation objective with near-universal support, is a dauntingly complex concept in practice. To begin with, there are at least three different types of biodiversity, including intraspecies genetic diversity, species diversity, and community or ecosystem diversity. A fourth level, landscape or habitat diversity, is also sometimes considered. The definition of biodiversity of the U.S. Office of Technology Assessment (1987) is: "the variety and variability among living organisms and the ecological complexes in which they occur. Diversity can be defined as the number of different items and their relative frequency."

Anderson et al. (1998) point out that management designed to increase biodiversity at one level may reduce it at another. For example, while moderate habitat disturbance can promote species diversity, it can also simultaneously reduce intraspecies genetic diversity by promoting the mixing of previously isolated genotypes. This becomes an issue when deciding whether or not to create ecological corridors for species or subspecies migration (an oft-suggested prescription to adapt to climate change).

"Natural" or "wilderness" ecosystems are not necessarily maximally diverse, as is often assumed. Often moderate management disturbance can increase some types of biodiversity. Where diversity and naturalness conflict, there is controversy in North America. For example, Smith and de Smit (1979) value the visual diversity provided by introduced conifers at Turtle Mountain and note that the plantations have greatly increased both floral and faunal diversity in the park. Ransom (1969b) is against this diversity on the grounds that it is not natural. The operative management plan for Turtle Mountain (MNR 1985) reflects the ambiguity towards introduced species diversity. It states on the one hand that: "No further white spruce plantations will be created in the park." But it also establishes that "existing plantations may be selectively removed as they mature," a policy that leaves room for discretion, and the plan states definitively that at Max Lake, a highly visible and visited part of the park, "the white spruce plantation will be thinned by selective cutting to promote tree growth and create an aesthetically pleasing softwood forest." The fact is that the local public is very fond of the spruce at Max Lake and it would be politically difficult to remove that stand. Henderson (1987) found park planners unwilling to consider the introduction of spruce at Moose Mountain, despite wildlife benefit, aesthetic advantages, and increased biodiversity. Sheard and Blood (1973), writing in the context of Canadian national parks policy in general, believe that parks policy should avoid "artificiality."

Europeans, by contrast, rarely hesitate to "improve" a landscape. Moore (1987) states that: "Moderate physical disturbance of habitats in British nature reserves is nearly always beneficial," while Mabey (1980) states that "the aim of management on most [British] sanctuaries is to increase the diversity of habitats, often to a degree that could never have existed 'naturally' on the site."

fertilisation. Most recently the EPA (2002) summarised experimental study and modelling on the question as indicating that forest productivity will increase in the United States overall, but that such increases would be strongly tempered by local conditions, such as moisture stress. In chapter 5 we noted that moisture stress at the time of regeneration was very likely the key constraint on tree success in our study region. If this is correct, then a key issue would be whether CO_2 fertilisation has any positive impact on the survival of very young seedlings under drought stress.

Modelling and empirical study by Gracia et al. (2001) suggest that any positive CO_2 fertilisation effect is neutralised amongst evergreens whose growth is constrained by moisture limitations. Even to predict overall changes in forest CO_2 uptake and storage, independent of inter-species variations, is not yet possible (Gitay et al. 2001). One major problem in predicting CO_2 enrichment impacts on a specific species is that the impact occurs on all vegetation simultaneously. It is not enough to know the CO_2 response of one species of tree, rather one needs to know the relative growth advantage, if any, gained by all vegetation species competing for resources at a given site, be that vegetation trees, shrubs or grasses. One possibility is that understory species will react more quickly than trees to increased CO_2, destabilising the forest ecosystem (Scott and Suffling 2000). In the European context, de Groot and Ketner (1994) predict that thermophilous species will invade forest understories; that forest ecosystems will thereby be destabilised; and that native species may disappear. CO_2 enrichment will also not ameliorate the increasing risk of fires and insect or pathogen attack that results from climate change (unless trees are made more robust through CO_2 fertilisation).

Several other important anthropogenic-origin atmospheric changes other than increases in CO_2 levels are ongoing as well. UV-b radiation levels and ground-level ozone levels are increasing. These can be expected to impact negatively on vegetation and to possibly nullify any positive CO_2 enrichment effect, but the detail is not known and there are many uncertainties (see the ongoing Wisconsin aspen field experiments at: http://oden.nrri.umn.edu/factsii/).

In the absence of moisture limitations or other constraints, a moderate increase in air temperatures and growing season at mid and higher latitudes might be expected to increase plant growth overall. Increased photosynthetic activity was in fact Myneni et al.'s (1997) empirical finding for much of Canada over the period 1981-91, results attributed to a longer growing season. However, this general finding of increasing plant growth does not tell us much about which species or assemblages will be relatively advantaged or disadvantaged in the increasingly moisture-constrained world of Plains island forests.

In addition to rising temperatures, rising CO_2 levels, a lengthened growing season, increasing ground-level ozone, rising UV-b levels and changing moisture regimes, there are still other ongoing climate changes. For example, there are changes in the timing and intensity of freeze-thaw events, diurnal temperature patterns (Gitay et al. 2001), and storm and wind stress events. All such factors may be expected to influence the distribution or survival of various tree species (Macdonald et al. 1998), but the detail of how this will occur is not known.

Climate variability is important too. Many impacts researchers are of the opinion that climate variability is increasing, even as "average" climate shifts globally. This would imply, for example, a greater future frequency of extreme drought and flood events. Thunderstorms and windstorms may also increase in frequency. An increased frequency of extreme events would be an accentuation of the catastrophic natural controls that already characterise the Plains environment (as described in the chapter 11). Future scenario CMI values refer to average years only and individual years will vary around that average. If there is increasing variability, it will be centred on declining average-year soil moisture availability and will likely further increase stress on trees.

Climate change is already under way and current CMIs at the island forest study sites will already be lower than those indicated in the 1961-1990 normals, i.e. they will already have shifted part way towards the 2020s scenarios. Averaged over all 5 study sites the mean modelled CMI decline amongst the 12 scenarios is 10.1 cm between the normals and the 2020s. None of the 5 sites deviates greatly from the mean CMI decline (the range is

8.6 to 11.0; see Table 5). As the year 2000 is halfway between the 1961-1990 normals (centred on the 1970s) and the 2020s, we are probably already experiencing island forest climates with roughly 5 cm less annual moisture available for plant growth. This is a shift one-third of the way between the boreal zone and aspen parkland, or between the parkland and grassland ecoregions. If we had used the period 1951-1980 as our baseline period, the indicated CMI decline by 2000 would be somewhat greater still.

Everything else being equal, one might therefore already expect to see evidence of declines in tree productivity, declines in reproductive success, or signs of stress. This has not yet been demonstrated in any systematic sense – but nor has it been systematically monitored for at any of the 5 sites. Some dieback may be beginning to occur in the case of aspen at Moose Mountain and the Alberta Cypress Hills West Block, but the evidence is inferential. Ironically, one conservation management concern at 4 of the 5 sites is tree expansion onto grassland enclaves, rather than tree decline. This expansion is widely agreed to be attributable to the suppression of wildfires. However, even given the continued suppression of fires, tree encroachment cannot continue indefinitely as CMIs decline further, although predicting precise thresholds is difficult.

The reason for threshold uncertainty is that vegetation responds after the fact to climate change, i.e. it is natural for a given ecosystem to be “behind” environmental conditions to some degree, a condition termed ecological inertia. According to Davis (1984), plant and animal communities are always to some degree in disequilibrium, continually adjusting to new climate trends. According to Pielou (1991), the belief that “the living world is marvelously and delicately attuned to its environment – is not so much a scientifically reasonable theory as a mystically satisfying dogma.” In a rapidly changing environment, such as that undergoing significant climate change, the degree of disequilibrium may become significant. It is true, as Pielou (1991) states, that: “Many plants have an astonishing ability to persist in unfavorable environments.” However, such persistence in the face of major environmental change may only be a prelude to sudden and rapid change, triggered by some disturbance event such as fire or insect attack. In this regard Anderson et al. (1998) warn that ecosystems can sometimes absorb stresses over long periods of time before crossing a critical threshold which may lead to rapid ecosystem and landscape modification. Such processes are not linear and are very difficult to forecast. Saporta et al. (1998) add that the climate change impact on mature trees is not likely to be noticeable until biological thresholds are reached and dieback results. Solomon (1994) agrees: “Loss of a cohort may not be evident in change to forest structure for decades or even a century as the established trees continue to survive under increasing environmental stress.”

We know from paleontological evidence that ecosystem shifts driven by natural climate change can be rapid. Around 10,000 BP, at the Pleistocene-Holocene boundary, pollen records show a dramatic synchronous spruce decline in a wide band across North America from Nova Scotia to Minnesota. Spruce was replaced by a much more diverse forest of jack pine, red pine, eastern white pine, balsam fir, white birch, elms and oaks (Pielou 1991). At some sites forest species changeover happened in less than a human lifetime (Watts 1983). While the North American peoples of 10,000 years ago certainly experienced a period of rapid natural climate change, current emissions-driven climate change is occurring at a much faster pace – faster in fact than any known period of natural climate evolution.

With respect to areas north of our study region, Scott and Suffling (2000) concluded, with reference to Lenihan and Neilson (1995), that all the prairie-parkland national parks (Elk Island, Prince Albert and Riding Mountain) would undergo a shift to another forest formation type, including grassland invasion, as a result of climate change. For these parks they predicted increased forest fire frequency and intensity; increased forest disease outbreaks and insect infestations; and loss of boreal forest to grassland and temperate forest. De Groot et al. (2002) also predict increased fire frequency and intensity for Elk Island, Prince Albert, Riding Mountain and Wood Buffalo National Parks. Scott and Suffling (2000) warned that climate change represents “an unprecedented challenge for Parks Canada” and predicted that “current ecological communities will begin to disassemble and ‘resort’ into new assemblages.” With respect to areas to the south of our study region, the *U.S. Climate Action Report – 2002* (EPA 2002) suggests ponderosa pine and arid woodland communities could expand in the western United States and that habitats of alpine and subalpine spruce-fir are likely to be reduced and possibly eliminated entirely. Conifer encroachment is currently displacing sagebrush and aspen communities.

Thorpe et al.'s (2001) study interest was primarily the impact of climate change on sand dune areas across the Canadian Plains. For all their study sites they determined that there would be a shift away from wooded cover and towards grassland. One of the sites they examined, the Nisbet Sand Hills in central Saskatchewan, happens also to be an island forest (Fort à la Corne). They concluded that the climate impact "will be especially serious for the heavily forested Fort à la Corne / Nisbet Sand Hills, and implies a gradual loss of timber production and forest-dependent biodiversity." By the 2050s they believe the likelihood is that "Fort à la Corne, currently almost entirely forested … will shift to a climate which apparently only supports grassland. Timber production at Fort à la Corne will decline as the climate becomes less conducive to tree growth, and eventually disappear as forest is converted to non-forest."

Loehle and LeBlanc (1996) warn that climate change induced changes in forest disturbance regimes may be substantial enough to alter or destroy current forest ecosystems. Change is obvious when a disturbance event is followed by failure of tree regeneration. However, the probable vegetation endpoint of the island forests (an absence of trees outside of localised favourable sites) is more predictable than the precise pathway of how a particular forest will arrive there. Harvell et al. (2002) note that average winter temperatures are expected to rise to a greater extent than average summer temperatures (our GCM scenarios are ambiguous on this question – see Table 3), which will lead to greater overwinter survival of pathogens and increased disease severity. Saporta et al. (1998) agree, noting that droughty conditions weaken trees' defences to more virulent pathogens. Gracia et al. (2002) note that as conditions become more xeric the lifespan of conifer needles is reduced and that in replacing needle loss a conifer is put under increasing stress. One possible pathway of change may therefore be slow and cumulative decline, such as aspen break-up as it passes maturity, is subject to insect attack, and experiences conditions too dry for regrowth.

Alternatively the change mechanisms may be spectacular and catastrophic, such as a major fire. Sediment cores from a lake in southwestern Nova Scotia show, for example, that every time a relatively new "immigrant" tree species became dominant over other species in the surrounding forest during a period of climate change, the changeover resulted from a fire event (Green 1986). Thorpe et al.'s (2001) prediction of a gradual forest decline at Fort à la Corne may be optimistic – tree loss may be sudden and permanent if the forest experiences a major fire or insect disturbance, for example, especially if occurring simultaneously with a particularly droughty period. Wells (1970) considers a "*coup de grâce*" administered by fire to be "a likely fate for nonsprouting boreal conifers stranded under an unfavorable climate." Aspen, which is normally stimulated to sucker by low intensity fires, may be destroyed by an intense fire that damages root systems (Perala 1990; Strauss 2001). Such intense fires are increasingly likely as stands age and fuel loads increase at all our study sites and, under conditions of climate change, could permanently set back aspen.

It is probable that most of the currently extant island forest tree species will be able to persist in a few sheltered and particularly favourable sites, just as they presumably did at the peak of the hypsithermal. Highly dissected landscapes such as the Cypress Hills, or landscapes with a least a few very steep coulees, such as the Sweet Grass Hills (Ribbon Gulch) and Spruce Woods (Epinette Creek), can again provide refugia for remnant species, while the relatively open, rolling terrain of Moose Mountain and Turtle Mountain offers less hope of refugia and a greater likelihood of species extirpation. However, subalpine fir and whitebark pine on the Sweet Grass Hills, apparently at the limits of survival now, will very probably disappear entirely.

As we noted in introducing this chapter, climate is only one co-determinant of the island forests' futures. We discuss management options below, some of which could at least delay forest disappearance and significantly influence forest evolution, though none can avoid change entirely.

13. INTERIM CONSERVATION MANAGEMENT MODEL OPTIONS

In the relatively new policy environment of rapid emissions-driven climate change it is remarkably unclear what our long-term nature conservation landscape visions should be. In practice the way we react to climate change depends critically on whether we treat it as a natural phenomenon or not. Peterson and Peterson (1992), for example, list natural and human-induced agents of change in the Cypress Hills and class climatic change as a natural agent. If we treat climate change as a natural event, then it follows that vegetation community changes that ensue from it are equally natural. Many would conclude that these vegetation changes are therefore not, in principle, to be resisted.

However, logically this viewpoint is unsustainable. Current climate change is, in largest part, emissions-driven. The majority of "natural agents" noted by Peterson and Peterson (1992b), such as forest succession, grazing and browsing by native animals, insect and disease epidemics, and wildfires, will, to an ever-greater degree, be driven by climate change. The superficially "natural" landscapes that result from these phenomena will actually be the by-products of atmospheric pollution.

Already on the Plains we have only semi-natural landscapes heavily impacted by cultural forces. By 1983 (Packer 1983) 280 of 1755 species of vascular plants in Alberta were non-native. In a world of significant emissions-driven climate change, the distinction between natural and exotic species, or between natural and cultural landscapes, breaks down completely. "Wilderness" may soon be a word that can only accurately be used as a descriptor of landscapes of the past, not those of the present or future. Yet the preservation of "native" or "natural" communities typically remains the touchstone of most conservationists.

Nash (1982) identifies the development of the North American belief that the need to preserve wilderness is the dominant and highest nature conservation objective. Under conditions of climate change, if we wish to maintain species and ecosystem diversity, i.e. if we wish to meet Leopold's (1949) injunction to "save all the parts," we may have to abandon the wilderness preservation model and adopt increasingly intensive management policies. These could include deliberately assisting the movement of species to newly suitable habitats in the style of British conservation. This question is particularly acute for small disjunct ecosystems such as island forests, where natural migration (for example, of new tree species or of climatically more suitable genetic varieties of extant tree species) will typically not be possible without human intervention.

If the passing of wilderness makes conservation management more anguished, it also makes it interesting, for it opens up a range of possible endpoint landscapes to aim for. No longer are we restricted to trying to maintain or recreate the original state of nature. Previously the conservation debate centred on the range of management methods to attain a pre-ordained wilderness end. Now it must consider and evaluate a range of possible landscape endpoints, as well as encompassing the range of management methods to reach those endpoints.

While climate change cannot be avoided, we are not helpless. We have the choice of a suite of landscape ecosystems viable within given climate parameters. As a matter of conservation policy we can aim to extend ecological inertia, have no impact on it, or reduce it. Vegetation associations that are most "in tune" with the evolving climate will require the least maintenance, i.e. the least degree of human intervention. Conversely, those vegetation ensembles increasingly outside of their natural climate norms can be expected to require increasingly intensive and active human intervention and management to survive. However, with a commitment to a high degree of human intervention, it will be possible in some sites to maintain vegetation (and associated fauna) assemblages that would otherwise certainly disappear. In the words of the *U.S. Climate Action Report – 2002* (EPA 2002): "For forests valued for their current biodiversity, society and land managers will have to decide whether more intense management is necessary and appropriate for maintaining plant and animal species that may be affected by climate change and other factors … One possible adaptation measure could be to salvage dead and dying timber and to replant species adapted to the changed climate conditions." Kurz and Apps (1993) believe forest managers need to abandon the idea of managing a steady-state system, given that we will be experiencing a changed disturbance regime and climate future. Stability may not be possible, and aiming to build resilience[3] into the ecosystem, and keeping our ecological options open, may be more appropriate. Pernetta (1994) believes that: "A static approach, will ultimately result in protected areas being over-taken by events, they may well exist in areas no longer suitable for the maintenance of the species and ecosystems they were originally designed to conserve." Halpin (1997) notes that natural disturbances will have to be managed, that exogenous stresses will need to be controlled, and that habitat modifications may be necessary to "reconfigure protected areas to new climatic conditions."

One point is worth noting before consideration of our conservation management model options. As the Plains areas which surround the island forests experience a drier climate, the already high value of island forests as oases of trees, small lakes and wetlands, and as sources of water to surrounding areas, will be accentuated. There will be a strong preference to conserve tree canopy of some kind wherever possible. It is very probable that the public will countenance even radical measures to maintain tree cover on island forest sites, if there is a solid understanding of the consequences of inaction. At present such a climate change understanding does not exist amongst professional managers, much less amongst the general public, which suggests a need for sustained consultations and information dissemination (see chapter 15).

We first present 2 historically and currently popular conservation models unlikely to be able to cope with climate change. We then present "Managed Retreat", our interim recommended management model. Then, in the chapter following, we briefly present 3 possible conservation visions which may emerge as mainstream North American conservation paradigms of the future.

(a) The As-if-wilderness Model

In this model we assume that wilderness exists or can exist in reality, that wilderness is the ideal state of a designated conservation area, and that a protected landscape is best left untouched by active management so far as possible. The approach is laissez-faire, except that where we have interrupted natural processes in the past (for example, by suppressing fire), we may seek to reinstate them or replicate their effects in so far as is practicable. Following this management model in the island forests would for the most part give the appearance of business as usual.

The advantages of this model in the island forests are that it is traditional, it is initially inexpensive (as there are few active management costs), and it is relatively painless in a managerial and political sense (as it involves no major shift in management direction). Philosophically the approach could be underpinned by assuming, for practical purposes, that climate change is simply a natural phenomenon that should not be opposed or actively managed for, but whose consequences should simply be lived with. In this view, if the Cypress Hills climate dries to the point where spruce no longer regenerate, we would allow the spruce to die out, even if by planting seedlings we could maintain its presence in the forest.

Adopting this approach in the island forests could be termed "unmanaged retreat." It would open the way to sweeping, sudden change, perhaps by major fire or insect disturbance, followed by an ecosystem shift that cannot be predicted in detail, but which would almost certainly result in a landscape with greatly reduced tree cover. Alternatively, forest decline might also occur gradually. Given the value of forests to people on the Plains, and

[3] Like biodiversity, resilience is a complicated concept with various definitions. Generally speaking, it refers to the ability of an ecosystem, however defined, to persist in the face of environmental changes and disturbances. Arrow et al. (1995) define resilience as "a measure of the magnitude of disturbances that can be absorbed before a system centered on one locally stable equilibrium flips to another." Resiliency may be dependent on the presence and health of a few keystone species, or ecosystem "drivers," in Walker's (1995) terminology. Resiliency is higher if the number of drivers is higher, or if other drivers exist with the potential to fulfil the current keystone species' role. Species "redundancy" as Malcolm and Markham (1996) coolly term it, generates resiliency. As the island forests possess relatively few tree species, there is relatively little redundancy, and it would fall to management to introduce new potential keystone species.

given people's natural conservative tendency to favour the landscapes they know (at least the aesthetic ones), unmanaged retreat is unlikely to be publicly acceptable in practice, unless change is so slow that people do not perceive it. However, under climate change the likelihood of catastrophic and highly visible disturbance and change is increasing.

Intellectually, the model is questionable in so far as it does not really result in a wilderness landscape, but rather one particular cultural landscape – the one arrived at by a laissez-faire response to emissions-driven climate change. Practically, our past actions against wildfires in the island forests already demonstrate our unwillingness to accept, or risk, catastrophic change to these valued ecosystems.

(b) The Frozen Landscape Model

In this model we assume that the ecosystem structure present on a given landscape at some given point in time is the correct one. There are at least 2 subvariants. The modest subvariant seeks to maintain the landscape as it is now by resisting (further) anthropogenic change. The more radical, purist or ambitious subvariant seeks to restore the landscape as of some point in the past by reversing anthropogenic changes. Most ambitious is the attempt to recreate the landscape as of a point prior to any (European) anthropogenic impacts. The Montana state lands forestry policy (MDNRC 1996), which aims to replicate forests that were historically present (no target date specified) on the Montana landscape, and the Shilo vegetation management plan (Dillon 1996), which aims to maintain Spruce Woods vegetation cover types and distributions as of 1988, are 2 operative examples of this model. Similarly, the operative vegetation management plan for Duck Mountain Provincial Park in Saskatchewan targets the restoration of the grassland component of that park to its pre-European areal extent (Wright et al. 1995). Recent air photo study and GIS mapping also makes this model potentially applicable in several of our island forest study sites beyond Spruce Woods.

The model has several advantages. First, it is in principle easily understood by professionals and the public. Second, when we target the preservation of the existing landscape we have the advantage of knowing exactly what we are trying to do and can measure success or failure relatively clearly. Even if targeting the re-creation of a past landscape, we are sometimes able to establish with reasonable certainty the nature and proportions of the basic ecological landscape constituents. Third, the fact these landscapes exist, or once existed, provides empirical credibility to the targeted landscape. They are not "pie in the sky" visualisations.

Nonetheless, the model has fatal weaknesses. As discussed in chapter 11, the Plains landscape, in the absence of human intervention, naturally changes radically on many time scales, be it from year to year or from century to century. It is hard to see what ecological basis there could be to justify targeting a Plains landscape as of an arbitrary given point in time. Most crucially, from a practical standpoint, any historic landscape becomes increasingly unsupportable as climate change advances. In the context of Canada's national parks Scott and Suffling (2000) believe, reasonably, that a landscape maintenance strategy is materially impossible, whatever its philosophical merits or demerits. It simply becomes too difficult to repress or counteract every disturbance, whether gradual or catastrophic, that acts to shift the landscape towards a new equilibrium more closely attuned to the changing climate.

(c) The Managed Retreat Model

This model assumes we wish to preserve island forests, or at least some elements of them, and are willing to intervene fairly aggressively to do so. The model accepts landscape change driven by climate change as inevitable, but by active management seeks to strategically delay, ameliorate and direct change. Lopoukhine (1990) believes active management is the only alternative for protected areas given the reality of climate change impacts. Scott and Suffling (2000) agree that active management is warranted in response to climate change and note that intervention strategies would often be species specific. Zoning will be useful within this model to allow differentiated responses to climate change within a given site.

A high disturbance rate changes the species composition of a forest regime and increases the rate of response to climate change (Overpeck et al. 1990). This works against the managed retreat model's objective of slowing the rate of change. Therefore, typical managed retreat tactics may include the aggressive control of wildfires, the active suppression of unwanted pathogens, and the aiding of regeneration of key extant species. For example, if lodgepole at Cypress Hills were threatened by a devastating mountain pine beetle attack, we might intervene so far as reasonably possible. Similarly, if climate change in the Cypress Hills results in spruce being unable to regenerate naturally, but by planting seedlings we are able to extend its presence, we would consider doing so. Intellectually the managed retreat model incorporates the reality that when faced with landscape-altering disturbances in future, we will most often not be able to effectively distinguish between a disturbance that results, indirectly, from climate change and one that would have occurred anyway.

The managed retreat model seeks not only to delay change, but to direct it. We may search both within the island forests and beyond for genetic strains of existing island forest species most able to survive in warmer and more xeric conditions. If necessary, we could introduce these strains to the island sites. For example, particularly drought-hardy varieties of green ash, a tree present in Spruce Woods, Moose Mountain and Turtle Mountain, can be found in the arid Big Muddy area of southern Saskatchewan (Bill Schroeder, AAFC-PFRA, pers. comm. 2002). Seed from this source could be considered for introduction in the aforementioned island forests, if necessary. If a particular species' demise is inevitable, replacement with a next-best substitute could be considered. For example, if at Cypress Hills the loss of lodgepole pine is unavoidable, while ponderosa pine appears to be viable, ponderosa might be introduced as part of a managed retreat of the pine ecosystem.

The concept of introducing new species to a protected natural area is in direct opposition to current official nature conservation policy and would meet with passionate resistance from some people. Unofficially we have never in practice stopped introducing new species, such as sport fish or game birds, into natural areas. In fact management has recently introduced wild turkey into the Cypress Hills. Such introductions typically have tourism or recreation rationales and are hard to defend on nature conservation grounds. The type of introductions the managed retreat model envisions would be more defensible philosophically, as the first-order objective is ecosystem conservation, rather than tourism.

Managed retreat is a conservation strategy employed in Britain, for example, in response to loss of coastal conservation areas as a result of sea level rise (a phenomenon driven there largely, though not entirely, by climate change). The strategy includes the creation of substitute ecosystems (for example, new salt-water marshes) as offsets for those likely to disappear. In the context of island forests, this would imply taking into account ecosystem changes going on beyond any one individual forest and considering the "big picture" of what is occurring in the entire Plains island forest archipelago. The knowledge that a particular species or ecosystem is under threat in one island forest, for example, may incline us to consider its introduction in another, should new climate conditions support it.

Although the name sounds defeatist, and is probably not best suited for public communications exercises, managed retreat does not necessarily end in defeat, for the probable eventual loss of some species may not necessarily be critical. Achuff (1992, citing, among others, Huntley and Webb 1989; Raven and Axelrod 1978; Johnson 1985) notes that modern ecological thought is trending away from the idea that natural ecosystems are tightly-knit stable entities in terms of structure and function and towards the view that the majority of species interrelationships are "individualistic." Under this view the loss or removal of one species is unlikely to lead to the collapse of the whole ecosystem and species substitutions can occur which maintain ecosystem processes. Obviously, however, if in the long run we are unable to preserve *any* species of a keystone group like trees at a given site, the basic ecosystem will be lost. Even the removal of a genus such as pines would have major implications. Equally, the loss or severe decline of aspen, on which so much flora and fauna is ultimately dependent, would be particularly difficult, or impossible, to compensate for. Under conditions of extreme climate change managed retreat can only delay major change, not prevent it.

A major practical concern with the managed retreat model is that it is more expensive than passive management. It requires careful ecosystem and species monitoring and study. As climate change advances, it will require increasingly intrusive – and expensive – active management to "hold the line."

Nonetheless, by extending the life of existing and highly-valued eco-landscapes so long and so far as is reasonably possible, managed retreat provides many public benefits. It reduces the risk of catastrophic change. Most importantly, it keeps options open. We do not know with certainty the speed or final outcomes of emissions-driven climate change and a strategy of managed retreat gives us both time and maximum choices as change is ongoing. In the most optimistic view, if we are very fortunate, we may even be able to retain significant stands of most keystone tree species indefinitely with active management. We recommend the adoption of the managed retreat model as an interim adaptation strategy to climate change in the island forests.

14. LONG-RUN NATURE CONSERVATION VISIONS

The managed retreat model provides an interim strategy only (although the interim may be many years). Successful retreats must end somewhere, and we must decide as soon as is practicable what we are retreating to. The challenge of agreeing on new target landscapes will be both exciting and controversial. Extensive communications and consultation exercises are required, for the issues of changing climate and changing ecosystems are both unfamiliar to most people and complex to explain in meaningful detail. In chapter 15 we examine the type of communications and consultations exercises that are required to enable people to choose amongst new target landscapes and management regimes. What their preferences will be is not yet known. However, in anticipation of possible outcomes, we outline 3 possible future visions of nature conservation. Something like one of these visions may eventually emerge as the public's preferred direction for long-term adaptation to climate change. Key underlying issues are whether to target minimal management and intervention, or maximise biodiversity, or give free reign to unsystematic personal and societal preferences for particular species and ecosystems.

(a) The Low-input Stability Vision

This vision has as its objective the restoration of a low-input management style on whatever new ecosystems are in reasonably stable and self-sustaining equilibrium with the eventual end state of emissions-driven climate change. The less human intervention required to support it, the more satisfactory the destination ecosystem, might be one measure of success. We could arrive at such a new eco-landscape by following either interim management model (a) or (c) in chapter 13. If we arrive at a new eco-landscape via model (a), we will have had no choice about the new ecosystems that result from climate change. However, if we arrive via model (c), we will have had a series of choices along the way, for example, about potential new species varieties or new species introductions, i.e. we will have made choices about destination ecosystems. Most likely we will have chosen to retain greater forest cover.

In effect, although the new eco-landscape could in no way accurately be termed wilderness, the objective would be to once again manage it in wilderness style. Just as in chapter 13's as-if-wilderness management model, a pragmatic variant of this vision might be unwilling in practice to countenance catastrophic disturbance events such as major wildfires, while a purist variant would allow even catastrophic disturbances to run their course. This vision could deliver a stable and healthy eco-landscape at low management cost. However, the choice of destination ecosystems would be relatively narrow and climatically prescribed, given the reluctance to engage in active management indefinitely.

(b) The Maximum Diversity Vision

In this vision the maintenance or increase of local or regional biodiversity is considered the overriding objective. New genotypes of extant species or entirely new species could be introduced to either buffer an existing site ecosystem against climate change or to provide habitat for species threatened elsewhere. Under this vision we would consider introducing and supporting new ecosystems, particularly to substitute for those under threat elsewhere. A high degree of intervention and management would likely be required indefinitely. Interim management model (c) would be compatible with this vision.

Permanently intensively managed ecosystems are common in current European conservation practice (Henderson 1992). They are philosophically very distant from the classical wilderness preferences of many North American conservationists. Nonetheless, this approach is gaining ground in North America – we increasingly design permanently managed habitats for some species, such as managed marshes for waterfowl. To gain the necessary financial support for managed systems the focus is often species specific, although a range of other species may collaterally benefit. This vision could deliver a high degree of biodiversity in response to climate change, but it would be expensive, potentially arbitrary in its particular species or ecosystem focus, and would not appear "natural" to some traditional conservationists.

(c) The Garden Vision

In this do-as-you-please vision ecosystem management is entirely driven by personal and societal preferences for particular species and ecosystems, preferences which derive from a complicated tangle of cultural, aesthetic, historical or utilitarian predispositions. No account is taken of local or global biodiversity considerations. Flora and fauna are introduced, maintained or eradicated in a societal parallel to the kind of individual decision-making that generates the landscape of a private garden. Interim management model (c) would maximise the possible garden vision options.

A bit of the gardening approach already exists in modern North American conservation practice. We all have our favoured species we are inclined to support by habitat manipulation or breeding site provision, be it rainbow trout, wood ducks, bluebirds or elk. We even have preferences amongst ages or types of individuals within a given species. For example, younger, vigorous aspen is perceived by some as more desirable than a "decadent" or insect-infested stand, however equally natural these states of aspen may be.

The gardening vision of nature conservation is widely applied at conservation sites in Britain and is also widespread in France and Japan, for example. Resulting landscapes tend to be diverse (though not always: sheep-grazed swards are favoured in Britain for aesthetic and historic reasons, but are not always diverse), high maintenance and highly attractive. In practice in North America the gardening approach would favour the introduction or support of economically valuable fish and game animals within a landscape, habitat manipulation to favour species such as hunted waterfowl or big game, and the retention or planting of aesthetically pleasing vegetation (such as the exotic spruce at Turtle Mountain). Hunting and recreation values would be maximised. Although this vision does not target biodiversity, it often delivers reasonable diversity in practice. However, it can at times damage ecological integrity via species introductions, while unfavoured species or ecosystems can be lost through disinterest and neglect. It is arbitrary in nature and expensive in practice, but often popular with particular stakeholders.

15. THE ROLE OF CONSULTATIONS

The Canadian Environmental Advisory Council (1991) states that the way to gain public support for the preservation of natural areas is through increasing the public's understanding of their nature and value. Scott and Suffling (2000) conclude that building stakeholder capacity is essential to the successful development and implementation of climate change adaptation strategies. The US National Assessment Synthesis Team (Joyce et al. 2001) recommend public and decision-maker education about the implications of climate change as part of an adaptation process. In the case of the island forests we are now faced with the unpleasant reality that straightforward preservation is no longer a viable option. It is necessary both to explain this to the public (a communications exercise) and also to ask the public how it wishes to see the island forests managed in response to the changing climate (a consultations exercise).

The communications exercise – explaining "the facts" to people – will not be easy. Climate change and its impacts are ongoing, complex and difficult to forecast. The issues are challenging to explain or understand. There is uncertainty as to the bounds of possible vegetation futures on a given landscape and uncertainty as to the timeframes and exact mechanisms of landscape change.

The consultations exercise – asking people how they wish us to adapt to climate change – will be difficult as well. Agreeing on a desired target landscape is the first consultations objective, while selecting from a range of management options to reach that end state is the second consultations objective. Agreement will not be easy. Many people have a deep attachment both to the concept of wilderness and to our current landscapes that may make them reluctant to rationally discuss inevitable change.

Despite these difficulties, communications and consultations exercises are both necessary and unavoidable. We have no certainty as to what public preferences might be as to target landscapes when forced to choose between landscape futures where a status quo option is not on the table. The landscape choices we make will be so far-reaching in their consequences that it would be morally unconscionable not to solicit public input. And on purely practical grounds, some of the more radical measures that managers may wish to take in response to climate change have little chance of implementation without public debate.

Of course, there is really no such thing as "the public." Key stakeholders that have an especial interest in the island forests include Native peoples, local residents and various conservation groups. The views of Native North Americans will be particularly important. There are reserve or reservation lands within or adjacent to the Cypress Hills, Moose Mountain and Turtle Mountain island forests and these forest sites have long been traditional Native hunting and cultural use areas. For some Native peoples the Sweet Grass Hills are a sacred site; during the EIA process around proposed mining operations in this area many Native peoples expressed concern about the possible visual and audible intrusions to the natural environment that development might impose (BLM 1996).

Local residents, both Native and non-Native, derive economic and recreational benefits from the island forests and may also value them for cultural or spiritual reasons. Local people are particularly important consultation partners because they are often the landholders of important adjacent private lands. Parts of all of the island forests are privately owned (particularly at Turtle Mountain) and consultations will be an important mechanism to influence private land management as well as win input to, and support for, appropriate public lands management.

The island forests themselves are logical sites for consultations if there are reasonable visitation numbers. Where the island forests encompass parks with interpretation centres, these centres should display and interpret scenarios of future climate impacts for their region. Visitors can also be surveyed as to their preferences for future landscapes at these centres. Many of the island forests will already have stakeholder advisory groups of some type which can be used as a basis for consultation on adaptation to climate change. Consultations with non-local interest groups will be necessary as well. Consulting on nature conservation adaptation in the island forests with interested regionally or nationally based stakeholders could serve as a useful stepping stone to similar consultations with these bodies over adaptation policy in the wider landscape.

The International Peace Garden at Turtle Mountain is an excellent potential site for species test plantings for experimental purposes, and also for climate change interpretation and social survey. The site is relevant both to the American and Canadian Plains, has a large binational pool of visitors, is directly adjacent to an impacted island forest, and is not government-owned (which means it can be more flexible in climate change interpretation and in social survey design, and will potentially be perceived with less suspicion by some individuals). Joint U.S.-Canadian institutional sponsorship of a climate change interpretation centre here would be most effective.

16. CONCLUSIONS AND RECOMMENDATIONS

(a) Introductory Comments

Rapid climate change fundamentally changes the context of protected areas planning and of nature conservation policy-making. However, this is little understood amongst policy-makers, managers, or the wider public. We note that a failure to incorporate climate change impacts within strategic planning is typical of conservation management throughout the northern Plains region well beyond the issue of island forests. For example, Montana's forestry policy management strategy for state lands (MDNRC 1996) postulates a return to historic forest landscapes, which climate change almost certainly makes an impossible objective. Manitoba's "Protected Areas Initiative" (Manitoba Conservation 2000), which aims to protect representative ecosystems of the province, is based on a division of the province into 18 natural regions and sub-regions defined by physiography and common climate, and does not reference climate change. Yet while physiographic factors may be reasonably enduring, the current climate is not. Some climate-physiography combinations may shrink in extent, some may disappear entirely, and entirely new ones may arise. Flora and fauna (i.e. the ecosystems and biodiversity the Protected Areas Initiative is intended to protect) will also change accordingly. Saskatchewan's "Representative Areas Network" is similarly intended to conserve the range of biodiversity in that province. It is based on a division of the province into 11 distinct ecoregions defined in part by climate, as well as by geology, soils, plants and animals, the latter three of which are ultimately dependent on climate. Alberta's "Special Places Program" is a comparable biodiversity protection initiative which also does not incorporate climate change in strategy design or implementation. A realistic biodiversity strategy must take into account that climate, and therefore flora, fauna, hydrology and soils, will not be static over this century, and that a conservation strategy based on trying to maintain the ecological status quo by protecting selected landscapes from human impacts other than climate change will not succeed. In a world of climate change, selection of protected areas may need to focus on site heterogeneity and habitat diversity (as these provide some buffer against climate change) rather than on representativeness. As well, preserving some elements of biodiversity will require increasing management counter-intervention across the landscape.

In the northern Plains, a region always on the edge of drought stress, future moisture levels represent the single most important climate change parameter. The degree of drying foreseen by our scenarios is sobering, with implications well beyond the issue of island forest survival. The impacts on agriculture, for example, can be expected to be serious. It is beyond the scope of this study to explore these wider issues here. In terms of trees, though, we note that there will be impacts where agriculture and trees have traditionally intersected on the northern Plains – in shelterbelts. Shelterbelts will be both more important than ever on the agricultural plains (owing to their ability to capture snow moisture and shelter fields from desiccating winds) and more difficult to establish and maintain (owing to increasingly arid conditions). Clearly shelterbelt providers and users will need to focus on the most drought-tolerant tree species and shelterbelt structural designs possible. Agroforestry, a more recent economic focus on the northern Plains, will be affected by increased aridity as well. Careful attention to drought tolerance may be necessary to increase the likelihood of eventual successful timber harvest when planting seedlings of a particular species.

(b) Issues and Recommendations

<u>Issue 1:</u> We nowhere found an analysis of how climate change would affect future landscapes of our island forest study sites or how such impacts would or should be managed.

<u>Recommendation 1:</u> *All vegetation management strategies for the island forest study sites should incorporate the probable range of climate change impacts in their analyses and include an analysis of management options to deal with these impacts.*

Issue 2: We found no consideration of the net CO_2 impacts of active or passive vegetation management at our study sites. Strictly speaking, issues of carbon sequestration or release are mitigation issues rather than adaptation issues. However, impacts on CO_2 balances are likely to become increasingly important considerations in future vegetation management everywhere. Grissom et al. (2000) discuss the policy implications with respect to the boreal forest. De Groot et al. (2002) provide estimates of the impact on CO_2 balances of different fire management regimes for Elk Island, Prince Albert, Riding Mountain and Wood Buffalo National Parks.

Recommendation 2: *All vegetation management strategies for the island forest study sites should include an estimation of the net CO_2 impacts of vegetation management alternatives.*

Issue 3: There is little or no ongoing monitoring of biophysical indicators of climate change at our study sites. It is important to monitor the productivity and regeneration success of both native and exotic tree species. Monitoring possible drought stress and studying the incidence and impact of natural disturbances on forest function is also advisable. At some sites monitoring of groundwater levels may be useful. In 2000 the "Climate Change Impacts and Productivity of Aspen in the Western Canadian Interior" study was established in 24 study areas from the Northwest Territories to southern Manitoba to monitor the long term health of aspen across the western Canadian interior (Hogg et al. 2001). It would be advisable to expand this early warning system to include the island forest study sites, in all of which aspen is a dominant or co-dominant component.

Recommendation 3: *Systematic monitoring for climate change impacts on Plains island forests should be implemented. The monitoring should be binational and linked across sites and jurisdictions.*

Issue 4: An anticipatory, rather than a reactive, response to climate change in the island forests will leave us more landscape options and be most cost-effective in the long run. Under increasingly xeric conditions it will be harder (if possible at all) to recreate a forest than to maintain existing tree cover. Even if regeneration from scratch is possible, many dependent species may be lost in the interim. Sound risk management suggests we should not wait for evidence of irreversible forest loss before responding to climate change.

However, we must also incorporate the fact that we cannot yet predict the details of forest responses to climate change, particularly when considering relatively small sites (IPCC Working Group II 1996). As well we must allow for the fact that public preferences as to future landscape options are not yet established. Realistically, neither scientific nor public preference uncertainties will be resolved in the immediate future. We therefore require a practicable interim strategy. In this context, pursuing a "no regrets" strategy as advocated by the U.S. Office of Technology Assessment (1993), where possible at reasonable cost, makes sense. In the context of the island forests, a no regrets conservation strategy would entail maintaining, or more controversially increasing, landscape heterogeneity to increase the probability of ecosystem stability in the face of climate change. Promoting heterogeneity – species and ecosystem diversity, but also successional stage diversity – is a general principle advocated by the U.S. National Assessment Synthesis Team, who also explicitly recommend active management, if necessary, to achieve heterogeneity (Joyce et al. 2001).

Interim model option (c), "managed retreat," detailed in chapter 13, meets the no regrets test. So far as possible it conserves existing diversity. It is also largely reversible – in the unlikely event climate change impacts prove not to be as severe as the scenarios indicate, we can later choose, for example, to allow wildfires to burn, or insect disturbances to run their full course. On the other hand, a passive management approach runs a much higher risk of irreversible forest loss through either slow cumulative, or spectacular disturbance event, decline.

Recommendation 4: *Adopt conservation management model option (c), "managed retreat," as described in this study, as an interim response to climate change impacts to reduce the risk of catastrophic forest loss and to support existing species and ecosystem diversity within the forest. Key specific measures include:*

- *maintain a diversity of age stands in the forest;*

- *manage fuel loads to avoid excessive fuel build-up;*
- *create and maintain fire breaks;*
- *apply prescribed fire where appropriate;*
- *undertake forest harvest where appropriate;*
- *target savannah-like landscapes less vulnerable to catastrophic disturbances, where appropriate;*
- *replant to aid regeneration where necessary;*
- *counter potentially catastrophic insect or vegetation disturbances by biological, chemical or physical controls, if necessary; and,*
- *determine the likely critical constraints on species' survival and determine and implement management to extend the survivability of extant species beyond their normal CMI range.*

Issue 5: Simply trying to maintain existing ecosystem components may not prove an adequate management response within small, isolated ecosystems. The US National Synthesis Team (Joyce et al. 2001) recommends that managers ensure "high levels of connectivity in aquatic and terrestrial systems" in response to climate change, a view supported by Scott and Suffling (2000). Given that island forests are, by definition, isolated from natural tree species migration, any "connectivity" for trees will have to be supplied by management. Managers could supply relatively mild connectivity via, for example, programs to find drought-resistant seed sources of extant forest tree species, including sourcing areas outside the island forests. Breeding of more drought-resistant varieties might also be useful. More radical connectivity could be supplied by the introduction of new tree species to increase ecosystem resiliency. This response to climate change is recommended by Ledig and Kitzmiller (1992) in a commercial forestry context.

Many old experimental plantations containing both native and exotic tree species and varieties can still be seen throughout the northern Plains, both within and outside our study sites. The goal of these provenance experiments was not to test trees for nature conservation purposes, but rather to test their economic potential as shelterbelt trees or as commercial timber. Some experimentation to these ends, particularly for improved shelterbelt trees, continues today. The new Plains trials proposed here would have a different intent – to find or breed species and varieties most suited for ecological salvage purposes under a changing climate.

Recommendation 5: *Undertake a discovery, provenance and breeding program, encompassing both extant tree species within the island forests and possible new species introductions, with the objective of establishing which varieties and species are best adapted to the range of probable future climates in the island forests. Such a program could include:*

- *collection of seed from microsites with particularly low CMIs within and outside the island forests;*
- *determination of related tree species to those now extant which might add resiliency to the island forests;*
- *the use of plantation trial sites within or adjacent to the island forests (where planting on public conservation lands is not considered appropriate, the use of nearby private lands under contract could be considered); and,*
- *the use of plantation trials in low CMI sites outside the island forests where such sites might serve as analogues for future CMI conditions at the island forest sites.*

Issue 6: We have focused on trees as keystone species in this study, but a similar logic applies to other flora and fauna within the island forests. Therefore:

Recommendation 6: *Consider the management options available to preserve fauna and non-tree flora which may be threatened as climate change progresses. Consider the introduction of new species whose habitats are under threat elsewhere or which could partially substitute for disappearing species in the island forests.*

Issue 7: Zoning, which is already employed as a management technique in many protected areas, will be a critical management tool. Zoning can facilitate a differentiated response to climate change and a shift towards multiple target landscapes within a protected area, with some zones proactively managed in response to climate change while other zones are managed passively or more traditionally. It would be reasonable, for example, to leave some stands of old growth forest alone or to let disturbances run their course in some areas, while intervening aggressively in other areas. This approach would promote landscape diversity and ecosystem robustness; possibly be supported by more stakeholders than intensive management throughout the island forest; and also have scientific and monitoring value.

Recommendation 7: *Integrate adaptation to climate change within protected area zoning plans and establish differentiated responses to climate change impacts in appropriate zones.*

Issue 8: The 5 island forests we examined are only a subset of a much wider set of Plains island forests extending to the north and south. We provide an illustrative overview of the entire Plains island forest archipelago in appendix B. Given the probable magnitude of climate change impacts, it is far from ideal to manage each island forest in isolation from the others. For example, an ecosystem or species under increasing threat in one island forest may require additional protection in a second, or be suitable for introduction in a third.

Recommendation 8: *Institute a climate change impacts assessment and management study of the entire Plains island forest archipelago in order to:*

- *establish the bounds of probable climate change at each island forest site;*
- *determine the vulnerability of all Plains island forests to probable climate change impacts;*
- *establish whether climate change risk management and adaptation strategies are in place;*
- *examine whether there is evidence of climate-change-related vegetation shifts;*
- *compare current climate change monitoring approaches and foster knowledge transfer of best practice across all Plains island forests;*
- *determine the utility of a trans-Plains island forest climate change monitoring system;*
- *establish whether particular species assemblages or ecosystems in a given island forest will likely become viable in another;*
- *identify potential seed and species stock for potential inter-island transfer;*
- *compare current management approaches and foster knowledge transfer of best-practice management across all Plains island forests;*
- *compare current climate change consultations and communications practice and foster knowledge transfer of best practice across the island forests; and,*
- *determine the utility of a formalised trans-jurisdictional information network for the island forests.*

Issue 9: Dancik (1992) believes there are only 2 principal concerns around the management of forest lands in North America: proper stewardship and the need for public involvement. Correct process (i.e. public involvement) is now especially critical, as the public has little knowledge of the probable impacts of climate change on the island forests and managers have no certainty as to where, within the bounds of possible future landscapes, the public will wish them to steer. Communications and consultations are therefore critical components of an island forest climate change strategy. A communications strategy would explain to affected and interested stakeholders that climate change cannot mean "business as usual" for nature conservation and it would accustom people to the idea that the island forests landscapes are certain to change. We also need to consult interested and affected people as to their vision for the island forests under climate change constraints. There are several methods of doing this. We could create display areas within the forests of various management activities and outcomes and ask the public to evaluate their acceptability. We could ask the public to evaluate and rate photomontages, or artists' drawings, or computer images of possible landscape futures. This visual survey approach has been successfully employed in British national parks to help determine the public's preferred target landscapes. We could also do standard question-and-answer social surveys. Whatever the consultations method, the central questions are twofold:

- what are the public's preferences among alternative possible future landscapes?
- what management methods to reach the preferred landscapes are acceptable?

Recommendation 9: *Design and undertake a communications and consultations strategy that is integrated across the island forests. The strategy should link, wherever possible, to site examples of monitoring or management for climate change. Existing island forest interpretation centres should interpret site-specific climate change scenarios and impacts, and survey visitor preferences for management responses. The Turtle Mountain International Peace Garden should be investigated as a potential binational Plains "flagship" climate change interpretation centre.*

Issue 10: Increased research and monitoring, and increasingly active management in the island forests, both of which are recommended by this study, imply the need for increased funding. Unfortunately, some of the climate change risks to island forests are difficult to understand and may appear too long term to enable researchers and managers to successfully garner the necessary funding support. But as understanding of the issues and needs grows, support for the necessary funding will increase. A parallel might be the growth of understanding that has slowly built support for the protection of native grassland. Some type of "Save our Prairie Forests" campaign may at some point become appropriate, perhaps accessing funding sources beyond government.

Recommendation 10: *Work to increase the understanding of Plains climate change island forest issues amongst managers, stakeholders, government and the general public, with a view to building the required funding support.*

Issue 11: Climate change of the degree foreseen here can be expected to have a major impact on isolated grassland ecosystems as well as on forest ecosystems. The Cypress Hills, for example, is the site of an island (fescue) grassland, as well as the site of an island forest. Our degree of ignorance around grasslands and climate change is large. It is more difficult to define and identify particular isolated grassland associations of interest and value than it is island forests. We have no certainty about the climate parameters, including moisture, that may define the continuance of particular grassland associations under climate change stress. And we know relatively little about how far and how fast we can expect different native grassland species to migrate to climatically newly favourable sites. Nor do we know whether we would view such migrations as desirable or not.

Recommendation 11: *Work to identify high-conservation value isolated grassland associations across the Plains, establish their probable vulnerability to climate change, and initiate appropriate management responses.*

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APPENDIX A – NOTES ON THE INTRODUCTION OF NEW TREE SPECIES

It is beyond the scope of this study to specify which tree species would be best suited for introduction in each island forest study site, should such introductions be accepted in principle. Nonetheless, we offer here some preliminary observations.

A decision in favour of the experimental introduction of new tree species in the island forests would likely have to occur at the relevant system-wide level first, with discussion on the detail of specific introductions then continuing at a site-specific level. Before or simultaneous to introducing new species there would likely also be a wish to increase regeneration assistance to extant species, or to introduce improved seed stock of extant species, or to import from other areas new varieties of extant species. White spruce, for example, native to Spruce Woods and the Cypress Hills, has a wide range of genetic variability and drought tolerance, and it is possible that seed sources from other areas may be better adapted to future climates.

Partly because we have so altered the pre-European fire regime, we do not know with certainty the viable current ranges of common tree species of the northern Plains region, particularly if we are willing to plant and support seedlings. Hogg (1994) and Wheaton (1997) note the desirability of research on these issues. For conservation purposes, not only is the question of the limits of where a species can survive and regenerate unaided of interest, but also where a species would be viable with various levels of management assistance – such as planting seedlings, managing grass, controlling pests, or protecting against wildfire, for example. One of the major findings of Scott and Suffling's (2000) study of climate change and the Canadian national park system was that there was relatively little knowledge of the sensitivity of individual species in the various national parks to climate change. The same can be said for the trees in our island forest study sites.

Wells (1965) makes the bold claim that "there is no range of climate in the vast grassland province of the central plains of North America which can be described as too arid for all species of trees native to the region." He argues that key restrictions on tree expansion on the Plains have been its flatness and lack of barriers to frequent fire. He accurately points out that some xerophytic junipers can be found in the most arid environments of the southern Plains. Since we can now stop most Plains fires, in theory we should be able to grow a least some tree species almost anywhere (at least under historical climate conditions). However, it cannot be said that broad-leaved trees could be self-sustaining everywhere on the Plains and, in the north, temperature constraints are a factor.

Pielou (1991) notes that there are 2 entirely different mechanisms for cold hardiness in trees for which temperatures of minus 40°C represent a critical boundary. Black spruce, white spruce, tamarack, jack pine, balsam fir, aspen and balsam poplar can survive temperatures below minus 40°C as in these species the sap from cooled tissues freezes harmlessly in intercellular spaces. Ashes, oaks and elms, however, some species of which are reasonably cold-hardy, cannot grow north of the isotherm for minus 40°C minimum temperature, as in these trees the sap forms ice crystals within the cells and thereby kills them (Arris and Eagleson 1989). As climate change raises winter temperatures, perhaps disproportionately (Harvell et al. 2002), some of these species may become viable much farther north, and certainly within the island forests under consideration here.

A key concern must be the positive or negative impacts management could have on the aspen component of the island forests. The importance of aspen for the persistence of many other plant and animal species can hardly be over-emphasised. It is at least conceivable that we could, if necessary, substitute a more drought-tolerant pine for lodgepole pine, or a more drought-tolerant spruce for white spruce, and maintain a recognisably familiar ecosystem at the relevant island forest sites. The collapse of aspen, however, would be very difficult or impossible to compensate for – no other native tree replicates its ecological niche and a large proportion of the existing island forests' flora and fauna is directly or indirectly dependent on aspen's presence.

In some situations aspen appears to be the most drought-tolerant tree on the northern Plains, as it is often found where no other trees are present. Indeed, the dominance of aspen alone, interspersed with grassland, is the definition of the Canadian parkland belt that lies between the northern Plains and the western boreal forest. Aspen benefits from its ability to regenerate by suckering and thereby avoid vulnerability to seedling drought stress. The current aspen distribution pattern no doubt partly reflects aspen's ability to regenerate quickly after the wildfires that were once common on the Plains and in adjacent forests. But conversely, under conditions of fire suppression, other species might benefit in the long term.

There are also xeric sites on the northern Plains where other tree species outperform aspen. For example, bur oak is common on south-facing slopes of the Saskatchewan-Manitoba eastern Qu'Appelle Valley that are too dry for aspen (aspen dominates the more humid north-facing slopes). Similarly, bur oak survives on the driest south-western exposures of the Turtle Mountain formation in North Dakota. Bur oak is also frequently considered a species which benefits from, or is maintained by, fire (NDPRD 1996).

Some mature conifers may survive drought as well or better than mature aspen. In the Cypress Hills, lodgepole pine, once past the seedling stage, seems to be able to grow in drier environments than aspen. For lodgepole regeneration, droughty conditions appear to be the critical constraint at the seedling stage, although rodents and vegetation competition can be important as well (Harris 1994). The same is true of other conifers. Hogg (1994) notes that there is good survival of jack pine, white spruce and Scots pine planted in afforestation projects of the 1930s and 1940s in dry areas of southern Saskatchewan. The absence of natural occurrences of jack pine and white spruce in these areas is likely owing to their inability to regenerate naturally under dry conditions.

Thorpe et al. (2001) note that intentional introduction of a species native to a nearby area may be a useful tool for adaptation to climate change and that such introductions are less likely than Eurasian exotics to be ecologically disruptive. They suggest, for example, that red pine, native to southeastern Manitoba, could be considered as a possible introduction in place of native jack pine in the western boreal forest. Eastern white pine is another possibility. It is likely that the public would favour the introduction of relatively local species before considering more distant exotics.

With respect to our island forest study sites, bur oak, present at Spruce Woods and Turtle Mountain, is an obvious possible introduction at Moose Mountain to increase diversity and forest resiliency in the face of declining CMI values. Bur oak shows great genetic diversity, with some varieties being particularly drought-tolerant. It has been reasonably well studied and is already supplied for habitat and wildlife plantings in Manitoba by AAFC-PFRA (the federal agriculture department). For Spruce Woods, Colorado spruce is a possible introduction, as some varieties have greater drought tolerance than white spruce (Bill Schroeder, AAFC-PFRA, pers. comm. 2002). At Cypress Hills, the range of possible first-order introductions suggested by the nearby Sweet Grass Hills would include Colorado spruce, limber pine, Rocky Mountain juniper and Douglas-fir, all of which could prove more drought-hardy than the current conifers at Cypress.

The natural range of ponderosa pine is to the south or west of all our study sites, but it has excellent drought tolerance – under otherwise favourable site conditions it is found in CMI conditions as low as minus 45. Ponderosa from seed sources in South Dakota and Nebraska currently grows successfully at test stands at the federal (AAFC-PFRA) Shelterbelt Centre at Indian Head, Saskatchewan. It could be considered as an introduction at the Cypress Hills or the Sweet Grass Hills.

A more radical step would be the introduction of conifers where none exist at present. For example, one could consider introducing pine or spruce or Rocky Mountain juniper at Moose Mountain or Turtle Mountain. Equally radically, one could consider increasing deciduous diversity at Cypress Hills or Sweet Grass Hills via the introduction of bur oak, green ash or American elm.

The ultimate northern Plains backstops for drought-hardiness derive from central Asia. Caragana is the most widely employed shelterbelt tree across the northern Plains. Some Asian varieties of Scots pine (such as Mongolian) are not only cold-hardy, but remarkably well adapted to xeric conditions, well able to survive the many dry years and to regenerate in the occasional wet year. Some Asian larch varieties are also very drought-hardy and would likely survive in the island forests as CMIs decline. These distant exotics should be considered *in extremis*, but are likely to be least acceptable from a conservation perspective.

APPENDIX B – ISLAND FORESTS ACROSS THE GREAT PLAINS: PRELIMINARY NOTES AND MAPPING

In this appendix we map the location of island forests throughout the Plains and provide some preliminary information as to the nature of the individual island forests. Primarily we note characteristic extant tree species at each site. Particularly within the United States there are no universally agreed boundaries to the Plains. For the purposes of this exercise we consider the same 10 U.S. states as did the U.S. National Assessment Synthesis Team in its climate change impacts summary (Joyce et al. 2001), together with the grassland ecoregion of the 3 Canadian "prairie provinces." Neither the maps nor the notes are definitive; they are intended to serve as an overview of the Plains island forest archipelago.

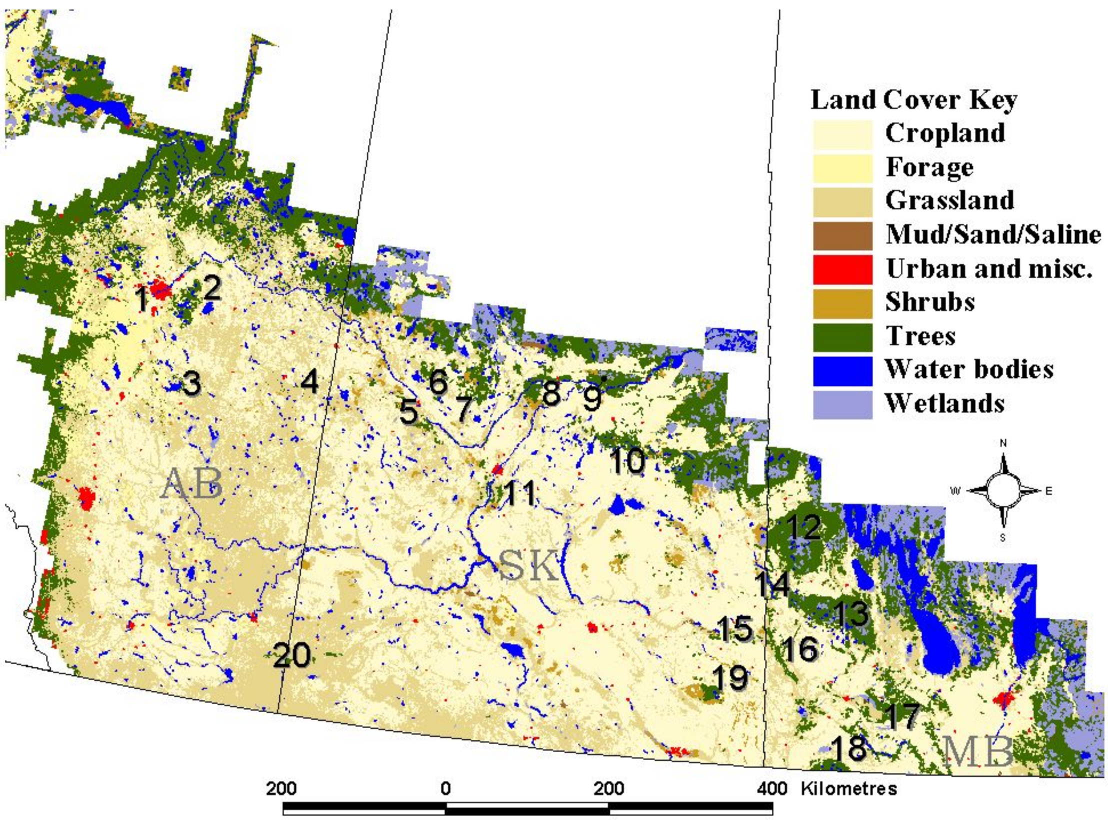


Figure B1: Island Forests of the Canadian Plains
(GIS data from: Ashton, J. PFRA's Generalized Landcover, Version 1. 2001.)

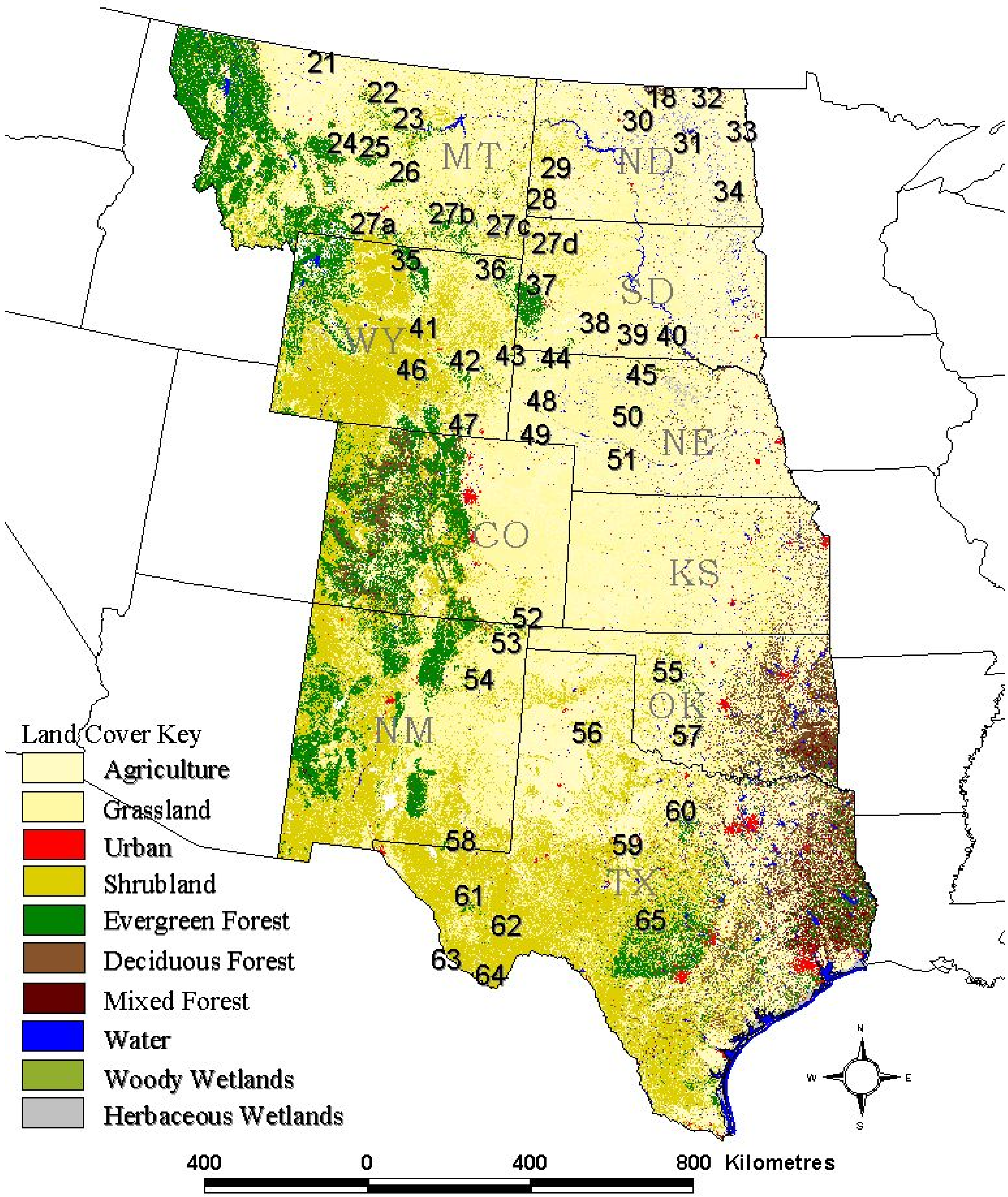


Figure B2: Island Forests of the American Plains

(GIS data from: USGS. National Land Cover Data. 1992.)

Alberta

1 Hasse Lake Municipal Park
- aspen dominant; some white spruce and black spruce
- former provincial park; very small

2 Elk Island National Park
- aspen dominant; some boreal forest elements, black spruce bogs
- 194 km^2
- forest also extends south of the national park

3 Buffalo Lake and Rochon Sands Provincial Park
- aspen dominant; some white spruce and black spruce

4 C.F.B. Wainwright and Dilberry Lake Provincial Park
- aspen dominant; some spruce
- aspen study site present at the Wainwright Dunes Ecological Reserve

20 Cypress Hills Interprovincial Park
- aspen and lodgepole pine co-dominant; widespread white spruce
- see this study for details

Saskatchewan

5 Missouri Coteau along the south shore of the North Saskatchewan River
- isolated aspen woodland running northwest to southeast along river
- overlap with Poundmaker, Little Pine, Sweetgrass, Mosquito, Grizzly Bear's Head and the Lean Man, and Red Pheasant Indian Reserves
- aspen study site present in this area

6 Battlefords Provincial Park and Moosomin and Saulteaux Indian Reserves
- aspen dominant; introduced conifers and caragana

7 Lucky Man Indian Reserve and the Thickwood Hills
- aspen forest; pockets of spruce

8 Prince Albert Forest / Nisbet Crown forest
- north of the North Saskatchewan River: boreal; white spruce, jack pine, white birch, aspen, balsam poplar
- south of the North Saskatchewan River: parkland; aspen dominant; some white spruce and jack pine
- some conversion of pine to grassland may be ongoing
- forest extends south along the South Saskatchewan River

9 Fort à la Corne forest and James Smith Indian Reserve
- white spruce, jack pine

10 Greenwater Provincial Park, Porcupine Hills and Pasquia Hills
- boreal; aspen and white spruce
- highland forests
- to the east becomes contiguous forest

11 Pike Lake Provincial Park, White Cap Indian Reserve and CFB Dundurn

- aspen dominant
- riparian/valley woodlands
- Scots pine and jack pine plantations at CFB Dundurn
- aspen study site present at Dundurn AAFC-PFRA community pasture

12 Duck Mountain (Manitoba and Saskatchewan)
- boreal: aspen, white birch, jack pine, white spruce, black spruce, tamarack, balsam fir
- highland forest

15 Qu'Appelle River
- aspen dominant; Manitoba maple, green ash, balsam poplar, bur oak in eastern valley
- riparian/valley forest

19 Moose Mountain Provincial Park
- aspen dominant
- see this study for details

20 Cypress Hills Interprovincial Park
- aspen and lodgepole pine co-dominant; widespread white spruce
- see this study for details

Manitoba

12 Duck Mountain (Manitoba and Saskatchewan)
- boreal: aspen, white birch, jack pine, white spruce, black spruce, tamarack, balsam fir
- highland forest

13 Riding Mountain National Park
- boreal: aspen, balsam poplar, white birch, green ash, Manitoba maple, bur oak, American elm, white spruce, black spruce, balsam fir, jack pine, tamarack
- prominent highland forest

14 Asessippi Provincial Park (Lake of the Prairies)
- aspen dominant; cottonwood, birch, balsam poplar
- riparian/valley forest

16 Assiniboine River
- aspen dominant; cottonwood, birch, balsam poplar
- riparian/valley forest

17 Spruce Woods Provincial Park and Provincial Forest
- aspen dominant with various hardwoods and white spruce
- riparian/valley forest
- see this study for details

18 Turtle Mountain Provincial Park
- aspen dominant with various hardwoods
- highland forest
- see this study for details

Montana

21 Sweet Grass Hills
- Douglas-fir, subalpine fir, limber pine, whitebark pine, lodgepole pine, hybrid spruce, Rocky Mountain juniper, aspen
- prominent highland forest
- see this study for details

22 Bear Paw Mountains (Rocky Boys Indian Reservation and Beaver Creek Park) and Little Rockies (Fort Belknap Indian Reservation, BLM land, Zortman mine)
- Bear Paw Mountains: aspen, Manitoba maple, cottonwood, ponderosa pine, lodgepole pine, Douglas-fir
- Little Rockies: aspen, white birch, ponderosa pine, lodgepole pine, Douglas-fir
- highland forests

23 Missouri River Valley
- gallery cottonwood forests undergoing conversion to aspen, ash, elm and oak
- parts contained in Russell National Wildlife Refuge
- riparian/valley forest

24 Lewis & Clark National Forest and Gallatin National Forest (Highwood Mountains, Little Belt Mountains, Castle Mountain, Big Snowy Mountains, Crazy Mountains)
- limber pine, ponderosa pine, Douglas-fir, lodgepole pine, Engelmann spruce, subalpine fir, whitebark pine, aspen, cottonwood, Rocky Mountain maple
- highland forests; Rocky Mountain outliers

25 Judith Mountains
- Douglas-fir, ponderosa pine, lodgepole pine
- Roughly 40% BLM ownership in the higher elevations (including the Judith Mountain Recreation Area); private ownership in the lower elevations
- highland forest

26 Bull Mountains
- ponderosa pine dominant; small amounts of Douglas-fir, Rocky Mountain juniper, stunted aspen
- highland forest

27 Custer National Forest
(a) Beartooth district (2,230 km^2): Douglas-fir, subalpine fir, Engelmann spruce, whitebark pine, limber pine, lodgepole pine, aspen, green ash, American elm
(b) Ashland district (1,620 km^2): ponderosa pine, green ash
(c) part of Sioux district (3,930 km^2): ponderosa pine, green ash, Manitoba maple, aspen
(d) see under South Dakota, Sioux district

North Dakota

18 Turtle Mountain
- aspen dominant with various hardwoods
- highland forest
- see this study for details

28 Little Missouri River Forest
- green ash dominant; American elm, oak, cottonwood along river

- riparian/valley forest

29 Little Missouri National Grasslands
- ponderosa pine and limber pine savannah in southern grasslands; Rocky Mountain juniper in northern grasslands; green ash, American elm, bur oak, cottonwood
- most northeasterly natural occurrence of ponderosa pine in North America

30 Souris River
- green ash, Manitoba maple, American elm, aspen, cottonwood, (exotic) Russian olive; small amounts of bur oak
- management concerns include the expansion of woodlands, Kentucky bluegrass and brome grass, and the presence of Dutch Elm Disease

31 Devil's Lake
- aspen, oak, ash, hackberry, basswood
- fire-shadow effect of lake has promoted forest growth

32 Pembina Gorge
- aspen, bur oak

33 Turtle River State Park
- basswood, green ash, Manitoba maple, aspen, cottonwood, American elm, bur oak
- riparian forest with some bur oak savannah on uplands

34 Sheyenne National Grassland Forest
- basswood, green ash, Manitoba maple, cottonwood, aspen,
- some bur oak savannah
- managed for wildlife, recreation and occasionally industry

Wyoming

35 Big Horn National Forest
- 60% forested, of which 50% is lodgepole pine and 25% is Engelmann spruce; also present are subalpine fir, Douglas-fir, ponderosa pine, limber pine, Rocky Mountain juniper, aspen, cottonwood

36 Thunder Basin National Grassland and forest along the Little Powder River
- ponderosa pine, Rocky Mountain juniper, cottonwood
- riparian forest

37 Black Hills National Forest (Wyoming): see under South Dakota

41 Granite Mountain
- lodgepole pine

42 Laramie Mountains (Medicine Bow National Forest)
- ponderosa pine, lodgepole pine, subalpine fir, aspen

43 Pine Ridge Escarpment (Wyoming): see under Nebraska

46 Northern edge of the Great Divide Basin
- lodgepole pine stands run west to east, south of the Sweetwater River

- BLM land

47 Part of Medicine Bow National Forest and Hutton Lake National Wildlife Refuge
- ponderosa pine, possibly some lodgepole pine, some aspen

South Dakota

27 Custer National Forest
(d) part of Sioux district (28,400 ha): ponderosa pine, American elm, green ash

37 Black Hills National Forest
- ponderosa pine, white spruce, aspen, white birch, bur oak, Rocky Mountain juniper, lodgepole pine, Douglas-fir, green ash, American elm, eastern hophornbeam
- a large highland forest (456,000 forested hectares)
- a local variety of white spruce exists: "Black Hills spruce"

38 Buffalo Gap National Grassland (Pine Ridge Escarpment and White River)
- ponderosa pine, Rocky Mountain juniper, green ash, Manitoba maple, hackberry

39 Rosebud Indian Reservation (Pine Ridge Escarpment and the South Fork of the White River)
- ponderosa pine, Rocky Mountain juniper, green ash, Manitoba maple, hackberry

40 Missouri River forest
- eastern redcedar on bluffs west of the Missouri River
- mostly private land; some federal land along Missouri River reservoir edge

44 Pine Ridge Indian Reservation (Pine Ridge Escarpment)
- ponderosa pine, Rocky Mountain juniper, green ash, Manitoba maple, hackberry

Nebraska

43 Ogala National Grassland (Pine Ridge Escarpment)
- ponderosa pine, Rocky Mountain juniper, green ash, Manitoba maple, hackberry

44 Nebraska National Forest, Fort Robinson State Park and Pine Ridge National Recreation Area (Pine Ridge Escarpment)
- ponderosa pine, Rocky Mountain juniper, green ash, Manitoba maple, hackberry

45 McKelvie National Forest, Fort Niobrara National Wildlife Refuge and Valentine National Wildlife Refuge
- ponderosa pine, eastern cottonwood, green ash, Manitoba maple, hackberry; small amounts of aspen
- riparian forest along the Niobrara River

48 Wildcat Hills and Scott's Bluff
- ponderosa pine, Rocky Mountain juniper
- Scott's Bluff is on the south bank of the North Platte River, almost contiguous with the Wildcat Hills
- scarp forests

49 Cheyenne Break Escarpment
- ponderosa pine, Rocky Mountain juniper
- forest is generally savannah-like, but thick on north slopes

50 Nebraska National Forest (Sand Hills)
- eastern cottonwood, green ash, Manitoba maple, hackberry; small amounts of aspen
- conifers not native to area, but ponderosa pine, jack pine and Scots pine plantations dating from the early 1900s are all doing well

51 East of the forks of the Platte River
- elm, ash, cottonwood

Colorado

52 Black Mesa de Maya Escarpment
- Colorado pinyon, ponderosa pine, one-seed juniper, Rocky Mountain juniper, wavyleaf oak

Oklahoma

52 Black Mesa de Maya Escarpment
- Colorado pinyon, ponderosa pine, one-seed juniper, Rocky Mountain juniper, wavyleaf oak

55 Parallel escarpments
- post oak and blackjack oak dominant
- northern escarpment runs along the North Canadian River; southern escarpment along the South Canadian River

57 Wichita Mountains National Wildlife Refuge
- post oak, blackjack oak, eastern redcedar, Rocky Mountain juniper, isolated sugar maple
- near the edge of the eastern deciduous forest

New Mexico

52 Black Mesa de Maya Escarpment
- Colorado pinyon, ponderosa pine, one-seed juniper, Rocky Mountain juniper, wavyleaf oak

53 Kiowa National Grassland (Canadian Escarpment)
- Colorado pinyon, one-seed juniper, Rocky Mountain juniper, wavyleaf oak, ponderosa pine, cottonwood, (netleaf?) hackberry
- riparian and escarpment woodlands

54 Canadian Escarpment
- ponderosa pine with small numbers of Rocky Mountain juniper dominant on north slopes; one-seed juniper dominant on south-facing slopes

58 Guadalupe Mountains (Lincoln National Forest)
- Colorado pinyon and juniper are dominant; oak, pine, fir, Douglas-fir, Engelmann spruce, corkbark fir, ponderosa pine, aspen, Texas madrone
- deep canyons and sheer cliffs topography

Texas

56 Northeastern edge of the Llano Estacado
- Pinchot juniper dominant

58 Guadalupe Mountains (Guadalupe Mountains National Park)
- see under New Mexico

59 Callahan Divide
- live oak, Shumard oak, Mohr oak, Ashe juniper, Pinchot juniper

60 Possum Kingdom Reserve
- live oak, Ashe juniper, mesquite

61 Davis Mountains (Fort Davis National Heritage Site)
- gray oak, alligator juniper

62 Santiago Mountains / Cathedral Mountain
- gray oak, alligator juniper

63 Chinati Peak
- gray oak, alligator juniper

64 Chisos Mountains / Emory Peak (Big Bend National Park)
- gray oak, alligator juniper

65 Edwards Plateau
- on north-facing slopes: Lacey oak, Ashe juniper, Texas persimmon, black cherry, red buckeye, black walnut, Texas ash
- dominant in upland areas: Ashe juniper, live oak, Texas persimmon
- in riparian areas: Ashe juniper, cedar elm, American sycamore, Texas persimmon, baldcypress, pecan
- site of Lyndon B. Johnson National Historic Park
- escarpment and riparian forest elements

APPENDIX C – SCIENTIFIC NAMES

Trees

limber pine *Pinus flexilis*
whitebark pine *Pinus albicaulis*
jack pine *Pinus banksiana*
red pine *Pinus resinosa*
lodgepole pine *Pinus contorta*
Scots pine *Pinus sylvestris*
Austrian pine *Pinus nigra*
ponderosa pine *Pinus ponderosa*
eastern white pine *Pinus strobus*
Colorado pinyon *Pinus edulis*
tamarack *Larix laricina*
Siberian larch *Larix sibirica*
black spruce *Picea mariana*
Colorado (blue) spruce *Picea pungens*
white spruce *Picea glauca*
Black Hills (or Blackhill) spruce *Picea glauca* var. *albertiana*
Norway spruce *Picea abies*
Douglas-fir *Pseudotsuga menziesii*
subalpine fir *Abies lasiocarpa*
corkbark fir *Abies lasiocarpa* var. *arizonica*
balsam fir *Abies balsamea*
baldcypress *Taxodium distichum*
northern white-cedar *Thuja occidentalis*
Rocky Mountain juniper *Juniperus scopulorum*
eastern redcedar *Juniperus virginiana*
one-seed juniper *Juniperus monosperma*
Pinchot juniper *Juniperus pinchotii*
Ashe juniper *Juniperus ashei*
alligator juniper *Juniperus deppeana*

aspen (poplar) *Populus tremuloides*
balsam poplar *Populus balsamifera*
eastern cottonwood *Populus deltoides*
narrow-leaved cottonwood *Populus angustifolia*
willows *Salix* spp.
black walnut *Juglans nigra*
pecan *Carya illinoensis*
white birch *Betula papyrifera*
eastern hophornbeam *Ostrya virginiana*
bur oak *Quercus macrocarpa*
post oak *Quercus stellata*
blackjack oak *Quercus marilandica*
wavyleaf oak *Quercus undulata*
live oak *Quercus virginiana*
Shumard oak *Quercus shumardii*
Mohr oak *Quercus mohriana*
Lacey oak *Quercus glaucoides*

gray oak *Quercus grisea*
American elm *Ulmus americana*
cedar elm *Ulmus crassifolia*
hackberry *Celtis occidentalis*
netleaf hackberry *Celtis reticulata*
American sycamore *Platanus occidentalis*
Greene mountain-ash *Sorbus scopulina*
black cherry *Prunus serotina*
mesquite *Prosopis* spp.
caragana *Caragana arborescens*
Rocky Mountain maple *Acer glabrum*
Douglas maple *Acer glabrum* var. *douglasii*
sugar maple *Acer saccharum*
Manitoba maple *Acer negundo*
red buckeye *Aesculus pavia*
basswood *Tilia americana*
Texas madrone *Arbutus texana*
Texas persimmon *Diospyros texana*
green ash *Fraxinus pennsylvanica*
black ash *Fraxinus nigra*
Texas ash *Fraxinus texensis*

Grasses

brome grass *Bromus* spp.
Kentucky bluegrass *Poa pratensis*
rough fescue *Festuca scabrella*
bluebunch fescue *Festuca idahoensis*
wild oat grass *Danthonia intermedia*
awned wheatgrass *Agropyron subsecundum*
western wheatgrass *Agropyron smithii*
northern wheatgrass *Agropyron dasystachyum*
June grass *Koeleria gracilis*
mat muhly *Muhlenbergia richardsonis*

Other Flora

beaked hazelnut (hazel) *Corylus cornuta*
chokecherry *Prunus virginiana*
saskatoon *Amelanchier alnifolia*
red-osier dogwood *Cornus stolonifera*
rose *Rosa* spp.
leafy spurge *Euphorbia esula*
dwarf mistletoe *Arceuthobium pusillum*
creeping juniper *Juniperus horizontalis*
thin-leaved snowberry *Symphoricarpos albus*
western snowberry *Symphoricarpos occidentalis*
thimbleberry *Rubus parviflorus*
heart-leaved arnica *Arnica cordifolia*
dwarf birch *Betula nana*
shrubby cinquefoil *Potentilla fruticosa*

silvery lupine *Lupinus argentea*
sagebrush *Artemisia cana*

Mammals

white-tailed deer *Odocoileus virginianus*
mule deer *Odocoileus hemionus*
bison *Bison bison*
elk *Cervus elephas*
moose *Alces alces*
porcupine *Erethizon dorsatum*
pine marten *Martes americana abietinodies*
dwarf shrew *Sorex nanus*
northern water shrew *Sorex palustris*
heather vole *Phenacomys intermedius*

Herptiles

western hognose snake *Heterodon nasicus*
red-sided garter snake *Thamnophis sirtalis*
northern prairie skink *Eumeces septentrionalis*
plains spadefoot toad *Scaphiopus bombifrons*
tiger salamander *Ambystoma tigrina*

Birds

wood duck *Aix sponsa*
peregrine falcon *Falco peregrinus*
ruffed grouse *Bonasa umbellus*
wild turkey *Meleagris gallopavo*
bluebird *Sialia* spp.

Fish

rainbow trout *Salmo gairdneri*

Insects

mountain pine beetle *Dendroctonus ponderosae*
forest tent caterpillar *Malacosoma disstria*
jack pine budworm *Choristoneura pinus*
spruce budworm *Choristoneura fumiferana*

APPENDIX D – ANNUAL MAXIMUM AND MINIMUM TEMPERATURE AND PRECIPITATION CHANGE OVER THE ISLAND FOREST STUDY REGION

(a)

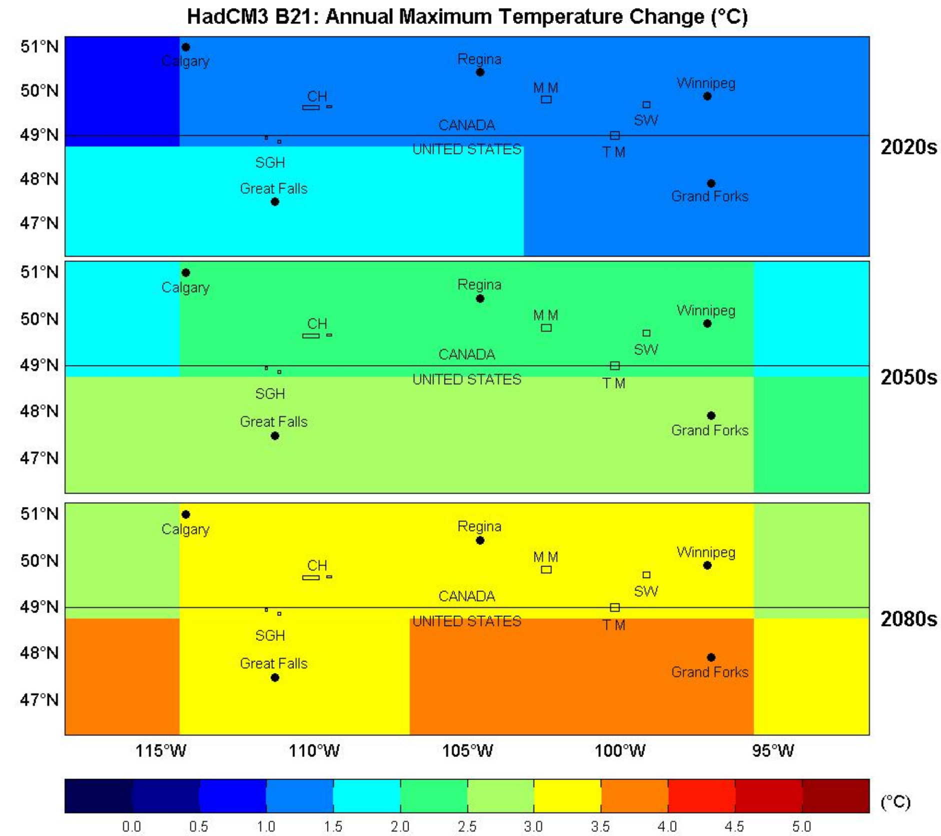
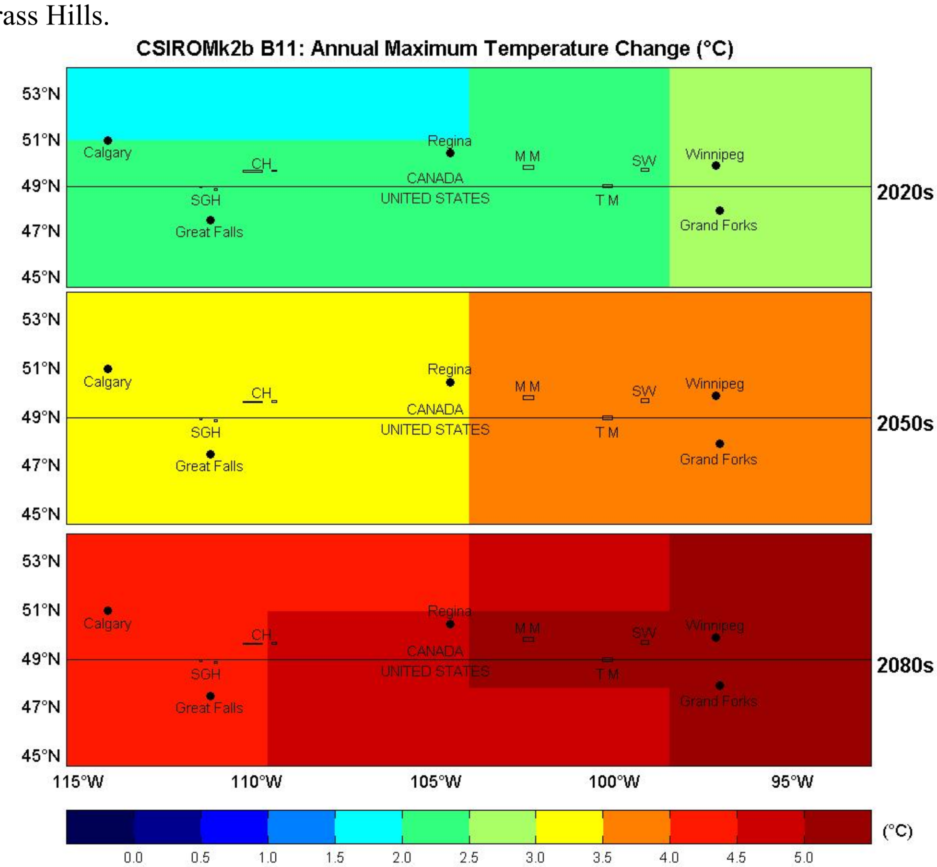


Figure D1: Annual maximum temperature change (°C) for (a) HadCM3 B21 (cool-wet) scenario and (b) CSIROMk2b B11 (mid-range) scenario for the 2020s, 2050s and 2080s. The island forest locations are: SW – Spruce Woods; TM – Turtle Mountain; MM – Moose Mountain; CH – Cypress Hills; SGH – Sweet Grass Hills.



(c)

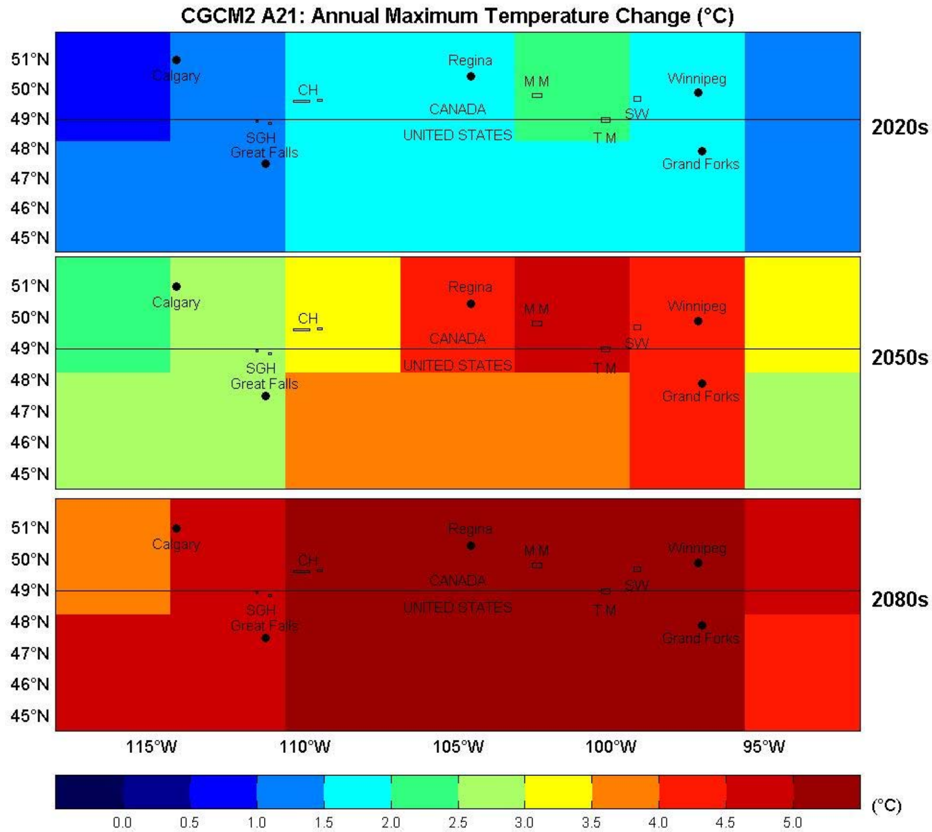


Figure D1 continued: Annual maximum temperature change (°C) for (c) CGCM2 A21 (warm-dry) scenario for the 2020s, 2050s and 2080s. The island forest locations are: SW – Spruce Woods; TM – Turtle Mountain; MM – Moose Mountain; CH – Cypress Hills; SGH – Sweet Grass Hills.

(a)

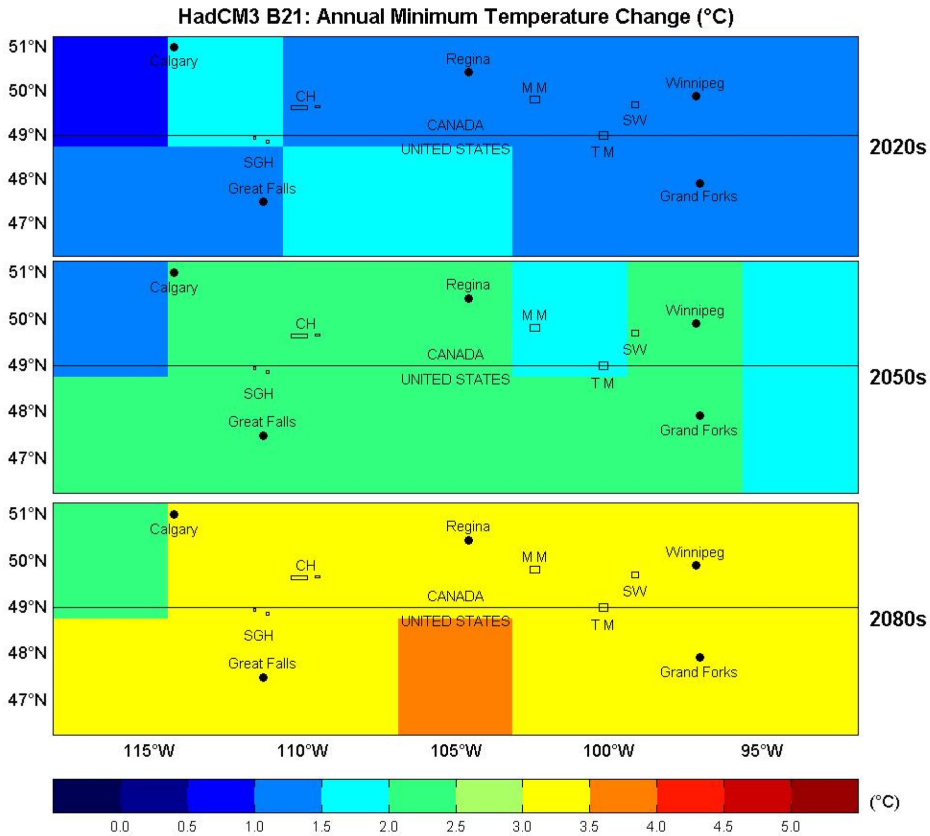
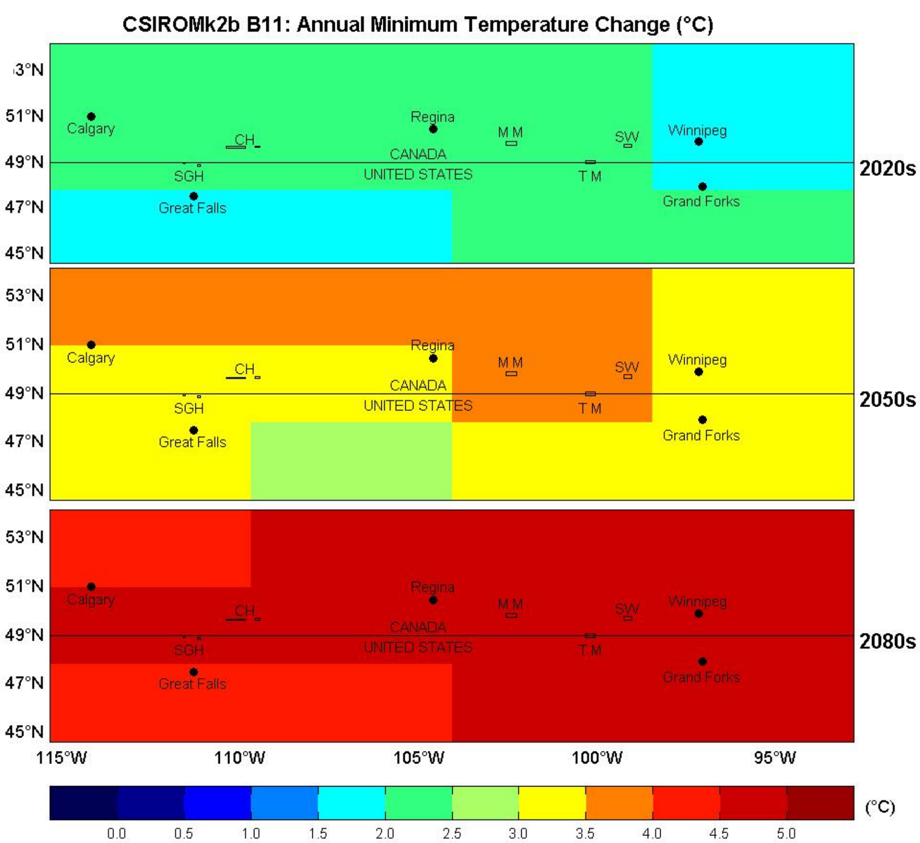


Figure D2: Annual minimum temperature change (°C) for (a) HadCM3 B21 (cool-wet) scenario and (b) CSIROMk2b B11 (mid-range) scenario for the 2020s, 2050s and 2080s. The island forest locations are: SW – Spruce Woods; TM – Turtle Mountain; MM – Moose Mountain; CH – Cypress Hills; SGH – Sweet Grass Hills.

(b)



(c)

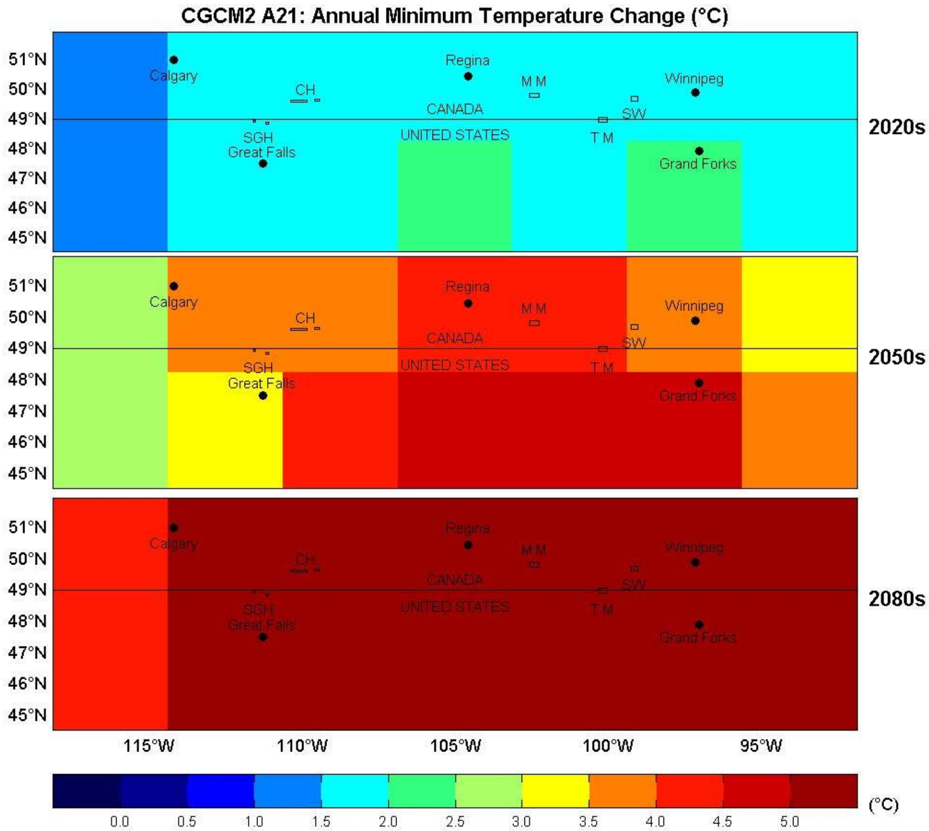


Figure D2 continued: Annual minimum temperature change (°C) for (c) CGCM2 A21 (warm-dry) scenario for the 2020s, 2050s and 2080s. The island forest locations are: SW – Spruce Woods; TM – Turtle Mountain; MM – Moose Mountain; CH – Cypress Hills; SGH – Sweet Grass Hills.

(a)

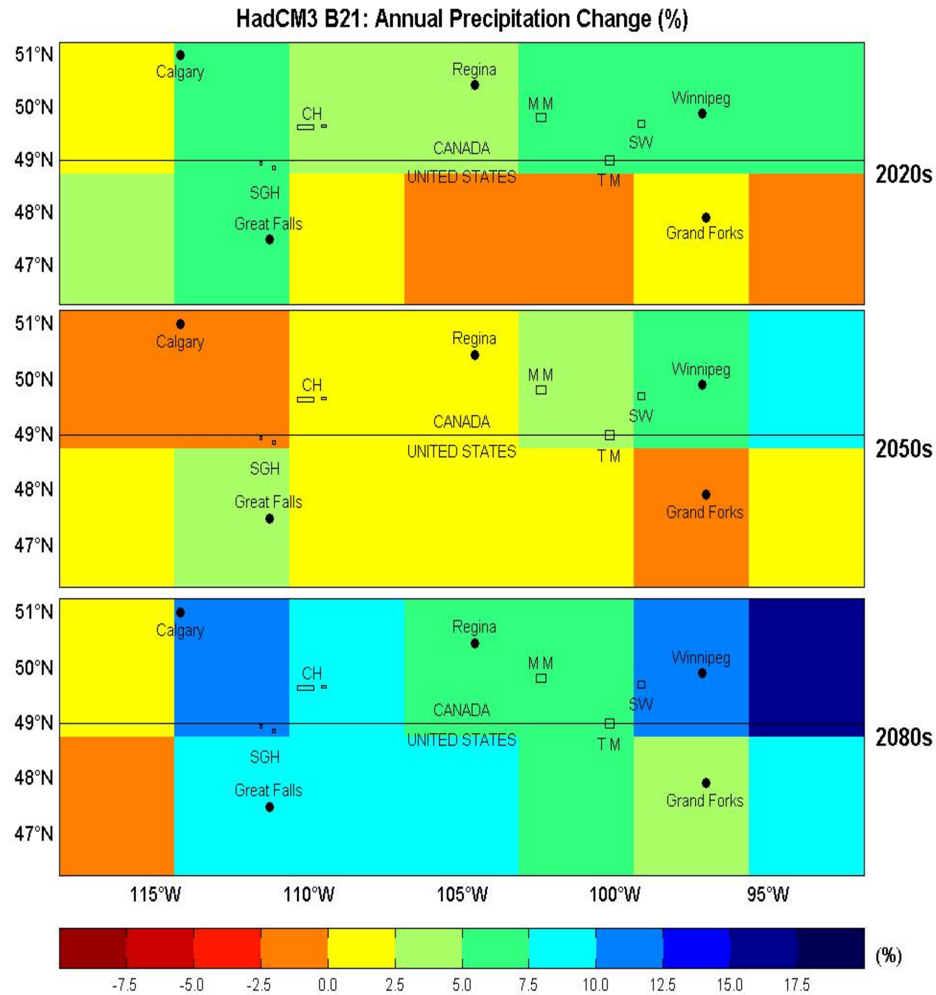
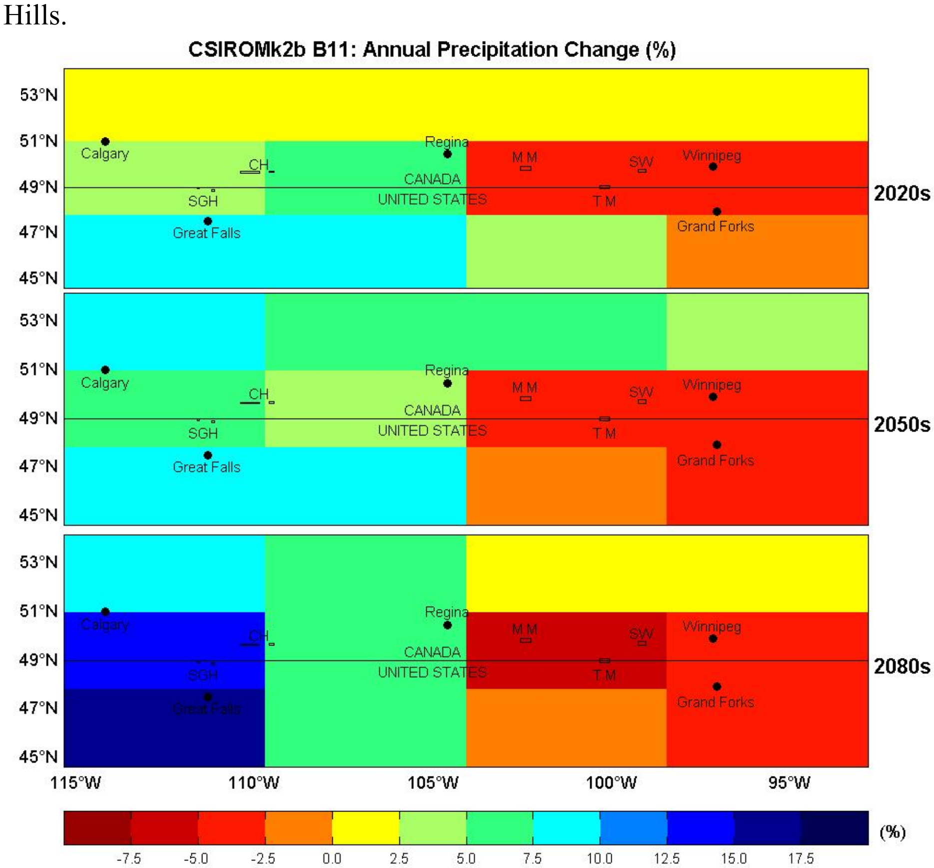


Figure D3: Annual precipitation change (%) for (a) HadCM3 B21 (cool-wet) scenario and (b) CSIROMk2b B11 (mid-range) scenario for the 2020s, 2050s and 2080s. The island forest locations are: SW – Spruce Woods; TM – Turtle Mountain; MM – Moose Mountain; CH – Cypress Hills; SGH – Sweet Grass Hills.

(b)



(c)

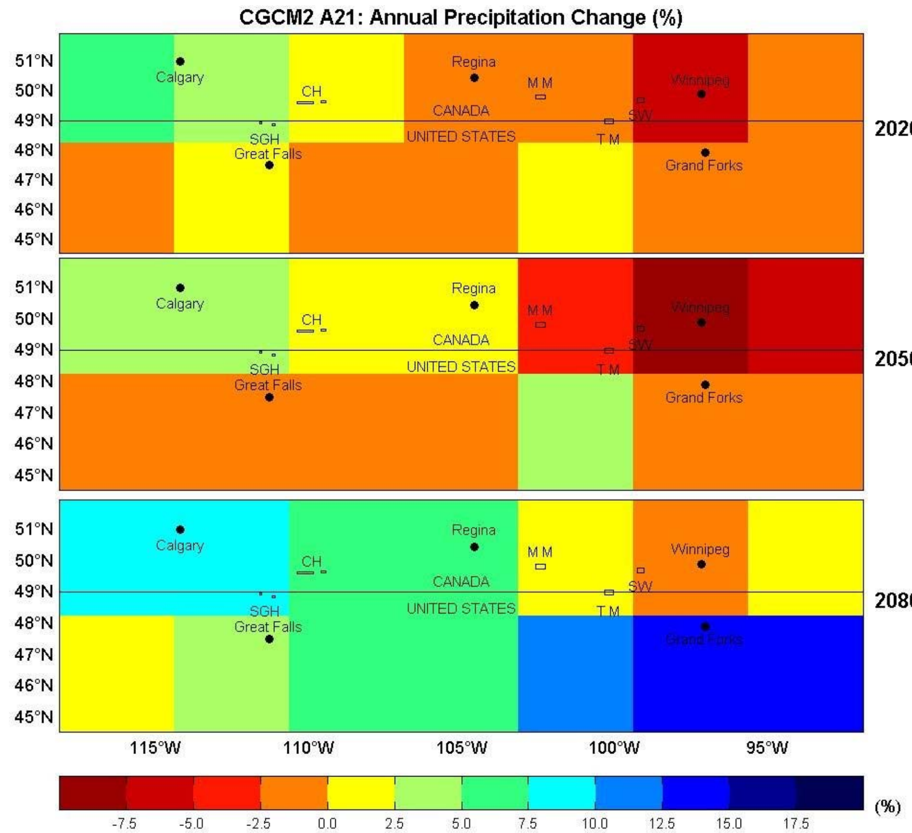


Figure D3 continued: Annual precipitation change (%) for (c) CGCM2 A21 (warm-dry) scenario for the 2020s, 2050s and 2080s. The island forest locations are: SW – Spruce Woods; TM – Turtle Mountain; MM – Moose Mountain; CH – Cypress Hills; SGH – Sweet Grass Hills.

APPENDIX E – OBSERVED DATA SOURCES AND DATA ANOMALIES

Observed 1961-1990 climate normals for Montana, North Dakota and Minnesota were obtained from the Western, the High Plains and the Midwestern Regional Climate Centers, respectively. Canadian station data were obtained from Environment Canada.

Analysis of station data identified problems with Montana maximum temperature data and with U.S. precipitation information (see Figures E1 and E2).

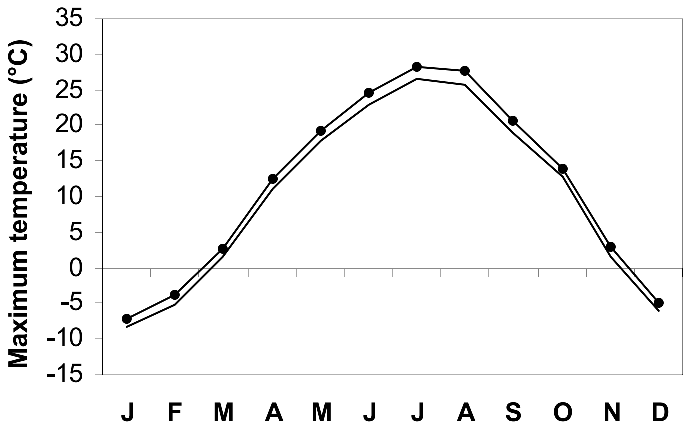


Figure E1: Monthly mean maximum temperature (°C) for the 1961-1990 climate normal period for Opheim, Montana (-•-) and West Poplar River, Saskatchewan (—).

West Poplar River (Saskatchewan) and Opheim (Montana) are adjacent stations, both located at 49.0°N, 106.4°W, with elevations of 876 metres and 908 metres, respectively. The slight difference in elevation is not sufficient to account for the observed differences in maximum temperature at these 2 sites (which would be expected to be about 0.1°C given the 32 metre difference in elevation). The differences illustrated in Figure E1 are almost certainly due to differences in observing practice between Montana and Canada (Ron Hopkinson, Environment Canada, pers. comm. 2002; Jim Ashby and Kelly Redmond, Western Regional Climate Center, pers. comm. 2002). Differences in observing practice lead to a time of observation bias at many U.S. stations (Karl et al. 1986). At most Canadian stations observations are taken at least twice daily, while at many stations in Montana only one daily observation is recorded at about 5:00 pm local time. Immediately after this late afternoon observation, the maximum temperature instrumentation is reset. The maximum temperature thermometer will then register the temperature as of 5:01 pm. If the afternoon is warm, the reading will be correspondingly high. If temperatures throughout the next day are cooler than the temperature as of 5:01 pm the previous afternoon, when the next day's reading is taken the previous afternoon's temperature will be erroneously read as the daily high. For example, if the high on August 5 is 34°C, and this temperature is reached at about 5:00 pm that day, 34°C will be recorded as the high for August 5. If the temperature now falls overnight, and then rises next morning to reach a high of 28°C in the afternoon of August 6, a high of 34°C will nonetheless again be recorded for August 6, as the thermometer will not have been reset since 5:00 pm on August 5. This explains the illustrated 1 to 2°C difference in maximum temperature between West Poplar River and Opheim stations. The problem appears to lie with recording practice in Montana but may possibly also apply in North Dakota. To correct for this anomaly would be a substantial task and would require access to both U.S. and Canadian daily station records so that station intercomparisons could be made and corrections calculated.

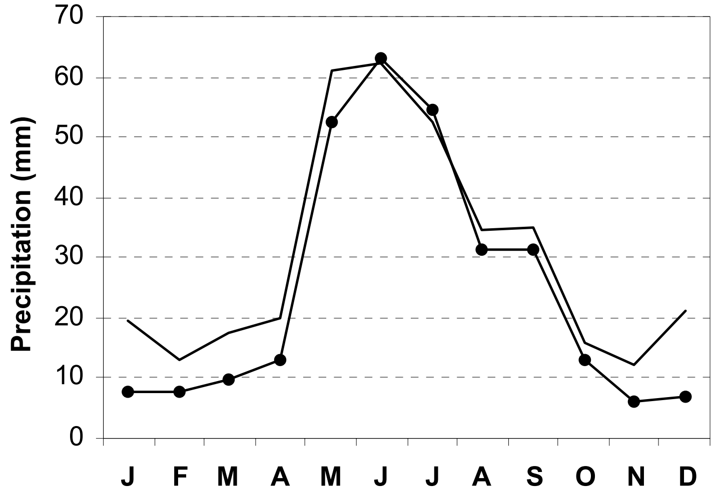


Figure E2: Monthly average total precipitation (mm) for the 1961-1990 climate normal period at Opheim, Montana (-●-) and West Poplar River, Saskatchewan (—). Note the differences in precipitation amount in the winter months.

A second anomaly is the fact U.S. precipitation values are generally lower than those recorded in Canada, particularly in winter (see Figure E2). This is most likely due to differences in snow measurement during the winter half of the year. In Canada the majority of weather stations employ a Nipher wind shield on the snow gauges which reduces gauge under-catch in windy weather.

It is difficult to determine the exact effect of these 2 data problems on CMI values. Certainly the indicated CMI values are slightly (for North Dakota and Minnesota) to somewhat (for Montana) too dry in the U.S. as a result of these problems. If the relationships between temperature, precipitation and CMI are linear, then we can confidently say that the change in CMI seen as a result of applying the different climate change scenarios is correct, but that the baseline (1961-1990) CMI values are too low. If, however, the relationships are non-linear, then it is more difficult to determine the effects on CMI, both in the current climate and in any of the future climate change scenarios considered here. However, for practical purposes, as the Sweet Grass Hills (Montana) and Turtle Mountain (North Dakota) study sites are adjacent to the international boundary, we can be reasonably confident that the climate normals and the magnitude of change indicated by the climate scenarios will not differ significantly south of the boundary from that shown just north of the boundary.

APPENDIX F – CLIMATE STATION INTERPOLATION

With the exception of one climate station in the Centre Block of the Cypress Hills (which in fact contains climate data only over part of the 1961-1990 normal period), climate stations are absent from our study sites. It was therefore necessary to calculate the climate values we would expect that a climate station within each study site would have recorded over our 1961-1990 normal period, had such a climate station existed. The process of constructing a set of data for an imaginary climate station is termed interpolation. Interpolation can be done with reasonable confidence if there are an adequate number of nearby stations. In general terms one interpolates by averaging recorded climate data of nearby climate stations while taking care to give greatest statistical weight to those stations closest to the interpolated station. It is also necessary to allow for the effect of elevation, as temperatures are cooler and precipitation is greater at higher elevations. It follows that the CMI also rises with elevation.

We sited our interpolated stations at the geographic centre of each of our island forest study sites. In the case of the Cypress Hills we interpolated 2 stations, one centred in Centre Block, and another centred in West Block. Equally, for the Sweet Grass Hills we also interpolated 2 stations, one for West Butte and one for East Butte.

Using climate normal data from climate stations surrounding each study site we determined a partial regression coefficient for elevation by multiple linear regression analysis of CMI, with elevation, latitude and longitude as the independent variables. The resulting partial regression coefficients (termed "elevation coefficients") are shown in Table F1. They indicate the relevant increase in CMI value, expressed in centimetres of water per year, per 100 metre increase in elevation at the relevant site.

Table F1: Estimated relationships between elevation and CMI for the study sites

Island forest site	Number of surrounding stations considered	Range of station latitudes (°N)	Range of station longitudes (°W)	Range of station elevations in metres	Elevation coefficient*
SGH and CH	30	48 - 50	108 - 112.5	713 - 1292	5.8
Spruce Woods	28	49 - 51	98 - 101	248 - 756	4.1
Moose Mountain	20	49 - 51	101 - 104	469 - 671	3.7*
Turtle Mountain	39	48 - 50	98 - 102	261 - 597	1.8*

*in practice these two coefficients not used; see discussion immediately following

The small values of the coefficients obtained for the areas surrounding Moose Mountain and Turtle Mountain are likely to be underestimates resulting from the relatively narrow range of elevation amongst the surrounding climate stations. We instead used the Spruce Woods coefficient (4.1) for interpolation of CMIs at Moose Mountain and Turtle Mountain. Spruce Woods has a greater elevation range in its surrounding climate stations and is geographically not too distant from either Moose Mountain or Turtle Mountain. Also, in previous analyses over 94 climate stations in northern Alberta, an elevation coefficient of 4.5 was determined (Hogg 1994), a figure close to that of Spruce Woods. Finally, using a larger elevation coefficient results in higher CMI values both in the 1961-1990 normals and in all scenarios. It is therefore conservative practice to use the higher value coefficient – using the lower value coefficients of Table F1 would produce marginally more xeric climate normals and marginally more xeric (and discouraging) scenarios for Moose Mountain and Turtle Mountain.

On the other hand, the calculated elevation coefficient of 5.8 for the Cypress Hills and the Sweet Grass Hills (these 2 sites are quite close to each other) may be an overestimate, particularly so in the case of the Sweet Grass Hills. This is partly because of the unusually high wind speeds and accompanying desiccation that the 2 sites are exposed to, an effect that would be most pronounced around the peaks of the Sweet Grass Hills. The calculation of the elevation coefficient also assumes that elevation alone generates increased precipitation. This is not strictly true, for the mass of the elevated feature that forces an air mass up will be a factor in the generation of precipitation as well. The Sweet Grass Hills are very small features compared to the large plateau of the Cypress Hills. Therefore frontal air masses at the Sweet Grass Hills may not be elevated for a sufficient length of time, or in sufficient mass, to generate the amount of extra precipitation that elevation alone would suggest we should expect. On the other hand, note must also be taken of the fact that the 1961-1990 CMI normals for Minnesota, North Dakota and Montana are likely somewhat underestimated owing to climate station measurement issues detailed in appendix E. Taking all these factors together, our judgement is that by using the calculated elevation coefficient of 5.8 to calculate CMI values for the Cypress Hills and the Sweet Grass Hills we are again being conservative (particularly in the case of the summit areas of the Sweet Grass Hills) in not overestimating current and future aridity at these 2 sites.

Once elevation coefficients had been calculated, it was then necessary to interpolate a CMI for each study site at some base elevation. The most convenient calculating base is zero elevation (i.e. sea level). The calculation was as follows:

a) the 2 closest climate stations in each of the 4 quadrant directions (NW, NE, SW and SE) were identified;
b) the CMI at each of these 8 climate stations was converted to its estimated value at sea level using the appropriate study site elevation coefficient; and,
c) the value of CMI at sea level was estimated for each island forest study site location by inverse-square distance weighting of the sea-level CMI values at the 8 surrounding climate stations.

Thereafter, based on the relevant value of CMI at sea level and the relevant elevation coefficient, estimated values of CMI were calculated for each of the study sites across the actual range of elevation present at each site. The results are displayed visually by the study site elevation bar charts which accompany the climate normals map and each scenario map (Figures 4 through 13 in chapter 5). Table 7 in chapter 5 provides a numerical summary of the expected CMI values around the maximum elevation point of each study site island forest.

APPENDIX G – CRITERIA GOVERNING THE CONSTRUCTION OF CLIMATE CHANGE SCENARIOS

Criteria governing the construction of scenarios of climate change were outlined initially by Smith and Hulme (1998) and then more fully in the TGCIA guidelines (IPCC-TGCIA 1999). They are:

- **Consistency with global projections**: scenarios should be within the broad range of climate change projections based on increased atmospheric concentrations of greenhouse gases. Most recently, globally-averaged surface air temperature is projected to increase by 1.4 to 5.8°C over the period 1990 to 2100 (IPCC 2001) compared with the 1.0 to 3.5°C range proposed in the earlier IPCC *Second Assessment Report* (IPCC 1996). The wider range and higher projected temperatures in the latest estimate are due primarily to lower projected sulphur dioxide emissions in the most recent emissions scenarios.
- **Physical plausibility**: scenarios should be consistent with the physical laws that govern climate. Changes in one region should be physically consistent with those in another region as well as with those at the global scale. Climate variables are often correlated with one another, so changes in one variable should be reflected by changes in related variables. For example, changes in sunshine duration should be reflected by changes in cloud amount which, in turn, may result in changes in precipitation amount.
- **Applicability in impacts assessments**: scenarios should describe changes in a sufficient number of climate variables at spatial and temporal scales that allow for climate impact assessments.
- **Representativeness**: scenarios should be representative of the potential range of future regional climate change so as to allow a realistic assessment of the possible impacts. For example, a set of scenarios which examines only wet or dry conditions will not help identify the full range of sensitivities to climate change.
- **Accessibility**: scenarios should be straightforward to obtain, interpret and apply in impacts assessments.

Although there are a number of methods for constructing scenarios of climate change, including synthetic or arbitrary approaches and temporal and spatial analogue techniques, GCMs "…are the only credible tools currently available for simulating the response of the global climate system to increasing greenhouse gas concentrations" (IPCC-TGCIA 1999). In addition to outlining criteria to which climate change scenarios should conform, the IPCC-TGCIA have also outlined recommendations for selecting the GCMs to be used for scenario construction, again based on criteria first put forward by Smith and Hulme (1998). They are:

- **Vintage**: more recent model simulations are more likely to be more reliable than those of an earlier vintage, since recent models are based on up-to-date knowledge, incorporate more processes and feedbacks, and are usually of a higher spatial resolution than earlier models.
- **Resolution**: over time GCM resolution has increased, partly due to advances in computing technology. More recent models have horizontal spatial resolutions on the order of 250 km and have about 20 vertical levels, compared with a resolution of about 1000 km and between 2 and 10 vertical levels in earlier GCMs. Although the more recent models contain more spatial detail, including, for example, better-defined land/sea boundaries and more complex topography, superior model performance is not guaranteed.
- **Validity**: probably the most persuasive argument for GCM selection is that of model performance, i.e. selection of GCMs which simulate the present-day climate most faithfully. It is generally assumed that these GCMs will also yield the most reliable representation of future climate. Although statistical methods can be used to compare model simulations of the current baseline period with observed climate data, the choice of GCM is still rather subjective and will depend very much upon the region of interest, particularly the size of the region, and the climate variables being analysed. It has been suggested that rather than trying to identify the GCM which simulates current climate the most accurately, a better approach may be to identify those models whose performance is unacceptably poor, particularly in the estimation of climatic features which are of critical importance to the impact application.

- **Representativeness of Results**: it is strongly recommended that more than one GCM be used in any impacts assessment and that the selected GCMs should show a range of changes in the key climate variables in the study region. At the regional level GCMs can display a large difference in their estimates of climate change, particularly for variables like precipitation, where one model may indicate wetter conditions whilst another may show significant drying. It is important to try and capture this range of possible future climate conditions in any assessment of climate change impacts.